HABITAT UTILISATION BY JUVENILE FISHES IN MHLATHUZE ESTUARY AND RICHARDS BAY HARBOUR

by

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Submitted to the Faculty of Science
in fulfilment of the requirements for the degree of
Master of Science
in the
Department of Zoology
University of Zululand

2002

January 2002 Supervisor: Professor D.P. Cyrus

Declaration

I hereby declare that this whole thesis is my own original work, and to the best of my knowledge, it contains no material previously published or written by another person nor material submitted in any form for the award of any other degree at another university. Where use was made of the work of others it has been duly acknowledged in the text.

Signed:

Steven Paul Weerts

Acknowledgements

I have purposely kept these acknowledgements short, not because the efforts of others in assisting me with this work were not commendable or appreciated, but rather in the hope that they will be read.

This project was undertaken with the financial support of the National Ports Authority (previously PortNet), the Coastal Research Unit of Zululand and the University of Zululand Research Committee.

I am indebted to my supervisor, Prof. Digby Cyrus for the opportunity to do this research and advice throughout. Several employees of the Coastal Research Unit of Zululand assisted with the fieldwork and I am also grateful to Leon Vivier of the Department of Zoology, University of Zululand, for the Afrikaans translation of the abstract. Dr Shael Harris confirmed identifications of several particularly young specimens.

I am fortunate in having two people close to me, Fiona MacKay and Kevin Weerts, who are estuarine scientists and were able to read drafts of sections of this work with a professional eye. Errors that remain bear no reflection on either of them. Fiona assisted with figures and improved the presentation of this manuscript. Her own enthusiasm for estuarine research was a constant source of encouragement and she was unfailingly supportive of me throughout this work. For this I am sincerely grateful.

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Units of measurement and abbreviations used

‰	parts per thousand
%	percent
mg/l	milligrams per litre
°C	degrees Celcius
km	kilometre
m	metre
mm	millimetre
NTU	Nephelometric Turbidity Units
CRUZ	Coastal Research Unit of Zululand
SL	Standard Length
ME	Mhlathuze Estuary
RBH	Richards Bay Harbour
IUCN	International Union for Conservation of Nature and Natural Resources

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ABSTRACT

Estuaries in South Africa are important nursery areas for many fish species, but this role has been threatened by habitat degradation due to anthropogenic impacts, especially in KwaZulu-Natal systems. This has emphasised the importance of harbours in the province for juvenile estuarine fishes, but as areas of ongoing development the future ecological value of harbours is not guaranteed. Descriptions of shallow water estuarine habitats and an assessment of their utilisation by juvenile fishes is important if fish stocks are to be effectively managed for conservation purposes, and recreational and subsistence fisheries. The aim of this study was to investigate assemblages of juvenile fishes associated with a range of habitats in Mhlathuze Estuary and the adjacent Richards Bay Harbour. Habitats in these systems are typical of those in permanently open KwaZulu-Natal estuaries, and by virtue of their size they bestow considerable conservation status on the estuary and the harbour.

Fish ranging from postlarval to adult were sampled using a fine-mesh seine net. Endemic and conservation worthy species, as well as fishes valued in recreational and subsistence fisheries were recorded in both systems. Distinct assemblages of fishes were associated with different habitats. Differences in species compositions were largely due to habitat structure rather than the influences of physico-chemical variables. Eelgrass, *Zostera capensis*, is an important nursery for many fishes not found in other habitats. Presently in KwaZulu-Natal *Zostera* only occurs in Mhlathuze Estuary and St Lucia. Fish diversity in several other systems where it historically occurred may now be limited by its absence. Deep access channels, clear waters and an abundance of prey items render the harbour an attractive foraging ground for piscivorous fishes. Mangroves in this system play an important role as predation refugia for high densities of small fishes. Other habitats in both systems, including mudflats and sandbanks were also important. Many species showed a high preference for only one habitat type and the loss of any habitat in the estuary or the harbour would reduce biodiversity and the value of these systems as nursery areas.

Comparison of fishes associated with structurally similar habitats in different systems indicated that physico-chemical variables do influence the role of habitat in estuarine ecosystems. Sandbanks in the clear waters of the harbour supported different assemblages of fishes from sandbanks in the more turbid estuary. This could be related to turbidity preferences of selected species as well as higher predation threat in the harbour. Mangroves in both systems also supported different fish communities, probably also as a result of different predation risk and therefore indirectly attributable to turbidity differences in the two systems.

Results from this study highlight the conservation importance of the systems studied. Whilst it is probable that a multitude of species-specific responses to physical and biological factors influence fish assemblages associated with different habitats, results are broadly applicable to other KwaZulu-Natal estuaries. The distinct roles of structurally different habitats, and the influence of physico-chemical conditions, in the nursery function to different fish assemblages, illustrate the need for clearly stated management objectives at a provincial level.

SAMEVATTING

Getyriviere in Suid Afrika is belangrike kweekareas vir baie visspesies, maar hierdie rol word bedreig deur habitat degradasie veroorsaak deur menslike invloede, veral in sisteme in KwaZulu-Natal. Dit het die belangrikheid van hawens in die provinsie vir onvolwasse visse in getyriviere beklemtoon. Vanweë volgehoue ontwikkeling, is die ekologiese behoud van hawens egter nie verseker nie. Die beskrywing van vlakwater getyrivierhabitatte en 'n beraming van die gebruik van sulke gebiede deur klein en onvolwasse visse is van groot belang vir die effektiewe bestuur van visbevolkings vir bewarings doeleindes en vir rekreatiewe en bestaans visserye. Hierdie studie het ten doel die bestudering van die klein en onvolwasse vis gemeenskappe in 'n verskeidemheid van habitatte in die Mhlatuze getyrivier en die aanliggende Richardsbaai hawe. Die habitatte in hierdie twee sisteme is tipies van permanente oop getyriviere in KwaZulu-Natal. Vanweë hulle uitsonderlike grootte, verleen hierdie habitatte besondere bewaringsstatus aan die getyrivier en die hawe.

Vis wat wissel in grootte van post-larvaal tot volwasse is versamel met behulp van 'n fynmaas treknet. Endemiese spesies sowel as vis van bewarings, rekreatiewe en bestaans waarde is aangeteken in beide sisteme. Eiesoortige gemeenskappe is gevind in verskillende habitat tipes. Verskille in spesies samestelling was hoofsaaklik te wyte aan verskille in habitat struktuur eerder as die invloed van fisiese-chemiese toestande. Palinggras, *Zostera capensis*, speel 'n belangrike rol in die ontwikkeling van baie visse wat nie in ander habitatte voorkom nie. Tans word *Zostera* in KwaZulu-Natal net in die Mhlatuze getyrivier en in die St Luciameer aangetref. Vis diversiteit in ander sisteme waar dit histories aangeteken is, mag tans beperk word deur die afwesigheid van *Zostera*. Diep toegangskanale, helder water en volop prooi maak die hawe 'n aantreklike voedingsgebied for visvretende visspesies. Mangrove in die hawe speel dus 'n belangrike rol as 'n toevlug teen predasie vir 'n groot versameling klein vissies. Ander habitat tipes in beide sisteme, insluitend modder- en sandbanke, vervul ook 'n belangrik funksie in hierdie verband. Baie spesies het 'n spesifieke voorkeur slegs vir een habitat tipe.

Die verlies van enige habitat tipe in die getyrivier of die hawe sal dus die biodiversiteit en die waarde van hierdie sisteme as kweekareas aansienlik verlaag.

'n Vergelyking met vis geassosieer met struktureel soortgelyke habitatte in verskillende sisteme het aangedui dat fisiese-chemiese eienskappe wel die rol van habitatte in getyriviere beinvloed. Sankbanke in die helder water van die hawe het vis gemeenskappe onderhou wat verskil van die wat voorkom by sankbanke in die meer troebel water van die getyrivier. Dit kan verband hou met die turbiditeitsvoorkeur van sekere spesies sowel as 'n hoër predasie druk in die hawe. Mangrove in die twee sisteme het ook verskil met betrekking tot die gemeenskappe wat hulle onderhou, waarskynheid vanweë 'n verskil in predasie risiko, wat gevolglik indirek toegeskryf kan word aan turbiditeitsverskille in die twee sisteme.

Voortvloeiend uit hierdie studie is dit duidelik dat die bewaring van hierdie sisteme van groot belang is. Al is dit waarskynlik dat 'n verskeidenheid spesie-spesifieke reaksies tot fisiese en chemiese faktore die vis gemeenskappe, wat met verskillende habitatte geassosieer word, mag beinvloed, is die resultate tot 'n groot mate toepaslik tot ander KwaZulu-Natal getyriviere. Die bepaalde rol van verskillende habitatte, en die invloed van fisiese-chemiese toestande in die ontwikkeling vir verskillende vis gemeenskappe, illustreer die behoefte aan duidelike bestuursdoelwitte op 'n provinsiale vlak.

CHAPTER 1

Introduction

Chapter 1

Introduction

1.1 Estuaries of KwaZulu-Natal: Definition, importance to fishes and status

Estuaries are regions of fluvio-marine interface. In a South African context they have been defined by Day (1980) as partially enclosed bodies of water which are either permanently or periodically open to the sea, and within which there is a measurable variation of salinity due to the mixture of seawater with freshwater derived from land drainage. By this definition estuaries may include a wide variety of coastal water bodies but certain generalisations are true in the southern African context, especially of the subtropical systems of KwaZulu-Natal. Estuaries here, with few exceptions, are shallow, turbid and sheltered from wave action.

As with estuaries elsewhere, physical and chemical conditions are often highly variable in seasonal, daily and tidal cycles (Day 1981a, Ketchum 1983). Animals that utilise these systems implicitly face high energetic costs in adapting to the physiological pressures imposed upon them (Yáñez-Arancibia 1985). However, estuaries are also characterised by exceptionally high levels of primary and secondary production (Day *et al.* 1989) and often support a great abundance of organisms. Therefore, fish species that are tolerant of the biotic and abiotic variability in estuaries, are able to exploit rich food reserves in the absence of competition or predation by species which are not (Whitfield 1998).

Estuarine fish communities typically comprise of marine and freshwater forms as well as species that breed, and complete their life cycles within estuaries. Fish speciation in estuaries has been less extensive than in other aquatic habitats and estuarine breeders usually represent a relatively small part of the fish fauna (Whitfield 1998). These species are usually not very mobile and are commonly small-bodied, reaching sexual maturity at lengths less than 70 mm (Whitfield 1990, 1994a). Marine and freshwater fishes that enter estuaries are generally more mobile and often free to move between these brackish water environments and more stable marine or riverine environments. However, relatively few marine or freshwater fishes make regular use of estuarine habitats compared with marine shelf waters or fresh inland waters. Of an approximate 1,500 fish species that occur in South African continental shelf waters (Wallace *et al.* 1984), 155 associate with estuaries and only 50% of these may be regarded as having a strong association (Whitfield 1998). Only 11 of the 270 recorded freshwater species (Skelton 1993) enter estuaries, and four of these, all freshwater eels, merely pass through these coastal systems between freshwater and marine habitats (Whitfield 1998). This relatively low species diversity is a feature shared by estuaries in many parts of the world (Haedrich 1983, Whitfield 1994b).

The affinities of most important families found in estuaries are marine, and of the three broad categories of fishes that occur in estuaries, marine types dominate by species and abundance (Haedrich 1983). The main feature of this component in South African estuaries is that the juveniles are predominantly estuarine, while the adults are primarily marine (Wallace 1975a, Whitfield & Marais 1999). In fact, the great majority of fishes in estuaries the world over are usually juveniles of euryhaline marine species (Haedrich 1983, Potter *et al.* 1990). It is therefore not surprising that the major significance of estuaries has often been attributed to their role as nurseries to fishes. Juvenile fishes probably benefit most from conditions of reduced risk of predation and elevated food abundance offered by estuarine habitats. By virtue of their smaller size and weaker swimming capabilities they are potentially at higher risk of being preyed upon than larger fishes. At this stage of their life cycles, rapid growth out of critically small size classes is important for many species.

The degree to which fishes are dependent on estuaries differs among species. Whitfield (1998) categorised fishes commonly occurring in southern African estuaries on the basis of their association with different aquatic environments, as well as the degree to which they depend upon these systems (Table 1.1).

Approximately half of the 155 indigenous species occurring in southern African estuaries may be regarded as having a strong estuarine association (categories Ia, Ib, IIa, IIb and Vb; Whitfield 1998). These species do not support commercial fisheries to the same degree as those occurring in estuarine habitats in other parts of the world (e.g. the hilsa, *Tenualosa ilisha*, fishery in India and Bangladesh; Blaber 1997). However, subsistence fishing is important in some regions of South Africa. In northern KwaZulu-Natal, for example, traditional traps made from mangrove poles and branches have been used for several centuries in the Kosi system to harvest fishes. Over 90% of the fishes caught in these traps are marine spawners that recruit into the system as a nursery area (Kyle 1995). Several estuarine dependent fishes are also important as recreational angling species (targeted in both estuaries and coastal waters) and therefore have commercial value in terms of generating tourism (rental of holiday accommodations) as well as sales of equipment (fishing gear, boats and outboard engines) (Wallace *et al.* 1984).

Table 1.1: Estuarine association categories (EAC's) of fishes that utilise southern African estuaries (from Whitfield 1998).

EAC	Description
I	Estuarine species which breed in estuaries:
Ia	Resident species which have not been recorded spawning in marine or freshwater environments
Ib	Resident species which have been recorded spawning in marine or freshwater environments
II	Euryhaline marine species which breed at sea but with juveniles that show varying degrees of dependence on estuaries:
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. Includes some species which may breed in both freshwater and estuarine environments
V	Obligate catadromous species which use estuaries as transit routes between the marine and freshwater environments:
Va	Obligate catadromous species which require a freshwater phase for their development
Vb	Facultative catadromous species which do not require a freshwater phase for their development

Beyond their importance in subsistence and recreational fisheries, the fish assemblages in southern African estuaries have components of significant conservation value. Thirty-eight species are endemic (i.e. only recorded on the African continent and/or adjacent waters south of 20°S; Whitfield 1998) and 14 are classified by the International Union for Conservation of Nature and Natural Resources (IUCN) as endangered or threatened (Hilton-Taylor 2000). Eleven of these endangered or threatened species are restricted in southern Africa to subtropical estuaries and occur most frequently in KwaZulu-Natal. Ichthyofauna diversity in these estuaries is higher than that in the temperate systems to the south (Whitfield 1998). Most fishes in KwaZulu-Natal systems are of tropical or subtropical Indo-Pacific origin and the east coast of South Africa represents a subtraction zone in their distribution (Wallace 1975a). The occurrence of these species southwards, and that of more temperate forms northwards, fluctuates with season (Blaber 1981, Whitfield 1998). Seventy-two of the 142 species that occur in subtropical systems are strongly associated with estuaries (Whitfield 1998). In other parts of the Indo-Pacific, where precipitation exceeds evaporation, high runoff results in estuarine zones extending onto wide, shallow continental shelf areas (Haedrich 1983). These waters are extensively used by estuarine fishes (Blaber 1997). However, in South Africa the brackish, shallow and turbid conditions which permit the nursery role of estuaries, are not found in the inshore waters of the narrow continental shelf (Blaber 1981). Furthermore, available knowledge of ocean currents off the KwaZulu-Natal coast and reproductive biologies of its estuarine fishes, suggests that larvae spawned in local nearshore marine waters are retained in these waters by gyres and counter currents. Thus, recruitment of fishes from subtropical and tropical areas north of South Africa is probably limited, and KwaZulu-Natal estuaries may be largely reliant on locally spawned recruits (Wallace & van der Elst 1975). These estuaries therefore represent habitats that are essential for local populations of estuarine-dependent species near the southern limits of their distribution ranges. The status of these estuaries, and their ability to function as nursery areas has important implications for population numbers of estuarine fish species in KwaZulu-Natal.

Many reviews of the condition and biological status of estuaries in South Africa indicate degradation, primarily as a result of anthropogenic influences (Day &

Grindley 1981, Cyrus 1991). Habitat destruction, mainly due to land use changes and hydrological manipulations (fresh water abstraction) is the dominant threat to fishes in these estuaries, although organic and inorganic pollution are becoming increasingly prevalent, as is possible overfishing in the case of certain species (Whitfield 1997). Impacts upon KwaZulu-Natal estuaries have resulted from sedimentation, eutrophication, bridge and causeway construction, building encroachment and abstraction of freshwater inflow (Begg 1978, 1984, Heydorn 1986). Only three of 73 KwaZulu-Natal estuaries studied by Begg (1978) in the late 1970's were regarded as "problem free" and impoverished faunas were principally due to indirect impacts through habitat degradation. Biotic surveys conducted in the commercial harbours of Durban and Richards Bay have indicted that both these systems sustain considerable diversities of mostly juvenile fishes, and that they still function as nurseries despite being highly modified estuarine habitats (Cyrus & Forbes 1996, Forbes *et al.* 1996). Their large size relative to the majority of estuaries in KwaZulu-Natal therefore renders them important to estuarine fishes, but as areas of ongoing development their continued contribution in offering a variety of habitat types is not guaranteed.

Concern regarding the degradation of fish habitat, and consequent loss of biological diversity and ecosystem function, has led to a realisation that the traditional species-by-species approach to conservation and management must be supplemented by efforts that consider whole ecosystems and their natural communities (Allee *et al.* 2000). In the United States of America (USA) legislation has been promulgated pertaining to the identification and protection of "Essential Fish Habitat", and the implementation of these various acts is a major activity of the National Marine Fisheries Service. However, although habitat is recognised as fundamentally important in providing the basic needs of any organism, the use of the term by researchers and managers is highly variable. In a coastal context, fish habitat has been defined at various levels ranging from entire estuaries and bays to channels, seagrass beds and mudflats. Use in this general sense to suit specific needs has been regarded as acceptable, but in attempting to provide an understanding of the role of habitat within ecological systems, Peters and Cross

(1992) advocated the framework suggested by Ryder and Kerr (1989), which stressed the importance of environmental structure. Habitat is defined as the structural component of the environment that attracts fishes, and is separated from other environmental factors such as temperature, oxygen content and pH. These non-structural (environmental) properties may have limiting or controlling effects on fishes and therefore influence the value, or use of a structural feature (Peters & Cross 1992). In a broad hierarchical classification of ecosystems and habitats, Allee *et al.* (2000) made provision to refine "eco-types" using a wide range of "local modifiers" to describe their lowest level of categorisation, the "eco-unit". Thus a salt marsh (eco-type) could be modified by local conditions of coastal geomorphology, topography, elevation, coastal hydrology and exposure (local modifiers) to produce functionally different eco-units, such as fringing, riverine, brackish or upland transition marshes.

The main elements of habitat structure in subtropical South African systems are substratum type and vegetation. Estuaries here are dominated by soft substrata, ranging from mud to sand. In open subtropical systems, vegetation mainly comprises of mangroves along intertidal banks. Submerged aquatic macrophytes, such as eelgrass *Zostera capensis*, occur less frequently. Shallow areas that support marginal and submerged aquatic vegetation have been identified as particularly favourable habitats for juvenile fishes in KwaZulu-Natal estuaries (Wallace & van der Elst 1975). However, studies conducted elsewhere have shown that some species have preferences for unvegetated habitats (Gray *et al.* 1996, Jenkins *et al.* 1997). Furthermore, juveniles of certain species may favour more than one habitat type with growth and development. As the needs of fishes change with life stage, so their habitat requirements might be expected to change. Thus, some species have been noted to migrate through several habitats in the period between recruitment into sheltered waters and emigration back into the marine environment proper (Potter *et al.* 1983, Rozas & Hackney 1984, Hyndes *et al.* 1997, Hannan & Williams 1998).

Important physico-chemical factors influencing fish distribution and abundance within South African estuaries have been identified [e.g. turbidity (Cyrus & Blaber 1987a,b,c), salinity and freshwater inflow (Whitfield *et al.* 1981; Whitfield 1994c; Ter Morshuizen *et al.* 1996) and salinity-temperature effects (Martin 1988)]. Some research has compared influences of different habitat structure in terms of reef or submerged aquatic vegetation (Blaber 1978, Branch & Grindley 1979, Beckley 1983, Hanekom & Baird 1984, Whitfield 1986, Paterson & Whitfield 2000). However, most work has targeted late juvenile and adult stage fishes and comparatively little has documented responses of late larval, or early juvenile stages. Habitat utilisation by fishes in size classes below 50 mm has generally not been well studied, even though the majority of species utilising South African estuaries recruit into systems at lengths well below this (Wallace & van der Elst 1975, Harris & Cyrus 1995, Whitfield 1998). These early life stages are clearly critical in the life cycles of estuarine associated fishes and it is important that their habitat requirements are recognised, especially as different juvenile stages may be adapted to very different environmental conditions compared with adults (Day *et al.* 1989).

1.2 Study aims

The broad aim of the project was to investigate the occurrence of newly recruited, early juvenile and small fishes in a range of shallow water habitats in Mhlathuze Estuary and Richards Bay Harbour. Small-bodied fish assemblages associated with different habitats were to be identified and assessed in terms of their dependence on estuarine environments. It was hoped that the results would contribute to a greater understanding of the role of habitat during largely unstudied life cycle stages of estuarine fishes. This would allow for more informed decision-making in the management of estuaries and harbours in KwaZulu-Natal. In this regard it was deemed important that results from this study be disseminated to the scientific community as well as estuarine and harbour managers. This has already been partially accomplished as results have been presented at two international conferences and published in a scientific journal as outlined below. Further publications are envisaged and a meeting with managers of the National Ports Authority is planned.

- Weerts, S.P. and Cyrus, D.P. (1999). Preliminary observations on habitat utilisation by post-larval fishes in a marine dominated estuary and adjacent harbour at Richards Bay, South Africa. Poster presentation at: the 23rd Annual Larval Fish Conference. National Marine Fisheries Service and Fisheries Society of the USA. Beaufort, North Carolina, USA.
- Weerts, S.P. and Cyrus, D.P. (2001). Habitat utilisation by young and small-sized fishes in a subtropical South African estuary and adjacent harbour. Oral presentation at: the 6th Indo-Pacific Fish Conference. Oceanographic Research Institute Annual. Durban, South Africa.
- Weerts S.P. and Cyrus, D.P. (2002). Occurrence of young and small-sized fishes in different habitats within a subtropical South African estuary and adjacent harbour. *Marine and Freshwater Research* 53: 447-456.

1.3 Thesis structure

Chapter 2 provides broad background information pertaining to estuaries in KwaZulu-Natal and describes the systems sampled during this study and their catchments. Details of different habitat types are given and their importance in a subtropical South African context is discussed. Methods and materials used in fieldwork, laboratory and statistical analyses are explained in Chapter 3. Results and discussions of findings from individual systems are dealt with separately in Chapter 4 (Mhlathuze Estuary) and Chapter 5 (Richards Bay Harbour). Direct comparisons of these two systems and the roles of similar habitats within them are made in Chapter 6. Finally, some general conclusions are drawn from the study in Chapter 7, and their potential application to management of KwaZulu-Natal estuaries and harbours is discussed.

CHAPTER 2

Study Area

Chapter 2

Study Area

2.1 Regional considerations

The province of KwaZulu-Natal is situated on the east coast of South Africa. The coastline of approximately 570 km has a reasonably straight configuration and an axis of orientation approximately south-west/north-east (Figure 2.1). The climate is subtropical and rivers are perennial and subject to summer rainfall maxima. Major estuarine systems along the coast are Mhlathuze Estuary, St Lucia and the Kosi Lakes. Durban and Richards Bay harbours, both originally large estuarine embayments, have been highly modified as commercial ports. These systems nevertheless play an important role to a wide array of typically estuarine species (Cyrus & Forbes 1996, Forbes *et al.* 1996, Weerts *et al.* in prep.) and due to their large size are still regarded as important estuarine areas. Most of the remaining systems are small and have areas of less than 100 ha (Ramm 1990).

Begg (1978) categorised KwaZulu-Natal estuaries into two major types, estuaries or lagoons, based mainly on the presence or absence of a free connection to the sea and their domination by either saline- or freshwater respectively. However, "lagoons" often undergo periods of exhibiting classic estuarine characteristics after breaching their sandbars (Begg 1978). This fluctuating nature to behave like different systems at different times depending on freshwater inflow and the nature of the mouth, is true of many smaller southern African estuaries (Day 1981b, Whitfield 1992). Despite this Whitfield (1992) provided a comprehensive classification of southern African estuaries which accounted for a variety of characteristics of the dominant phase prevailing in each estuary. By this classification, system types can be categorised as estuarine bays, permanently open estuaries, river mouths, estuarine lakes and temporarily open/closed estuaries.

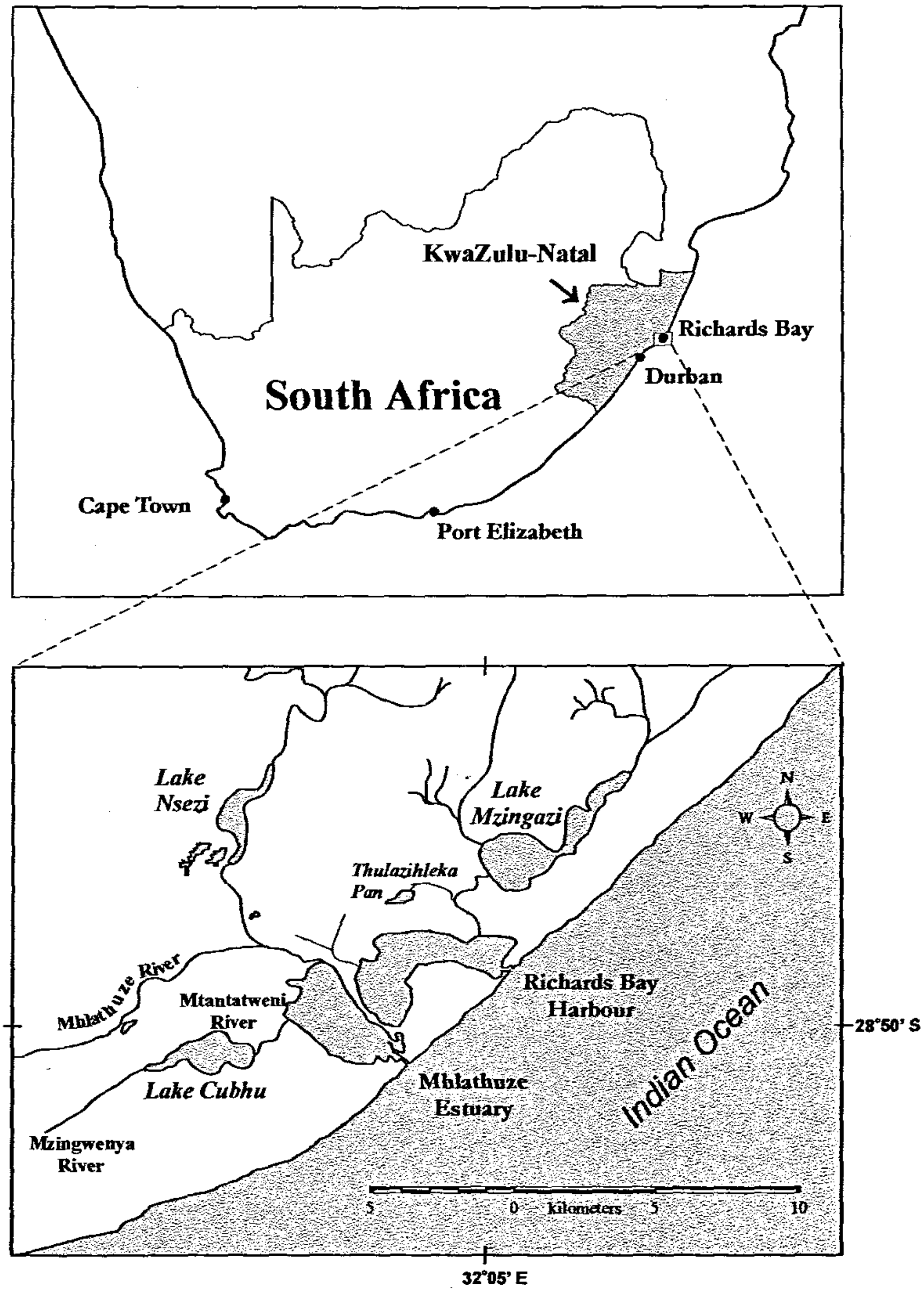


Figure 2.1: Locality of study systems and other significant aquatic environments in their vicinity.

Representatives of all five types of estuary are found along the coast of KwaZulu-Natal. Durban and Richards Bay Harbours are estuarine bays whilst the Mhlathuze is a permanently open estuary. The Thukela is a river mouth for most of the year and Lake St Lucia and the Kosi Lakes are estuarine lakes. By far the majority (>75%) of the KwaZulu-Natal's 73 systems from Kosi Bay in the north to the Mtamvuna in the south are small temporarily open/closed estuaries (Begg 1978).

Cooper *et al.* (1999) recognised southern and northern KwaZulu-Natal as distinct geomorphological zones in terms of major controls on the morphology of estuaries. Hinterlands south of the Thukela are steep and narrow and the coastal geomorphology is characterised mainly by bedrock-confined river valleys. North of the Thukela the coastal topography flattens and widens into the Zululand Coastal Plain, which extends northwards through the greater part of Mozambique. Sediments on this plain are composed of unconsolidated Tertiary to Pleistocene deposits with a thin Holocene veneer. Lateral erosion of these unconsolidated surface sediments has resulted in most northern KwaZulu-Natal estuaries having relatively large surface water areas but being comparatively shallow (Cooper *et al.* 1999).

Tides along the coast of KwaZulu-Natal are semidiurnal and maximum tidal range at Richards Bay is approximately 2 m. The entire coastline is subject to high energy waves (Day 1981c). The swift-flowing Agulhas current, a southward deflection of the Indian Ocean South Equatorial current, dominates the nature of nearshore marine waters. It lies closest to the coast north of Richards Bay and south of Port Shepstone (Bang *et al.* 1978) but seldom drifts nearer than about 5 km from the shore (Heydorn 1973). Nevertheless, its tropical origin has a profound influence on the distribution of marine and estuarine fauna on the KwaZulu-Natal coast (MacNae 1962, Heydorn 1973, Bang *et al.* 1978). Eddies generated off the western edge of the Agulhas, and the north-easterly movement of atmospheric low pressure systems along the coast, produce counter currents so that inshore water movements may often be in a northerly direction up the coast (Wallace 1975a).

Surface salinities in the Agulhas are typically 35.2 to 35.4‰ but inshore salinities can be less, especially in summer, due to local run-off from the land (Bang *et al.* 1978). Water temperatures recorded behind the surf line at Richards Bay during the study period ranged from a minimum of 18.7°C to a maximum of 26.0°C (Natal Sharks Board unpublished data). Average monthly temperatures are shown in Figure 2.2 and clearly indicate late summer maxima and late winter minima.

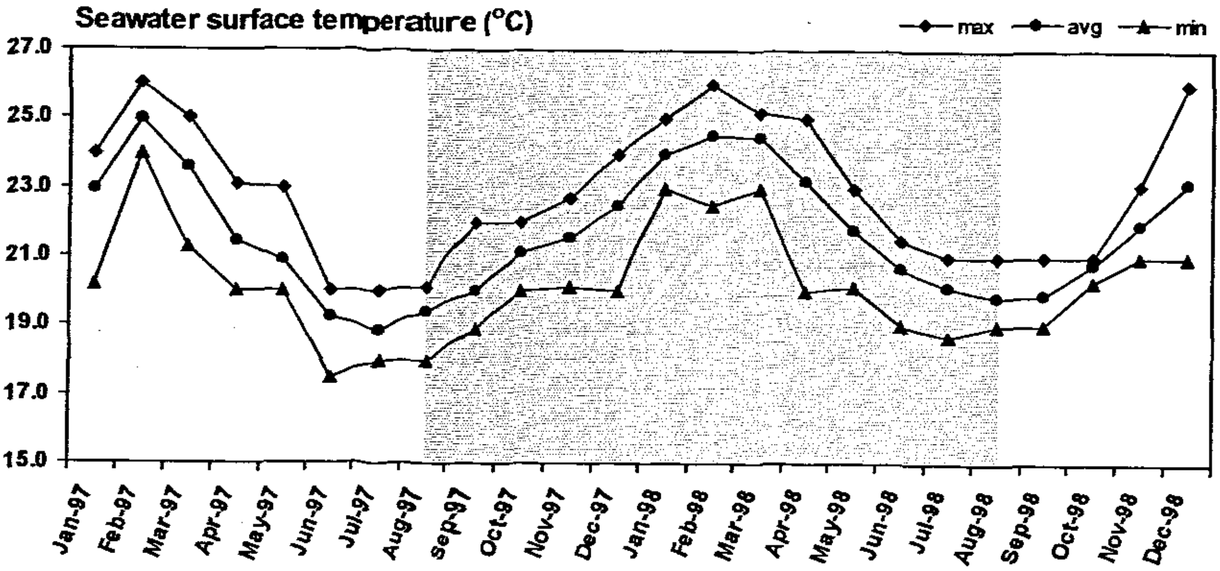


Figure 2.2: Sea surface water temperatures (°C) off Richards Bay swimming beach (data kindly supplied by the Natal Sharks Board). Shading indicates study period.

2.2 Location

In northern KwaZulu-Natal, Mhlathuze Estuary and Richards Bay Harbour are located approximately 160 km north-east of Durban, on the Zululand Coastal Plain close to its southern extremity. Long sandy beaches backed by high, forested dunes are characteristic of this stretch of the KwaZulu-Natal coast. In the vicinity of the study systems this dune belt is up to 1 km wide and 100 m high. Behind the dune ridge the terrain is level leading to gently sloping hills some 12 km inland (Bosman 1986). The climate is subtropical with warm to hot summers with a relatively low diurnal temperature range. Mean maximum and minimum temperatures during summer (December, January and February) are approximately 29°C and 21°C respectively. Winters (June, July and August) are mild with mean maximum and minimum temperatures of approximately 23°C and 12°C respectively. Although the area is a summer rainfall region significant falls may occur at any time of year (Table 2.1). Wind directions are typically from the north-east or south-west throughout the year.

Table 2.1: Mean climatic data Richards Bay, 1961 - 1990 (data supplied by the South African Weather Bureau).

Climatic variable	Month												
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Mean max. temp (°C)	29.2	28.9	28.9	27.0	24.8	23.1	23.0	24.0	24.9	25.4	26.7	28.7	26.2
Mean min. temp (°C)	21.2	21.2	20.4	18.1	15.2	12.3	12.3	14.1	16.0	17.3	18.6	20.4	17.3
Max. max. temp (°C)	41.3	39.0	36.5	36.9	35.2	34.5	31.4	37.0	40.0	42.2	42.5	41.6	42.5
Min. min. temp (°C)	10.5	13.0	13.5	7.8	6.9	6.0	4.0	5.1	5.7	9.5	11.4	12.9	4.0
Precipitation (ppt) (mm)	172	167	107	103	109	57	60	65	77	105	114	86	1228
ppt 24hr max (mm)	317	145	259	130	88	82	135	62	65	99	135	78	317
Rel. humidity @ 2pm (%)	70	71	71	71	63	61	59	59	66	67	70	69	67

Other significant aquatic systems in the lower Mhlathuze catchment besides the estuary, harbour and river, are the three coastal lakes, Nsezi, Cubhu and Mzingazi (Figure 2.1). All three of these lakes are regulated at their outflows by weirs and maintain a surface elevation above sea level. Their origins have been described by Hill (1975). Lake Nsezi is a drowned river valley. It has a surface area of 614 ha (Harding 2000), a muddy substratum and generally turbid water. It is fed by the Nseleni River and augmented via an intrabasin water transfer from the Mhlathuze River (Cyrus *et al.* 1999). Drainage via the Nseleni flows into the lower reaches of the Mhlathuze River close to the tidal and saline head of the estuary. Lake Cubhu, with a surface area of 450 ha (Harding 2000) is a relict estuarine lake with a substratum comprising of sand and detritus around the margins, and mud in the deeper central areas (Cyrus & Martin 1988). It drains into Mhlathuze Estuary via the Mtantatweni River. Mzingazi is also a relict estuarine lake with a substratum of mostly marine sands. With a surface area of 1,216 ha (Harding 2000) it is the largest of the three lakes and the only one that reaches depths below mean sea level in its basin (Kelbe & Germishyuse 2000). This has prompted the construction of a saltwater barrier across the draining Mzingazi River to prevent possible intrusion of saline water into the lake during drought years. Saline harbour water pushes up the Mzingazi Canal to the base of this saltwater barrier with the incoming tide.

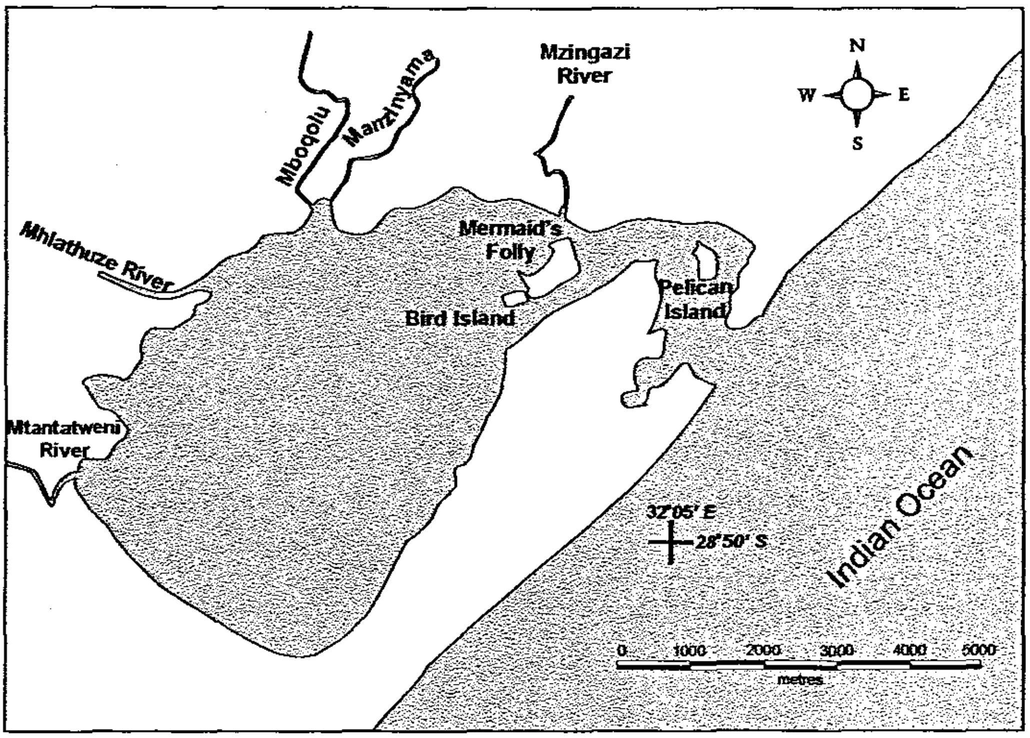
Of the three coastal lakes, Cubhu and Mzingazi have historically acted as significant nursery areas for estuarine associated fishes. This role has significantly declined in recent years due to the construction of weirs at their outlets prohibiting successful recruitment by most species (Weerts & Cyrus 2002). This has increased the regional importance of Mhlathuze Estuary and Richards Bay Harbour as nursery habitats for estuarine fishes (Weerts & Cyrus 1998).

2.3 History, harbour development, and conservation status

The British named Richards Bay after Sir F.W. Richards who landed troops there during the Zulu War in 1879. Before this early Portuguese navigators charted it as Rio dos Peixos ('river of many fish') and the Zulus referred to it as eChwebeni ('the place of the large sheet of water') (Begg 1978). The configuration of the original system is shown in Figure 2.3A. Millard and Harrison (1954) gave an account of the estuary as it was in 1949 to 1951. Covering about 3,100 ha it originally comprised two main parts, a broad shallow lake and a narrower channel to the sea. It was one of the most productive fishing resorts in KwaZulu-Natal with rich beds of *Zostera capensis* that supported abundant prawns and small fishes. At this time it was regarded to have suffered less extensive siltation than most other estuaries on the east coast of South Africa (Millard & Harrison 1954).

However, as early as 1955 it was anticipated that a new harbour, to handle bulk cargo such as coal, would be needed on South Africa's east coast to augment the port at Durban (Cooks & Bewsher 1993). The potential of Richards Bay as such a deep-water harbour had long been recognised and development commenced in

A) Richards Bay 1964 redrawn from Begg (1978)



B) Richards Bay and Mhlathuze Estuary 1997

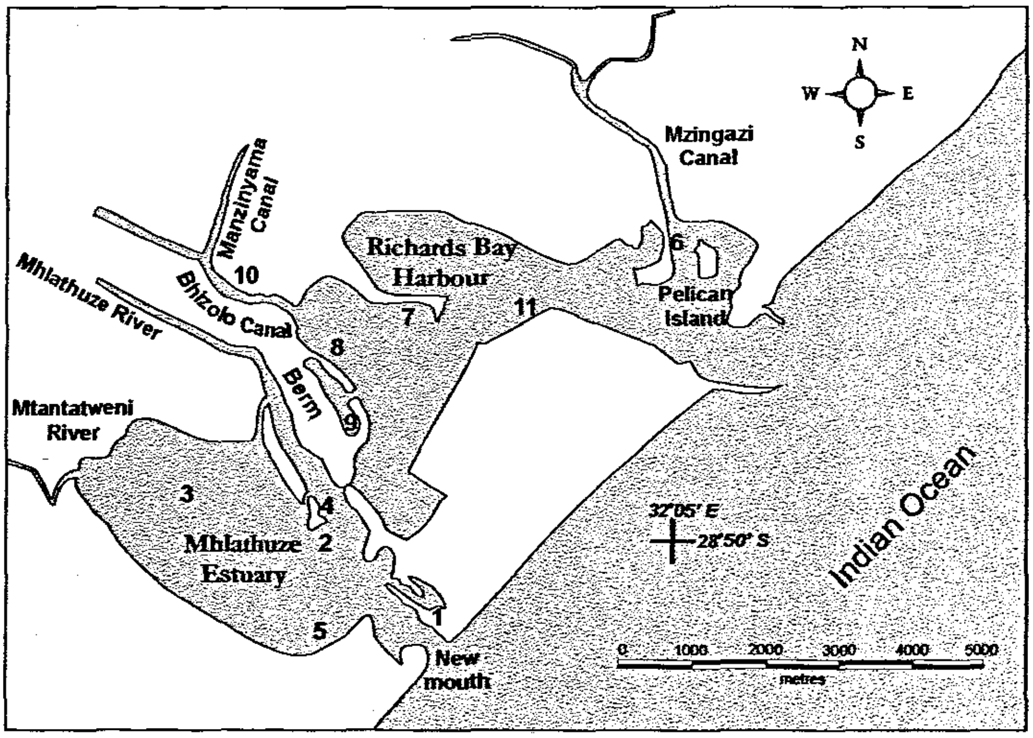


Figure 2.3: Study systems A) Richards Bay prior to harbour development and B) Richards Bay Harbour and Mhlathuze Estuary after harbour development with study sites indicated.

1972. This involved the division of the relatively undisturbed estuarine system into two sections by the construction of a 4 km long causeway or berm. The bay to the north-east was given over to harbour development and has grown into South Africa's largest harbour (Richards Bay Harbour). The remaining area (approximately 1,200 ha; Anon 1978) to the south-west of the berm was designated as a nature sanctuary (Mhlathuze Estuary) following the re-routing and canalisation of the lower Mhlathuze River, and cutting of a new estuary mouth (Begg 1978, Figure 2.3B). Aside from the obvious loss of habitat due to dredging, quay construction and other activities related to harbour development, hydrological changes, brought about by the alterations to the inflowing river and mouth, have had far reaching impacts upon the nature of the remaining sanctuary area.

Millard and Harrison (1954) warned of the potentially disastrous effects should the river be canalised and swamps drained in the lower Mhlathuze system. Their forecast proved correct and the major effect of canalisation of the river has been to increase the sediment input into the estuary (Begg 1978, Cooks *et al.* 1986, Cooks & Bewsher 1993). Papyrus (*Cyperus papyrus*) swamps on the original Mhlathuze delta acted as sponge in absorbing floodwaters and filtering silt. With the bypassing of these swamps by canalisation, and their eventual destruction, this filtering function was lost. A few years prior to harbour development the average silt load from Mhlathuze River was judged to be less than 20,500 m^3 per annum (Anon 1972). In 1993 sediment transport into the estuary was estimated as 237,000 m^3 per annum, 90% of which was silt/clay (Huizinga & van Niekerk 2000).

Whilst the increased sediment load of Mhlathuze River is accepted fact the contention that the estuary is rapidly silting up has recently been cautiously questioned. Much concern has been expressed regarding accelerated siltation of the system, especially subsequent to harbour development (Begg 1978). Cooks *et al.* (1986) predicted that under their estimated rate of siltation, the system might be completely silted by 2015. However, Huizinga and van Niekerk (2000) compared water level recordings made in 1997 with historical measurements, and maintained that some areas in the estuary have actually deepened since harbour development and the cutting of the new estuary mouth. They suggest that complete infilling of the Mhlathuze could take at least a few centuries. From the description of Millard and Harrison (1954) it is apparent that the area of the original system that now remains as a nature sanctuary historically comprised mainly of shallow mud flats. Huizinga and van Niekerk (2000) hypothesised that under wind and wave action, fine non-cohesive bottom material is brought into suspension, and under conditions of increased tidal exchange in the new system (after harbour development) much of these suspended sediments were flushed to sea on ebb tides. Furthermore, the configuration of the system subsequent to harbour development is such that most of the increased sediment influx from the river probably bypasses the estuarine embayment and is flushed directly out to sea.

The increase in tidal range in Mhlathuze Estuary after harbour development is well documented. The original system's mouth was shallow and narrow, and restricted tidal range in the estuarine lake sections to only 0.35 m (Millard & Harrison 1954). This increased to 1.8 m in the embayment once tidal currents had scoured a large new mouth from the original breach canal (Anon 1974). Freshwater swamp communities along much of the system's shoreline, which had never been exposed to flooding by saline water, suffered severe mortalities (Begg 1978). As a further consequence of the increased tidal range, extensive areas of shallow, subtidal mudflats became intertidal and were exposed at low tides. With the tidal inundation of high salinity water, this led to the colonisation and proliferation of white mangrove (*Avicennia marina*). These factors have probably both contributed to exaggerated estimates of sedimentation rates, especially those based on increases in mangrove coverage. Since 1975 the area of mangroves in the estuary has increased by 435%, from 150 ha to 652 ha (Adams *et al.* 2000). This represents over half the total mangrove habitat in South Africa and Mhlathuze Estuary was rated as the most important system in the country by the Botanical Importance Rating of Adams *et al.* (2000). Recent reviews of the system's ichthyofauna (Weerts & Cyrus 1998), macrobenthic fauna (MacKay & Cyrus 1998) and aquatic avifauna (Cyrus 1998) have also indicated the estuary's regional and national importance. The system has been afforded the status of Marine Protected

Area and is managed to permit recreational fishing (restricted to the mouth region) and limited subsistence utilisation. A considerable amount of illegal netting also occurs and it is possible that poaching is now reaching a level that may constitute a small-scale commercial fishery, with markets as far afield as Durban being served.

The area of the original system given over to harbour development also now experiences the full marine tidal range (Begg 1978) and mangrove proliferation has also occurred on suitable substrata. Richards Bay Harbour plays an important role in the life cycles of many marine species that associate with estuaries despite its highly modified nature (Cyrus & Forbes 1996, Forbes *et al.* 1996, Weerts *et al.* in prep.). Sampling of fishes by the Coastal Research Unit of Zululand (CRUZ) in 1996 and 1997 yielded more species in the harbour than in the adjacent estuary (CRUZ unpublished data) but this was due to the presence of more marine stragglers than estuarine species. Recreational angling in the harbour is almost entirely restricted to easily accessible areas at the mouth.

2.4 Habitat types in Mhlathuze Estuary and Richards Bay Harbour and their significance in KwaZulu-Natal estuaries

Mhlathuze Estuary and Richards Bay Harbour are important in that between them they offer almost the complete range of habitat types found in tidal reaches of subtropical South African estuaries. These include sandbanks, mudflats, mangroves (*Avicennia marina* and *Bruguiera gymnorrhiza*) and eelgrass (*Zostera capensis*). Saltmarshes are largely restricted to the more temperate systems in the south, but other submerged macrophytes and fringing vegetation do occur in KwaZulu-Natal systems. However, in open estuaries those in the form of reeds and sedges are most often restricted to the upper reaches and inflowing river channels where tidal exchange, salinities and water currents are reduced (Adams *et al.* 1999). Species of mangroves occurring in South Africa other than the two listed above are mostly restricted to the Kosi system in northern KwaZulu-Natal, or are found on the landward margin of mangrove swamps which are subject to tidal flooding only on high spring tides (Steinke 1999).

Eelgrass

Zostera capensis is the most widely distributed seagrass along the South African coast (Edgecumbe 1980). The species is a major component of estuaries in the southern Cape but *Zostera* is essentially a temperate zone genus and St Lucia represents the northern limit of its distribution on the east coat of South Africa (Howard-Willams 1980). Seagrass beds have been recognised as important habitats for juveniles of many fish species in South Africa (Beckley 1983, Whitfield *et al.* 1989) and worldwide (Pollard 1984). Immediately prior to harbour development at Richards Bay, *Zostera* covered approximately 63 ha above the mouth in the vicinity of Mermaid's Folly (EAS 1996, Figure 2.3A). Dredging of deep-water channels and basins during harbour construction obliterated these beds. The feasibility of re-establishing *Zostera* in the harbour was subsequently raised in various forums and specimens collected at St Lucia were transplanted to an area in the southern corner of the system in the early 1990's (L. Dermont pers. comm.). The species failed to establish and no subsequent attempts have been made. However, eelgrass was successfully introduced into the estuary in 1991 (R. Jones pers. comm.). Initially it was planted on the north bank in the vicinity of the mouth. This small bed soon died out but not before the species re-seeded south of the mouth. Subsequently, a strip running along the south bank of the embayment has become well established. In 1998 these beds covered approximately 5 ha (Adams *et al.* 2000), less than 8% of the area covered in the original system prior to harbour development.

Seagrasses are vulnerable to degradation by human activities (Adams *et al.* 1999) and while the infrequent occurrence of *Z. capensis* in subtropical South African estuaries is probably natural, its presence is becoming increasingly rare. In KwaZulu-Natal, *Zostera* has been reported from at least six systems [St Lucia, Richards Bay (pre-harbour), Mlalazi, Matigulu, Durban Bay and Mgababa (Begg

1978)], but it is now restricted to St Lucia and Mhlathuze Estuary. The beds at St Lucia are highly variable in spatial and temporal terms. During periods of low salinity *Zostera* dies back and *Ruppia* becomes more dominant and widespread (C. Ward pers. comm.). However, during optimal conditions *Zostera* covers a surface area of approximately 28 ha (R. Taylor pers. comm.). Eelgrass in the Mhlathuze also fluctuates in area but the major impact appears to be anthropogenic rather than natural. In 1996 the disposal of dredged spoil material on the beach between Richards Bay Harbour and Mhlathuze Estuary led to the intrusion of sediment-laden water via the estuary mouth (Cyrus *et al.* 2000). This caused major diebacks of *Zostera* in the estuary by increasing turbidities and depositing fine sediments on leaf blades, reducing the ability of the eelgrass to photosynthesise. Subsequent to the cessation of dredged spoil disposal these beds have proliferated and by mid-1999 they covered approximately 28.5 ha in the estuary (Cyrus *et al.* 2000). These beds therefore presently represent the most extensive area of *Zostera* in KwaZulu-Natal. However, the planned resumption of dredging activities in the Richards Bay Harbour threatens the continued existence of eelgrass in Mhlathuze Estuary. Given its vulnerability and rarity, *Zostera* is probably the most threatened of all estuarine habitat types in subtropical South Africa.

Mangroves

Whilst the area of eelgrass has decreased due to human activities in the Richards Bay area, the opposite is true of mangroves. The history of mangrove proliferation in both Mhlathuze Estuary and Richards Bay Harbour has already been discussed. However, only one species, the pioneer white mangrove, *A. marina*, has responded positively to anthropogenic intervention in these two systems. Red mangrove, *Rhizophora mucronata* and black mangrove, *Bruguiera gymnorrhiza*, were abundant in the original system (Millard & Harrison 1954) but both have been negatively impacted by the development of the harbour and associated infrastructure. The threat of localised extinction posed to remaining *Rhizophora* stands was recognised soon after harbour construction was completed (Begg 1978). Steinke (1999) gives all three species as being present in the greater

Richards Bay area, but the red mangrove occurs only as isolated specimens within stands dominated by the other types (*pers. obs*). The original system supported a flourishing community of *B. gymnorrhiza* (Begg 1978) but much of this was decimated during harbour construction, or died shortly after. Few original specimens are still found in the estuary and these appear to be stressed (*pers. obs*). Saplings have been observed in stands dominated by *Avicennia* and in *Phragmites* beds in the upper estuary, a habitat where the *Bruguiera* seedlings are known to develop particularly well (Berjak *et al.* 1997). In the harbour *Bruguiera* has persisted in two stands on either side on the entrance channel. The construction of a dry dock and associated harbour infrastructure will eventually destroy the northern stand. In recognition of the importance of the other, the National Ports Authority has motivated that the area be proclaimed a Natural Heritage Site. However, this proclamation is still forthcoming, and the conservation status of this mangrove stand has yet to be officially recognised. This is of particular concern as this will be the only stand of mangroves that predate port development remaining in the harbour, and the only representative *Bruguiera* mangrove forest left in the greater Richards Bay area.

The significance of mangrove habitat in Mhlathuze Estuary and Richards Bay Harbour is indicated by the fact that these two adjacent systems accommodate a greater area of mangroves than all other estuaries in South Africa combined (EAS 1996, Adams *et al.* 2000). As noted above mangroves in the estuary cover 652 ha (Adams *et al.* 2000). Approximately 17 km of shoreline is colonised with *Avicennia.* This equates to 14 ha of mangrove fringe as sampled during this study (assuming 8 m immediately adjacent to trees is fringe habitat). Mangroves in the harbour cover approximately 305 ha, of which over 85% comprises *Avicennia* which has colonised areas subsequent to harbour development (EAS 1996). Approximately 11.5 km of the perimeter is fronted with *Avicennia* (9.5 ha), but only 700 m (0.5 ha) comprises *Bruguiera* mangroves. It is noteworthy that only another five of twenty KwaZulu-Natal estuaries north of Durban Bay support mangroves (Steinke 1999). Destruction of mangroves, and consequent negative effects on fisheries have been reported from many subtropical and tropical regions worldwide (Blaber

1997). Significant reductions in South African mangroves have resulted from direct exploitation or indirectly through poor agriculture practices (Steinke 1999). Increasing demands for waterfront properties and coastal development further threaten this habitat type. Thus, while plentiful in a local context, mangroves at Mhlathuze Estuary and Richards Bay Harbour represent an important habitat along the South African coast.

Sandbanks

Intertidal sandbanks occur in the embayments and near the mouths of both the estuary and the harbour. These banks have remained free from colonisation by macrophytes, possibly due to their sediments being too low in organic content and/or too well drained. Naturally occurring sandbanks were extensive in the lower reaches and mouth of the original system prior to harbour development. Although these areas were lost to harbour construction, sandy areas in the harbour are probably now more extensive through the creation of artificial spits and infilling with sandy spoils (EAS 1996). Sandflats presently cover approximately 380 ha in the harbour. Conservatively only half of this area can be regarded as shallow water habitat less than 1.5 m in depth. In the estuary sandy substrata comprise an even smaller area. Open waters in this section of the original system were characterised by mud substrata (Millard & Harrison 1954). The high sediment load of Mhlathuze River and silt deposition has muddied much of the perimeter and permitted mangrove colonisation. Sandy areas are restricted to the lower embayment and mouth where tidal currents keep the substratum free of fine sediments. Thus only approximately 70 ha of shallow water habitat comprises sandy sediments in the estuary.

Despite the fact that soil erosion in catchments is recognised as a major threat to estuaries in KwaZulu-Natal (Morant & Quinn 1999) the conservation value of sandflats in these estuaries is underrated. Deposition of fine sediments obviously smothers sandy substrata and creates muddier estuaries. This in turn leads to changes in the abiotic and biotic character of affected systems (Morant & Quinn 1999). Significant components of typical subtropical estuarine ichthyofaunas may have distinct preferences or requirements for sandy substrata (Blaber 1997) and the loss of this habitat, to be replaced by a uniform layer of mud, causes a reduction in species diversity. The Kosi system is one of the only KwaZulu-Natal estuaries which has remained relatively free of sedimentation and has a littoral substratum comprising predominantly of sand. Despite its large size, Kosi has a small catchment, and almost all freshwater input derives from a high surface water table rather than precipitation borne by rivers (Wright *et al.* 1997). Extensive areas of intertidal sand which are found at Richards Bay and Durban Harbours therefore comprise a significant proportion of this habitat type in KwaZulu-Natal estuarine systems (Cyrus & Forbes 1996, Forbes *et al.* 1996).

Mudflats

Prior to harbour development mud was the main substratum type in Richards Bay basin south of Bird Island (Millard & Harrison 1954). However, little of this area was tidally exposed. After the widening and deepening of the original mouth for the harbour entrance channel, and cutting of a new mouth for the estuary, changed tidal amplitudes resulted in tidal exposure of large areas of these mudflats. Mangrove proliferation has resulted, as outlined above, but large subtidal mudflats are still to be found in the estuary, and they constitute the greater area of the estuarine embayment, covering approximately 450 ha. Mudflats occur in the south-western corner of the harbour (125 ha) and have been identified as important areas for estuarine associated macrocrustacea (Weerts *et al.* in prep.).

Given the widespread siltation in KwaZulu-Natal estuaries it seems unlikely that mudflats could be considered a particularly threatened habitat type. However, as systems become shallower marginal vegetation in the form of mangroves or reeds encroach and open water habitat is lost. This is a natural process but the rate at which it is occurring is problematic (Morant & Quinn 1999). The mudflats at Mhlathuze Estuary and Richards Bay Harbour are important by virtue of their extent, as there are few large, open mudflats along the KwaZulu-Natal coast (EAS 1996).

Canal habitat

Canal estates and marinas have not been developed in subtropical South Africa to the same extent as some regions in the United States and Australia. Such developments in these countries have involved the loss of extensive areas of mangroves (Blaber 1997). In contrast, the area to the west of the main harbour embayment at Richards Bay has remained relatively undisturbed since canals were created there during harbour development in the mid-1970's. These canals are now lined with *Avicennia marina* and comprise the most productive area in the harbour in terms of crustacean abundances and diversities (Forbes *et al.* 1996, Weerts *et al.* in prep.). Their significance as habitat for estuarine fishes has not been established.

Mangrove-lined, man-made canals in subtropical South African estuaries are limited to Richards Bay Harbour, and possibly to some areas in the lower reaches of St Lucia Estuary. Both systems have supported small fisheries targeting penaeid prawns (Forbes 1999). At St Lucia, where the fishery has been closed, only natural channels were trawled while in Richards Bay, where the fishery is still operational, trawling is restricted to the artificial canals described above. Comparisons of annual catch per unit effort nevertheless indicated that the Richards Bay canals were frequently more productive than St Lucia (Forbes 1999).

2.5 Catchment characteristics

2.5.1 Mhlathuze Estuary

Drainage into the estuary is derived mainly from the north via Mhlathuze River and the west via Mtantatweni River flowing from Lake Cubhu (Figure 2.1). A small canal that drains nearby sugarcane fields flows into the estuary between these two rivers. The significance of this canal as a freshwater source is minor compared to Mhlathuze River, and to a much lesser degree Mtantatweni River. Catchment characteristics of these two rivers will be dealt with below.

2.5.1.1 Mhlathuze River catchment characteristics

Topography, Geology, Geomorphology and Hydrology

The Mhlathuze is one of the larger rivers to cut across the Zululand Coastal Plain. It arises in the Babanango mountain foothills and has a catchment of approximately 4,209 km^2. Topography in the upper catchment is rugged and the land descends rapidly from 1,519 m to 200 m above sea level over a distance of only 60 km. For the next 65 km a more gradual rate of decrease in altitude takes the river to the ocean (Steyl *et al.* 2000). In the upper reaches the catchment geology is dominated by Dwyka tillite. This changes to consist mainly of Natal Structural and Metamorphic province rocks such as granite and basaltic lava. The geology of the middle catchment comprises mostly of Sedimentary rocks of the Ecca group, whilst that of the lower catchment is dominated by Quaternary sands (Dollar 2000). Sand based soils predominate in the catchment. Three main groupings are sandy clay in the upper catchment, sandy loam in the middle section and sand in the coastal area (Steyl *et al.* 2000).

Mean Annual Runoff

Mean annual rainfall varies from 870 mm in the upper and middle catchment to approximately 1,100 mm in the lower catchment (Dollar 2000). The catchment has a mean annual runoff (MAR) of 560 x 10^6 m^3. Small dams within the catchment have an estimated storage capacity of 5.2 x 10^6 m^3 (Germishuyse *et al.* 1998). The major dam is the Goedertrouw located about 85 km inland of Richards Bay. It was completed in 1980 to supply water for irrigation in the lower catchment and assist in the bulk water supply for urban and industrial development near the coast. Original capacity was 321 x 10^6 m^3 but this has been reduced by siltation to about 300 x 10^6 m^3, or 53.6% MAR (Steyl *et al.* 2000). In 1984 an abstraction weir *(Mhlathuze Weir) was constructed across the river about 11 km from the estuary* mouth. Water, released from Goedertrouw, is pumped from the resulting impoundment to Lake Nsezi where the water treatment plant that serves domestic and industrial demands in Empangeni and Richards Bay is located.

Until recently the ecological needs of aquatic systems downstream of Goedertrouw have not been considered in the operation of the dam. The hydrological regimes of these systems are therefore highly modified and atypical of natural seasonal fluctuations. Highest managed discharges from the dam are made in the drier winter months when agricultural needs are highest, and municipal and industrial water demands cannot be met by the reserves in Lake Nsezi and its catchment. During these winter months, river discharges downstream of Goedertrouw are often managed at a level determined by the pump capacity at Mhlathuze Weir and the need for water in Lake Nsezi. Discharge over the weir and into Mhlathuze Estuary is regarded as water loss. Monthly discharge volumes from late 1995 to 1999 are given in Figure 2.4. Although these years can be regarded as wet in terms of cyclical rainfall patterns, it can be seen that discharge over the weir has been avoided in winter months. Thus downstream flows and freshwater input into Mhlathuze Estuary is sometimes reduced to water that leaks from the sides and base of the weir.

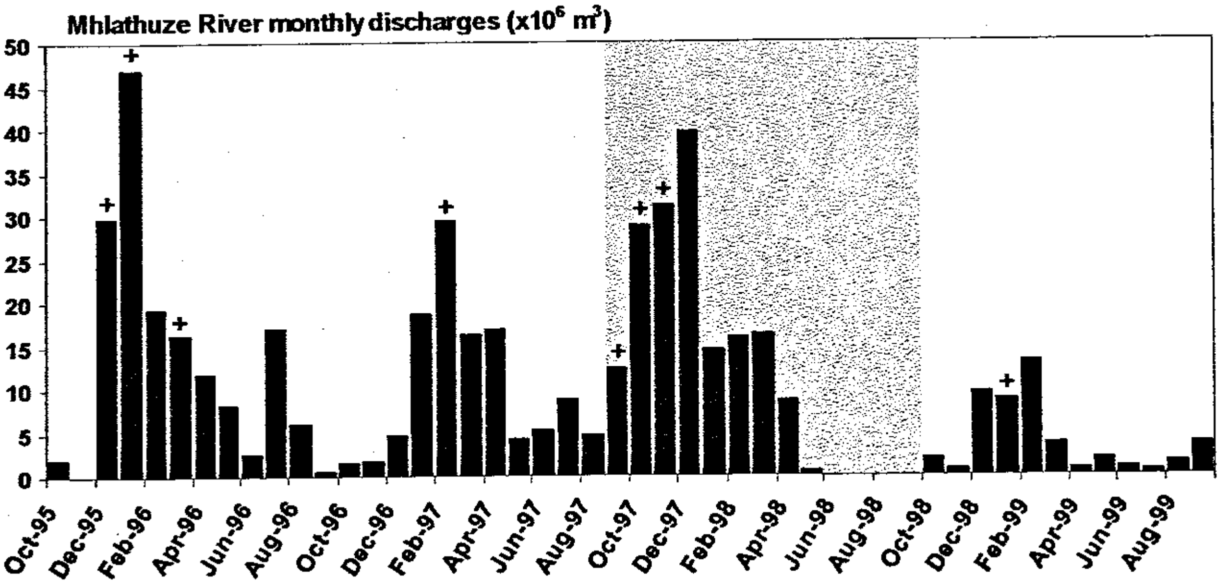


Figure 2.4: Mhlathuze River monthly discharges (x10^6 m^3) recorded from a weir 11 km from the estuary mouth (data kindly supplied by the Department of Water Affairs and Forestry). + = actual discharge greater than indicated, shading indicates study period.

In order to comply with the requirements of the new National Water Act (Act 36 of 1998) which makes provision for the ecological needs of rivers, lakes and estuaries, studies have now been conducted and 'reserves' set for major water bodies in the lower Mhlathuze catchment (Kemper 2000a). The implementation of recommendations made, and the adoption of a set of operating rules governing the quantity and timing of releases from Goedertrouw will serve to partly re-establish the natural hydrological regime of this area of the catchment.

Land use, Catchment Vegetation, Riparian Vegetation

Natural vegetation in the upper catchment is scrub and transitional forest. Tropical bush and savannah are found in the middle catchment and natural vegetation of the lower catchment is dominated by coastal tropical forest (Dollar 2000). However, major land use in the catchment includes commercial forestry (wattle, pine and gum) and some citrus farming. Extensive areas in the lower catchment have been put to sugarcane production. Much of the rural land that is not commercially farmed is characterised by subsistence farming.

Exotic species dominate the riparian vegetation in the higher river reaches. Black wattle (*Acacia mearnsii*), silver wattle (*Acacia dealbata*), poplar (*Poplus* spp.) and blue gums (*Eucalyptus* spp.) predominate (Kemper 2000b). Extensive bedrock and shallow soils in the gorge areas upstream of Goedertrouw Dam limit riparian habitat and dry indigenous riparian species are prevalent. These include *Diospyros lycioides*, *Rhus gerrardii*, *Ziziphus mucronata*, *Acacia karroo*, *Maytenus senegalensis*, *Spirostachys africana* and *Peltophorum africanum*.

Below the dam the river has been extensively modified due to channel narrowing and deepening and vegetation encroachment. Much of the riparian zone is completely choked with reeds (*Phragmites* sp. and Spanish reed). Important contributing factors here are the highly regulated flows from the dam to meet winter irrigation needs and the high nutrient loads of the irrigation return flows (Kemper 2000b).

From the confluence with the Mfule the river forms a mobile sinuous single thread channel with braided bars. A more diverse riparian habitat develops, characterised by individuals of *Ficus sycomorus*, *Syzygium guineense*, *Trichelia emetica*, *Celtis africana*, *Trema orientalis* and *Ficus sur*. A number of exotic species such as syringa (*Melia azedarach*), lanata (*Lantana camara*) and triffid weed (*Chromalaena odorata*) have also successfully recruited into these reaches, but reed beds are still dominant. The lower coastal plain reaches still have remnants of natural riparian vegetation in the form of *Rhus chirendensis*, *Vangueria infausta*, *Bridelia micrantha*, *Erythrina lysistemon*, *Albizia* sp., *Ficus capreifolia*, *Phyllanthus reticulatus*, *Nuxia oppositifolia*, *Dichrostachys cinerea*, *Pluchea dioscorides*, *Acacia robusta* and *Hibiscus tiliaceus* (Kemper 2000b).

The section of river downstream of the Mhlathuze Weir is the most modified of the whole system. Originally this area was part of a large papyrus swamp which was drained and cleared for sugarcane farming. The river channel now essentially comprises of an artificial canal which was cut during harbour development (Steyl *et al.* 2000). Low salinity water occasional penetrates up this canal to a distance of about 3 km below the weir (Weerts & Cyrus 1998) and mangroves (*A. marina*) *begin appearing on the channel banks about 2 km downstream of this.*

A large tidal prism results in strong tidal currents at the system's mouth, and the estuary remains permanently open even during very low river flows. As is common in other permanently open South African estuaries with large tidal prisms (e.g. Kosi, Keurbooms, Kariega) tidal currents at the mouth have deposited marine sediments to form a flood-tide delta, giving rise to intertidal sandflats in the lower reaches (Cooper *et al.* 1999).

2.5.1.2 Mtantatweni River catchment characteristics

Topography, Geology, Geomorphology and Hydrology

The topography of this catchment is relatively flat and the geology comprises Holocene alluvium sediments over Pleistocene sands (Orme 1974 cited in Begg

1978). The Mtantatweni River drains Lake Cubhu 3 km upstream from Mhlathuze Estuary. The Mzingwenya River, flowing north-west parallel to the coast, is the main source of water for the lake and has a catchment area of approximately 80 km^2, with an annual precipitation of 1,300 mm (Hemens 1980). Some smaller streams drain off the back of the dune ridge south of the lake.

Mean Annual Runoff

Hemens *et al.* (1970) estimated MAR of the Mtantatweni River to be 3.7×10^6 m^3. Limited data in the form of four flow recordings taken during 1997 from Mzingwenya Stream (the main stream flowing into Lake Cubhu) indicated a much higher MAR of 7.1×10^6 m^3 (Weerts *et al.* 1998).

Land use, Catchment Vegetation, Riparian Vegetation

Land use in the catchment comprises forestry, sugarcane production, subsistence farming, urban development and some rural settlement. Biological and water quality data indicated that sites on Mzingwenya River, just upstream of Lake Cubhu, were impacted upon by activities in the nearby town of eSikhawini (Weerts *et al.* 1998). Natural vegetation that still exists probably comprises mostly of coastal tropical forest and grasslands. Riparian vegetation in the form of *Barringtonia racemosa* swamp forest is still found in isolated stretches of some streams. *Cyperus papyrus* is the dominant fringing hydrophyte in Lake Cubhu although some swamp forest (*Barringtonia*) is found in the headwaters.

2.5.2 Richards Bay Harbour

Drainage into the harbour derives from a catchment area very much smaller than that of Mhlathuze estuary. Freshwater flows into the harbour via two main canals, the Mzingazi Canal in the north-east and the Bhizolo in the west. Whilst general information pertaining to the harbour catchment characteristics was available, detailed data pertaining to these two sub-catchments could not be sourced. For this reason they are dealt with below in a general discussion on harbour catchment characteristics.

2.5.2.1 Richards Bay Harbour catchment characteristics

Topography, Geology, Geomorphology and Hydrology

The catchment contributing to freshwater input to Richards Bay Harbour is 183 km^2. Topography is flat and the underlying geology comprises compact sedimentary strata. Soils are moderately deep with sandy texture (Walmsley *et al.* 1999).

Mean Annual Runoff

Annual rainfall at Richards Bay averages 1,228 mm (Table 2.1). Mean annual runoff into the harbour is 30 – 35 x 10^6 m^3 (Walmsley *et al.* 1999). Most freshwater enters via the Mzingazi River which drains Lake Mzingazi to the north. At the head of the Mzingazi Canal this water is augmented by flow from the Ngodweni Canal, which drains Thulazihleka Pan. Drainage from swamps to the west of the harbour flows into the Bhizolo Canal (Figure 2.1). This freshwater is augmented by a small stream which flows into the Manzinyama Canal.

Land use, Catchment Vegetation, Riparian Vegetation

Urban development covers approximately 14 km^2 of the catchment area. Other main land uses are forestry (pine and *Eucalyptus*) and sugarcane farming (Walmsley *et al.* 1999). Major industries in close vicinity of the harbour include a bulk coal storage and handling facility, a fertiliser plant, woodchip exporting plants, two large aluminium smelters and a paper mill. Natural vegetation comprises mainly of coastal dune forest, coastal grassland, swamp forest and coastal bushveld (Weisser 1978). Significant areas of the marsh, which drain into the Bhizolo Canal to the west of the harbour, are dominated by papyrus (*C. papyrus*). This area was formerly on the wet Mhlathuze floodplain and represents a remnant of more extensive papyrus swamps which were lost to sugarcane fields and industrial development.

2.6 Study sites

2.6.1 Mhlathuze Estuary

Morphometry of the estuary mouth is fairly dynamic, probably in response to variable sediment loads, river flow and longshore drift. It is typically of the order of 300 m wide. Depths in the channel vary between 1.5 and 2.5 m and maximum current speeds reach about 1 $m.s^{-1}$ (A. Viljoen pers. comm.). The estuary's greater surface area and volume comprises an embayment, which is almost entirely mangrove-lined. This appears to be related to the nature of the sediments with muddy shores being colonised by the pioneer mangrove species *A. marina*. Intertidal sandy beaches, which remain free of mangrove recruitment, occur in the lower reaches of the main basin and at the mouth. Eelgrass *Zostera capensis* occurs in shallow water near the southern bank of the main embayment. Open water mudflats and intertidal mangrove banks are the only habitats to be found in the middle and upper reaches of the estuarine basin. Mangroves give way to *Phragmites* reed swamps as the dominant marginal vegetation in the canalised sections of upper estuary about 6 km from the mouth, presumably due to decreasing tidal influence and salinity. Reed swamps were not sampled for the purposes of this study.

2.6.2 Richards Bay Harbour

A wide and deep harbour mouth is maintained to facilitate shipping traffic (900 m between breakwaters, maximum channel depth 21.7 m). Offshore sediment traps are continually dredged to prevent longshore sediment movement creating shallow shoals across the harbour entrance. Dredging is also conducted in the harbour to maintain a depth of about 20 m in the shipping channels and two main basins. Unvegetated sandy beaches occur in the mouth region, as well as in the harbour proper where a sand spit separates the two main basins. Only the eastern edge of the southern basin is dredged, and a large shallow flat has developed south of the sand spit across to the opposite shore and west to the Bhizolo Canal outflow. Sediments on this flat become muddier further away from the sand spit and closer towards the Bhizolo Canal. *Avicennia marina* mangrove stands have established in the canal area and all along the south-western border of the harbour.

Bruguiera gymnorrhiza mangroves are restricted to small stands on either side of the main entrance channel. Sediments at the fringes of these mangroves are sandy, probably as a result of their proximity to the main harbour channel and tidal currents. At the outflow of Mzingazi Canal, nearer the harbour mouth, the substratum comprises well-sorted marine sands which have remained free of colonisation by mangroves.

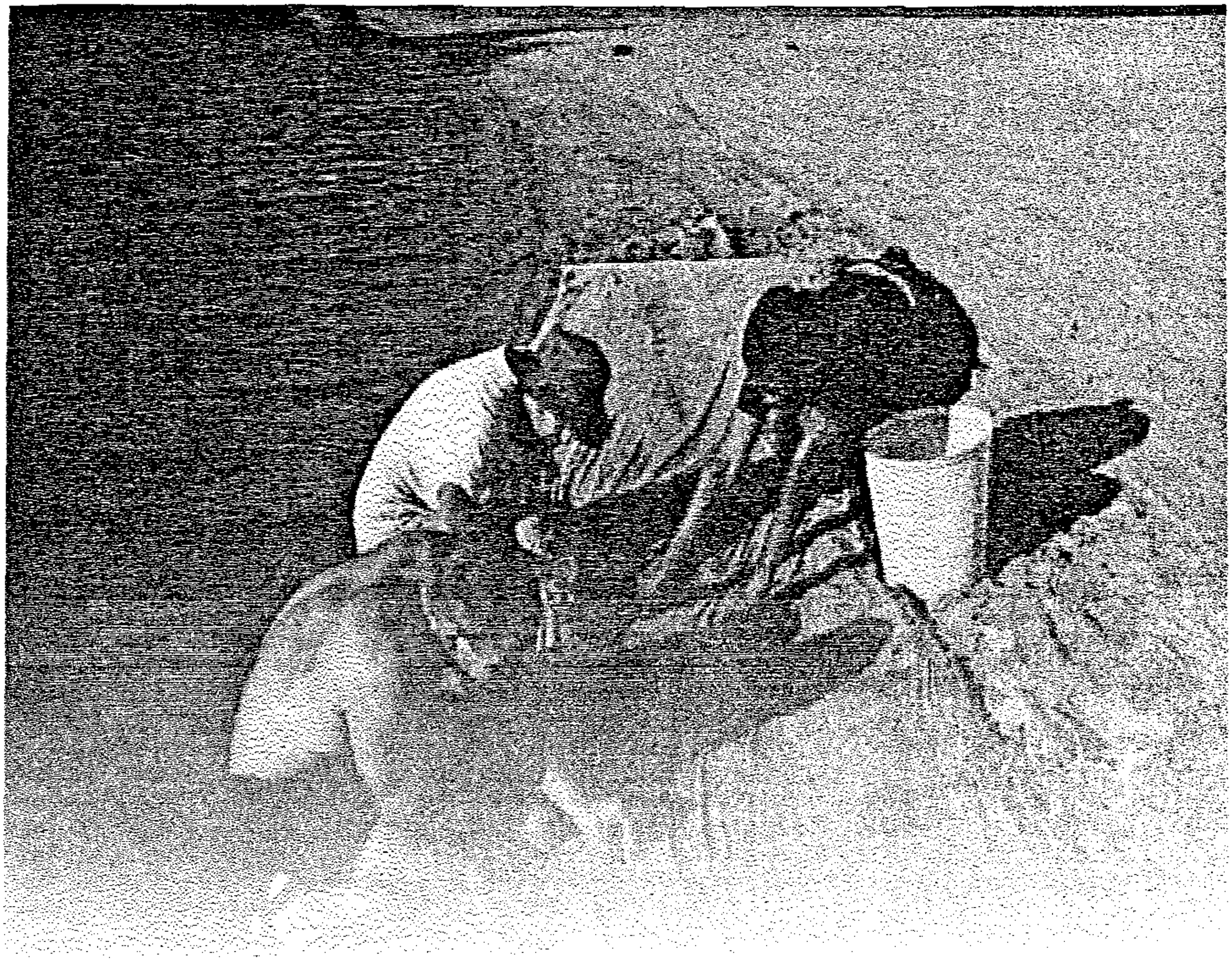
2.7 Sampling sites

Sites were selected in the two systems to reflect the diversity of structural habitat. Elements of structure considered were sediment type and vegetation (Table 2.2, Figure 2.3). Substrata were regarded as sand or mud, and vegetation comprised of eelgrass *Z. capensis* and the two mangroves species *A. marina* and *B. gymnorrhiza*. A subjective distinction between 'sand' and 'mud' was made on the basis of sand being regarded as firm underfoot. Walking over it did not visibly increase turbidity. Mud was soft underfoot and one sank into it when walking. This caused a marked increase in turbidity which persisted in the absence of any current to move the suspended fine sediment away. This subjective assessment was confirmed by taking sediment samples and conducting grain size analyses. Unvegetated sandbanks and the *Zostera* (Mhlathuze Estuary) and *Bruguiera* (Richards Bay Harbour) sites were dominated by fine to medium sands while open water mudflats and *Avicennia* mangrove fringes were predominately muddy (standard Wentworth scale sediment size classes).

Table 2.2: Descriptions of sites and habitats from Mhlathuze Estuary and Richards Bay Harbour. Averages and ranges of physical water variables indicated. Site numbers correspond with locations on Figure 2.3B.

Site		Habitat description	Average (minimum-maximum) Temp. (°C)	Salinity (‰)	Turbidity (NTU)
1	estuary	intertidal sandbank (mouth) sand	24.55 (20.14-30.37)	30.80 (17.7-36.7)	16.83 (11-29)
2		intertidal sandbank (basin) sand	24.22 (20.68-30.03)	34.20 (25.9-37.3)	12.67 (2-23)
3		subtidal mudflat mud	23.51 (17.44-28.23)	27.33 (12.4-36.4)	24.25 (7-72)
4		mangrove (*Avicennia*) bank mud	24.53 (21.17-29.83)	28.76 (2.1-36.8)	18.08 (6-37)
5		eelgrass (*Zostera*) meadow sandy mud	23.07 (18.18-30.26)	30.07 (24.1-36.4)	12.75 (4-27)
6	harbour	intertidal sandbank (mouth) sand	23.84 (19.99-27.13)	31.60 (22.9-36.1)	3.92 (3-5)
7		intertidal sandbank (basin) sand	23.01 (20.30-28.69)	35.21 (33.4-37.5)	8.33 (3-21)
8		subtidal mudflat mud	22.37 (18.51-26.87)	35.14 (32.8-36.8)	11.0 (7-23)
9		mangrove (*Avicennia*) bank mud	23.35 (20.56-27.88)	35.40 (33.1-37.3)	7.25 (3-17)
10		mangrove (*Avicennia*) bank (canal) mud	22.87 (18.52-29.22)	31.17 (13.6-35.4)	13.25 (7-29)
11		mangrove (*Bruguiera*) bank sand	22.98 (20.26-26.36)	35.22 (32.7-37.3)	6.42 (4-14)

In both the estuary and the harbour sandy beaches in the embayment areas and those in the lower regions were regarded as separate habitat types. Although structurally similar, these areas differed in their proximity to the respective systems' mouths. Mudflats only occurred in the main basins of the systems sampled. No unvegetated littoral mud substrata were sampled as muddy banks in both systems support mangrove stands, and these were considered as distinct habitat. The *Avicennia* fringe in the Bhizolo Canal was sampled as canal habitat distinct from other white mangrove areas in the harbour. *Bruguiera* mangroves occurred in the harbour but not the estuary, and the opposite was true of *Zostera*, which was sampled in the estuary but did not occur in the harbour.



CHAPTER 3

Materials & Methods

Chapter 3

Materials and Methods

3.1 Study design

The general study approach used was to sample ecological and physico-chemical characteristics of selected habitats in Mhlathuze Estuary and Richards Bay Harbour. Habitats were selected on the basis of observable structural elements, such as presence or absence of different vegetation types, nature of the substratum and whether the habitat was littoral or not. Physico-chemical water characteristics were regarded as environmental parameters that may have influenced fish use of habitats. As both Mhlathuze Estuary and Richards Bay Harbour are well-mixed, marine dominated embayments, it was expected that within each system, spatial differences in water characteristics would be limited to few of the parameters measured. If this was confirmed with suitable statistical analyses, differences in fish assemblages associated with different habitats (if any) could be attributed to habitat structure. If not then some qualification of water quality would need to be incorporated into the definition of habitat.

Data were gathered in such a manner as to allow each system to be investigated separately. However, the consistency in the sampling regime and methods employed also allowed comparison of fish faunas and habitat utilisation in Mhlathuze Estuary and Richards Bay Harbour. By comparing structurally similar habitats in different systems (i.e. omitting from analyses habitat types that occurred in one system only, e.g. *Zostera* beds in the estuary, *Bruguiera* stands and canal mangroves in the harbour) patterns and trends in habitat utilisation could be investigated. Given the proximity of the two systems, factors such as latitude, local climate and available pools of potential recruits (newly spawned fishes) could be assumed to be constant, and therefore ignored as confounding variables. Differences noted in faunas associated with structurally similar habitats could therefore be ascribed to environmental parameters and the different nature of the systems in terms of factors such as hydrological regimes, mouth and basin morphologies or anthropogenic modification.

The protocol followed for each system was to investigate the physico-chemical water quality characteristics and to determine if there were significant differences in any of the parameters recorded at each site over the sampling period. Differences in the fish assemblages at different sites were then investigated and considered in relation to differences in habitat structure and/or environmental characteristics. Once patterns of habitat utilisation had been assessed in the estuary and harbour, physico-chemical and biotic data from sites with common habitat structure from both systems were investigated in a similar manner.

3.2 Dates of sampling

Mhlathuze Estuary and Richards Bay Harbour were sampled on a monthly basis from September 1997 to August 1998. September is the first month of the austral spring and generally coincides with an increase in sea surface temperatures after winter minima in June, July and August (Figure 2.2). Summer begins in December and ends with peak sea surface temperatures in February. March, April and May are autumn months, characterised by falling ocean water temperatures. Fieldwork at the two systems was performed on consecutive days, during daylight hours, on neap tides three or four days before the full moon spring. This tide was selected because it permitted access to all selected habitats in each system and sufficient daylight time to sample them. Moderate low water levels meant that boat launching and commencement of sampling could begin soon after the morning turn of the tide. It had the added benefit of largely standardising sampling across the range of habitats to an incoming tide, although the last sites were usually sampled just after full high during most trips.

3.3 Field techniques

3.3.1 Abiotic characteristics – Physico-chemical parameters

Physico-chemical characteristics of the water at each site were recorded and turbidity samples taken immediately prior to sampling of fishes. Depth was measured (10 m from the shore in the case of littoral sites) and the following water parameters were determined *in situ* at mid water using a Hydrolab Datasonde 3 Water Quality Multiprobe and Logging System: salinity (‰), temperature (°C), dissolved oxygen content (mg/l) and pH. Turbidity samples collected in the field were stored in a cool box, returned to the laboratory and analysed using a Hach Model 2100A Turbidimeter (Nephelometric Turbidity Units, NTU).

3.3.2 Biotic characteristics - Fish sampling

The juvenile and small fish faunas of the two systems were sampled using a small seine net (1 mm bar mesh, 2.5 m depth, 8 m weighted toe line and 10 m float line). While it was not possible to seine in intertidal mangrove forests the net could be effectively deployed in open water immediately adjacent to mangroves. In such habitats seine nets are likely to provide a more accurate indication of the true abundance and composition of the fish community than otter trawls or beam trawls (Morton 1990). Hauls at *Avicennia* sites generally ended in sweeping an area with pneumatophores. A small seine net was considered to be the sampling gear that could be most consistently and efficiently deployed at all the selected study sites. The standardised use of a single sampling device allowed meaningful comparisons of fish densities recorded from different habitats. Similar equipment has been found effective in sampling similar habitats in other studies (e.g. Beckley 1983, Laegdsgaard and Johnson 1995).

The tendency for seine nets to undersample species closely associated with the substratum (Connolly 1994) was addressed by heavily weighting the toe line and fitting spreader poles to the ends of the net, in a similar manner to that employed by Paterson and Whitfield (2000) for the same reasons. The dimensions of the net were decided upon after trials during a pilot study. Seines tend to underestimate abundance of larger fishes, especially if not long enough or deployed incorrectly (Morton 1990). At 8 m the net was found to sample a sufficiently wide area to catch small fishes more effectively than a beam trawl or tow/push nets with narrower widths. However, it was still short enough for two people to easily pull through the water (given the small mesh size) and over the *Zostera* without rolling. Gray *et al.* (1996) used a 10 m seine net to sample fishes from vegetated and unvegetated habitats and regarded the net as effective for an assessment of habitat utilisation by fishes up to 100 mm total length. A limit of 50 mm SL was decided upon for this study which is conservative. A longer float line allowed a bag effect to develop as the net was moved through the water. This had the effect of reducing the net's advance pressure wave and herding fishes toward the centre of the net thereby decreasing the numbers that escaped around the sides. Sites sampled were never deeper than 1.5 m and the depth of the net (2.5 m) sufficed for use in the habitats investigated. Extra net depth was useful in that it allowed the toe line to be angled forward and then pushed up whilst moving, thus scooping fishes in the water column from substratum to surface at sites where the net could not be hauled ashore.

Every month three replicate hauls were taken a distance apart from one another at each site, with the exception of the *Bruguiera* stand in the harbour and *Avicennia* mangroves in the Bhizolo Canal. Steep drop-offs restricted suitable areas for landing nets and sampling was reduced to single hauls at these sites. The distance of each haul was recorded so that the area sampled could be determined and catches expressed in terms of numbers of fish per 100 m^2 of surface sampled. Where possible hauls of 10 m were made, but at some of the mangrove sites haul distances were reduced so that only the water in close vicinity to the mangroves was sampled rather that the mudflats adjacent to them. At littoral sites over sandflats and in the mangroves the net was pulled ashore. It was assumed that net was deployed at full breadth of the 8 m toe line, but in reality some bowing occurred during each haul. Fish densities recorded were therefore expected to underestimate actual abundances (because of overestimating the area seined). This was assumed to be constant across all habitats.

Hauls over the seagrass and mudflats were generally ended by scooping the toe line up, and out of the water while still moving forward and taking care not to let the top line, which was floated, immerse below the water surface. After all hauls the net was searched for fishes. Specimens that could be positively identified and accurately measured to the nearest millimetre (SL; tip of snout to caudal peduncle) were recorded and returned to the water. All other fishes were preserved in a 4% buffered formaldehyde solution and returned to the laboratory.

The 1 mm mesh size allowed fish from a minimum of 3 mm SL to be sampled. The net was therefore effective in sampling 'settlement' or 'transforming' stage larvae of most of the species expected (Harris & Cyrus 1996). This developmental stage, alternatively called postlarval, represents the end of the larval phase and that at which fishes become efficient at actively maintaining their position in their preferred juvenile habitats (Neira *et al.* 1998). For the purposes of this study only fish up to 50 mm SL were considered with the exception of two pipefish species (Syngnathidae) which, due to their elongate body shapes, were considered up to 110 mm SL. These are not mobile fishes and it is unlikely that they avoided capture by swimming around the sides of the seine net at sizes larger than 50 mm. These size classes of fish, which included 0+ juveniles of all species as well as adults of some smaller bodied fishes (such as gobiids and ambassids) were the subjects of investigation. The presence of larger fishes was noted, but they were not included in any analyses as their recorded abundances were potentially biased due to net avoidance.

3.4 Laboratory techniques

Fishes were identified in the laboratory with the aid of several reference texts (Leis & Trnski 1989, Smith & Heemstra 1991, Neira *et al.* 1998, Leis & Carson-Ewart, 2000). Use was made of a Leica MZ6 binocular microscope when necessary. In some cases species were pooled at the level of genus due to morphological similarity at their early juvenile stages (e.g. *Ambassis* spp.). Mullet were pooled at the level of family due to difficulty in distinguishing small specimens, even to the level of genus.

3.5 Data analyses

3.5.1 Abiotic characteristics

Formal analyses of physico-chemical data were performed using univariate and multivariate methods. Water temperature, dissolved oxygen, pH, salinity and turbidity were all subjected to analysis of variance (ANOVA). In most cases variables failed to meet the statistical assumptions of parametric ANOVA techniques [i.e. normality (determined by Kolomogrov-Smirnov Tests) and homogeneity of variances (determined by Levene Median Tests)] even after transformations were made. For this reason nonparametric methods of ANOVA were resorted to. Where three or more sets of samples were compared (e.g. comparison of physical variables amongst different habitat types or seasons) the Kruskal-Wallis procedure was used to conduct analysis of variance by ranks. Where two sets of samples were compared (e.g. comparison of physical variables between two systems) the Mann-Whitney Rank Sum Test was used. When the null hypothesis of no difference was rejected at a probability $P < 0.05$, differences of ranks were compared using pairwise multiple comparisons (Student-Newman-Keuls Method). Thus differences in water parameters could be investigated across habitat and season groups in the case of the individual systems (Chapters 4 and

5), and across system, habitat and season groups when data from Mhlathuze Estuary and Richards Bay Harbour were pooled (Chapter 6).

Multivariate analyses were conducted using the computer software package PRIMER v5 and included techniques of clustering (hierarchical group average linkage) and ordination [non-metric Multi-Dimensional Scaling (MDS)]. Both these techniques were based on similarity coefficients computed between every pair of samples. In the case of sample-by-environmental variable matrices these coefficients measure how similar samples are across the range of environmental variables measured. Similarity coefficients were determined for each pair of samples. Data were log(x+1) transformed prior to analysis and Normalised Euclidean Distance was used as the similarity measure as suggested by Clark and Warwick (1994) for dealing with environmental data.

In clustering, each sample (e.g. site) is assigned to one of several groups on the basis of the relative similarity of its attributes (e.g. physico-chemical characteristics or abundances of similar species occurring) with those from other samples (Randerson 1993). The aim of the analysis is to find natural groupings of samples so that those within a group are more similar to each other than samples in different groups. The endpoint is a dendrogram which indicates the successive fusing of samples into clusters starting at the highest mutual similarities (Clarke & Warwick 1994).

In ordination, each sample is placed on one or more constructed axes so that its geometrical position relative to other samples reflects its similarity to them (Randerson 1993). In this way MDS constructs a configuration in a specified number of dimensions (usually two dimensions to facilitate visual interpretation) so that samples more similar to one another are plotted closer together on the configuration than samples that are dissimilar (Clarke & Warwick 1994). Thus, while clustering is a technique that imposes discontinuities on the data by assigning samples to discrete groups, ordination detects continuous variation in the measured parameters, or the distribution and abundance of species between samples (Randerson 1993). The adequacy of an MDS in representing similarities amongst samples in a low dimension ordination plot can be measured by the extent to which the ordination preserves the rank order of dissimilarities. A stress value can be determined and gauged against benchmarks (Clarke & Warwick 1994). For two-dimensional ordinations:

- stress <0.05 gives an excellent representation with no prospect of misinterpertation
- stress <0.1 gives a good ordination that is unlikely to lead to misinterpretation
- stress <0.2 gives a potentially useful ordination which may be interpreted with some caution
- stress >0.3 gives an ordination that should be treated with scepticism

For the purposes of this study, when two-dimensional ordination stress was >0.2 the three-dimensional ordination was examined. If there were no marked inconsistencies with the planar representation, and the two-dimensional stress was in the lower half of the range 0.2 to 0.3 the ordination was still used with superimposed groups from a cluster analysis. Clarke and Warwick (1994) recommend this combination of clustering and ordination analyses as a means of checking the adequacy and mutual consistency of both representations.

Tests for significance of differences in suites of measured physico-chemical variables were performed using analysis of similarities (ANOSIM). This procedure uses ranked similarity distance to compute a statistical test of the validity of differences among groups of samples specified *a priori* (Clarke & Warwick 1994). The protocol used for the present study of sampling a fixed set of sites at several times was a 2-way crossed design and the appropriate ANOSIM was test selected. The ANOSIM test statistic (R) is defined (Clarke & Warwick 1994) so that:

- $-1 \leq R \leq 1$
- R = 1 only if all replicates within sites are more similar to one another than any replicates from different sites
- R approaches zero if the null hypothesis is true, so that similarities between and within sites will be the same on average

Permutation tests, using a computer to calculate R for different permutations of relabelled sites, can be conducted and the probability of the observed R being derived from a random rearrangement can be determined (Clarke & Warwick 1994). A probability level (P) of <0.05 was regarded as significant.

3.5.2 Biotic characteristics

Fishes sampled were categorised according to their dependence on, or association with estuaries in southern Africa. Estuarine fish assemblages are generally composed of a mixture of species restricted to estuaries, and euryhaline freshwater and marine species. Stenohaline marine fishes often occur in the lower reaches of permanently open systems. These are often species which are not dependent on estuaries to any degree and have been termed "marine stragglers" (Potter *et al.* 1990). They are not usually regarded as part of the estuarine ichthyofauna (Wallace 1975a) but in embayments such as Mhlathuze Estuary and Richards Bay Harbour they could form significant components of the fish fauna.

Estuarine association categories of Whitfield (1998; Table 1.1) are the most recent and comprehensive of a series of classifications proposed by various ichthyologists over the years for southern African estuarine fishes. The application of these categories to taxa sampled in this study was difficult due to the unavoidable grouping of some species into genus and family complexes. Mullet, for instance, although all marine spawners, could fall into several of Whitfield's (1998) categories depending on species. Juvenile *Liza macrolepis* are dependent on estuaries as nursery habitats (EAC IIa) whereas juvenile *Liza dumerilii* and

Valamugil buchanani occur in estuaries as well as at sea to varying degrees (EAC IIb and EAC IIc respectively). *Myxus capensis* is a facultative catadromous species, which typically uses estuaries as transit routes between marine and freshwater environments (EAC Vb; Whitfield 1998). All these mullet species, and others, occur in Mhlathuze Estuary (Weerts & Cyrus 1998) and Richards Bay Harbour (CRUZ unpublished data) but were grouped as a species complex in this study. A more generalised classification was therefore needed to categorise each taxon according to its association with the estuarine environment.

Harrison (1993) classified fishes occurring in KwaZulu-Natal estuaries into four broad categories: freshwater species, estuarine species, estuarine-dependent marine species and marine species. In this categorisation local experts were consulted and information from a wide range of studies was used [including Wallace (1975a), Wallace *et al.* (1984), Bruton *et al.* (1987), Bok (1988), Potter *et al.* (1990) and Cyrus (1991)]. Similar species designations were used for the present study and the same form of categorisation was adopted to classify species not recorded by Harrison (1993), with reference being made to the estuarine association category lists of Whitfield (1998) to assist where necessary. The classification of catadromous species using this protocol was problematic. However, of the three catadromous fishes that utilise South African estuaries, only freshwater eels (Anguillidae) have an obligate freshwater phase (EAC Va; Whitfield 1998). These eels do not use estuaries as nursery habitats, but merely as transit routes between freshwater systems and the sea. The single occurrence of an Anguillidae leptocephalus larva from sampling in the present study was therefore omitted from any analyses. Freshwater mullet *Myxus capensis* and oxeye tarpon *Megalops cyprinoides*, although regarded as facultative catadromous species (EAC Vb; Whitfield 1998) do use estuaries as nursery habitats (Whitfield 1994d). This is especially true in Mhlathuze Estuary and Richards Bay Harbour, where access to significant freshwater reaches is denied by weirs and barriers to migration (Weerts & Cyrus 2002). They were therefore designated as estuarine-dependent marine species. Categories assigned to the taxa sampled are listed in Appendix 1. The percentage contribution made by each category to the total recorded density of fishes in each habitat type was calculated and results compared across habitats, seasons and systems. Similarities in habitat preferences and seasonal occurrences amongst fishes within these broad estuarine association groups were also investigated statistically using multivariate techniques discussed later in this section.

Biotic data generated from field sampling produced large species-by-samples matrices typical of ecological studies. A variety of formal statistical analyses were used to investigate patterns in community structure. These comprised both univariate and multivariate techniques. Univariate indices, such as total number of individuals or total number of species, are often used as the simplest measures of community structure in biological studies. Diversity indices, which measure the way in which the total number of individuals is divided up among the different species, tend to be more informative (Clarke & Warwick 1994). Many diversity indices have been developed and all emphasise species richness and equitability components of diversity to varying degrees. It was therefore felt prudent to consider several univariate measures for this study:

- S = total number of species recorded per sample
- N = total density of fishes recorded per sample
- d = Margalef's species richness
  $d = (S-1)/\log N$
- H' = Shannon-Wiener diversity
  $H' = -\Sigma_i p_i(\log_e p_i)$ where p_i is the proportion of the total sample density arising from the i^{th} species
- J' = Pielou's evenness index
  $J' = H'_{observed} / H'_{max}$ where H'_{max} is the maximum possible diversity which would be achieved if all species were equally abundant

Diversity indices, as described above, all effectively collapse full sets of species counts in samples into single coefficients, which can be validly treated using univariate statistical methods (Clarke & Warwick 1994). The significance of differences in univariate indices amongst habitats and seasons was tested by ANOVA. Where necessary data were transformed to meet the assumptions of normality and homogeneity of variances. Appropriate transformations were determined using a method suggested by Clarke and Warwick (1994). This technique relies upon the gradient of the fitted straight-line plot of log standard deviation against log mean of the data set. A slope of zero implies no transformation, 0.5 implies the square root, 0.75 the 4th root and 1 the log transform (Clarke & Warwick 1994). Multiple range tests (Tukey's honest significant differences) were used to make comparisons between habitats and investigate homogeneous groups for the means. Bar charts, used as graphical representations of univariate indices, were plotted as means with 95% confidence limits as an indication of significant differences amongst sites. In cases where data were transformed for the purposes of ANOVA, means and the 95% confidence limit endpoints were back-transformed for graphical presentations.

Community data are inherently multivariate and it was desirable to investigate fish assemblages with consideration of their constituent species. Appropriate techniques to deal with multivariate data often rely upon reducing the complexity of large species-by-samples matrices by graphical representation of the biological relationships between samples. This is followed by statistical testing to identify and characterise changes, and to relate these to changing environmental, or experimental conditions (Clarke & Warwick 1994).

Clustering and ordination methods as described above for abiotic analyses were employed. Similarity matrices upon which these techniques are based were constructed using the Bray-Curtis coefficient rather than Normalised Euclidean Distance. This coefficient is commonly used in ecological research (Clarke & Warwick 1994) and is a measure of how similar abundance levels are for each species, averaged over all species in different samples. Bray-Curtis similarity is defined so that 100% represents total similarity and 0% complete dissimilarity.

All data were square root transformed ($x^{0.5}$) prior to the computation of similarity matrices. The role of transformation in non-parametric tests is to weight the contributions of common and rare species, rather than to validate statistical assumptions as is the case for parametric techniques. A root transform can be regarded as moderate and was chosen as it allowed intermediate abundance species to play a role in determining the measure of similarity of samples (Clarke & Warwick 1994). More severe transformations, such as $x^{0.25}$ or log(x+1), would have placed emphasis on uncommon species and chance occurrences and no transformation would effectively have reduced any significant influence in determining similarity of assemblages to common species only.

Tests for differences in structure and composition of assemblages were performed using analysis of similarities (2-way crossed ANOSIM) as described for the abiotic data, but using ranked Bray-Curtis similarities to compute a statistical test rather than ranked similarities based on Normalised Euclidean Distance. Species determining the observed sample patterns were identified using the similarity percentage routine (SIMPER) in the PRIMER software package. This analysis is an exploratory tool rather than a statistical testing framework, and it indicates which species are principally responsible for any observed clustering patterns or differences between sets of samples defined *a priori*, and confirmed to differ in community structure by ANOSIM (Clarke & Warwick 1994). It does this by identifying the relative contribution of different taxa to the similarity within a group of samples from a habitat type for example, as well as determining which species contribute most to the dissimilarity between groups of samples taken from two different habitats.

These statistical techniques were also suitable for investigating similarities amongst groups of fishes in their habitat preferences and seasonal occurrences. This is the inverse of the analyses conducted above of similarities amongst habitats and seasons in their associated fish faunas. This supplemented analysis of contributions from different estuarine association categories described earlier, and allowed comment on whether fishes with different estuarine associations, or different biogeographical affinities, were similar in their distribution amongst samples (i.e. if their numbers fluctuated in parallel across habitats or seasons). Biogeographical affinities were investigated because fluctuations in the distribution of tropical and temperate fishes with season may have influenced community assemblages in the systems studied. Using the methods described above similarities were defined between every pair of species. In species analysis similarities between rare species have little meaning and their inclusion confuses and disrupts patterns in clustering and ordination analysis (Clarke & Warwick 1994). The original sample matrices were therefore reduced by removing all species which accounted for less than 1% of the abundance of fishes in each sample. Bray-Curtis similarities were then calculated on row standardised and untransformed data as suggested by Clarke and Warwick (1994). In the resulting ordinations and significance tests (ANOSIM), species were labelled according to their association with estuaries (as described above) and also according to their geographical ranges in southern African waters. Species information pertaining to geographical ranges was taken from Whitfield (1998) and Smith and Heemstra (1991), and allowed designation of fishes into subtropical, warm-temperate and cool-temperate categories. Many species encountered were not restricted to discrete geographical ranges in southern African waters and often ranges spanned more than one biogeographical region. In most cases this involved subtropical and warm-temperate regions but in some instances fishes were noted to occur along the full reach of the South African coastline. Geographical range classes were hybridised to accommodate this. Thus a particular species may have been classified as being subtropical/warm-temperate. Estuarine asscciation classes and geographical ranges used are listed in Appendix 1.

CHAPTER 4

Mhlathuze Estuary

Chapter 4

Mhlathuze Estuary

4.1 Results

4.1.1 Abiotic characteristics

Over the study period the average depth of the sampled sites in Mhlathuze Estuary was 0.63 m (SD = 0.18 m). Other physico-chemical variables recorded during monthly sampling trips are listed in Appendix 2 and discussed below.

Mean water temperature was 23.98°C (SD = 3.12°C) during the twelve months sampled with a range of 17.44 to 30.37°C. Temperatures did not differ amongst sites but there were significant seasonal trends (Table 4.1). January, February and March were months of highest mean temperatures whilst the lowest was recorded in June. (Figure 4.1). Dissolved oxygen content recorded at different sites ranged from 5.44 to 10.64 mg/l and averaged 7.05 mg/l (SD = 1.02 mg/l). Habitats were not distinguishable in terms of this variable but in autumn dissolved oxygen content was significantly lower than in winter, spring or summer (Table 4.1, Figure 4.1). Over the study period mean pH from all sites was 8.32 (SD = 0.28) with no significant spatial variation. A high mean monthly pH of 8.76 was recorded at the beginning of the sampling period in September 1997. Mean values dropped to 7.91 in November and then climbed steadily to 8.74 at the end of the sampling period in August 1998. Summer and autumn pH values were significantly lower that those during spring and winter (Table 4.1, Figure 4.1).

Mean salinity in the estuary was lower than that of seawater at 30.63 ‰ (SD = 7.50‰). The mudflat and mangrove sites were most markedly influenced by freshwater flow due to their locations. The mudflat site was located at the head of the embayment near the outflows of the Mtantatweni River and a distributary of the Mhlathuze River, while the mangrove site was at the main outflow channel of the Mhlathuze River (Figure 2.3). Average salinities at these sites (27.30 and 28.80‰

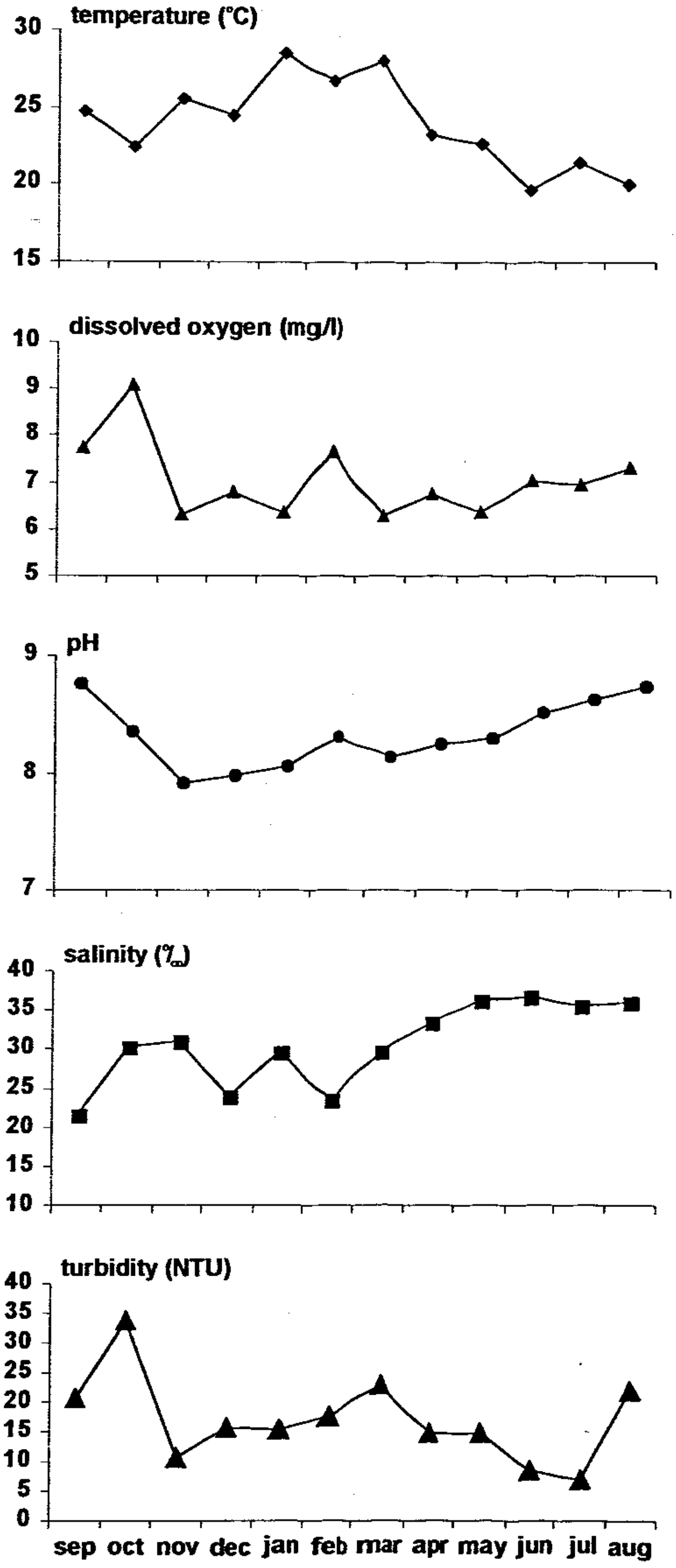


Figure 4.1: Physico-chemical variables recorded on a monthly basis from 1997 to 1998 in the Mhlathuze Estuary (averaged across habitats).

respectively) were lower than the rest of the estuary but these differences were not significant (Table 4.1). Significant seasonal trends were apparent with freshwater influences manifested in lower salinities from September 1997 to April 1998 but not from May 1998 to August 1998. This confers well with flow data from Mhlathuze River, which indicated that little to no water was flowing over the Mhlathuze Weir upstream of the estuary during the latter months (Figure 2.4).

Table 4.1: Nonparametric analysis of variance (Kruskal-Wallis tests) of physico-chemical variables at different habitats in Mhlathuze Estuary. N = total number of replicates, df = degrees of freedom, SNK = Student-Newman-Keuls tests, * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$, ^{ns} = $P > 0.05$, not significant.

Source	Variable	H statistic	SNK, comparison of median and percentiles
Habitat N = 12 df = 4	Temperature	2.038^{ns}	
	Dissolved O_2	4.190^{ns}	
	pH	1.954^{ns}	
	Salinity	6.127^{ns}	
	Turbidity	8.021^{ns}	
Season N = 15 df = 3	Temperature	32.140***	[summer > spring, autumn, winter], [autumn > winter], [spring > winter]
	Dissolved O_2	12.755**	[autumn < spring, summer, winter]
	pH	27.075***	[winter > autumn, summer, spring], [spring > summer, autumn]
	Salinity	30.016***	[winter > autumn, summer, spring], [autumn > summer, spring], [spring > summer]
	Turbidity	5.861^{ns}	

Turbidity ranged from 2 to 72 NTU with a mean of 16.9 NTU (SD = 10.60 NTU) during the study period. Turbidity over 50 NTU was recorded only once and 80% of all turbidities fell within the range of 10 to 50 NTU. Differences amongst habitats were not significant (Table 4.1), but turbidity was on average lowest at the estuarine sandbank and in the *Zostera* bed (12.7 and 12.8 NTU respectively) and was highest over the mudflat (average 24.3 NTU). Seasonal averages for all habitats sampled in the estuary did not differ significantly (Table 4.1) but turbidities were lowest in June and July 1998 which were both months of very low freshwater inflow (Figure 2.4). Turbidity was also influenced by factors other than river flow. During August 1998 turbidity was higher at all sites than the previous two months, despite low freshwater inflow. A strong south-westerly wind during sampling in August generated waves that brought soft sediments over the wider area of the estuary into suspension and increased turbidities.

From the above, and the summary of univariate statistical analyses presented in Table 4.1, it was apparent that there was significant seasonal influence on all but one of the measured abiotic variables. However, there was no difference in water quality parameters amongst habitats sampled in the estuary. Results of multivariate analysis are shown in the MDS ordination in Figure 4.2. A reasonably low stress of 0.12 indicated that the results could be safely interpreted. The ordination is presented twice to illustrate the influence of different factors. Samples have been labelled according to habitat (Figure 4.2A) and season (Figure 4.2B). There was little difference in the set of physico-chemical variables sampled at each habitat, but a distinct seasonal influence was apparent. Analysis of similarities confirmed this (Table 4.2) and the null hypothesis that environmental variables were not different across habitat groups (averaged across season groups) was accepted (R = -0.118, P >0.05), while the differences across season groups (averaged across habitats) were found to be significant (R = 0.243, P <0.001). Pairwise comparisons of similarities indicated that the conditions in the estuary in winter differed from those in spring, summer and autumn (Table 4.2).

Table 4.2: Analysis of similarities of physico-chemical variables recorded from different habitats in Mhlathuze Estuary. * = P <0.05, ** = P <0.01, *** = P <0.001, ns = P >0.05, not significant.

Test	R statistic	P
Differences amongst habitats		
Global Test	-0.118	ns
Pairwise Tests		
sandbank (mouth) vs. sandbank (basin)	-0.194	ns
sandbank (mouth) vs. mudflat	-0.176	ns
sandbank (mouth) vs. mangrove	-0.204	ns
sandbank (mouth) vs. *Zostera*	-0.102	ns
sandbank (basin) vs. mudflat	0.037	ns
sandbank (basin) vs. mangrove	-0.167	ns
sandbank (basin) vs. *Zostera*	-0.250	ns
mudflat vs. mangrove	-0.167	ns
mudflat vs. *Zostera*	0.000	ns
mangrove vs. *Zostera*	-0.167	ns
Differences amongst seasons		
Global Test	0.243	***
Pairwise Tests		
spring vs. summer	-0.080	ns
spring vs. autumn	-0.015	ns
spring vs. winter	0.222	**
summer vs. autumn	-0.074	ns
summer vs. winter	0.844	***
autumn vs. winter	0.541	***

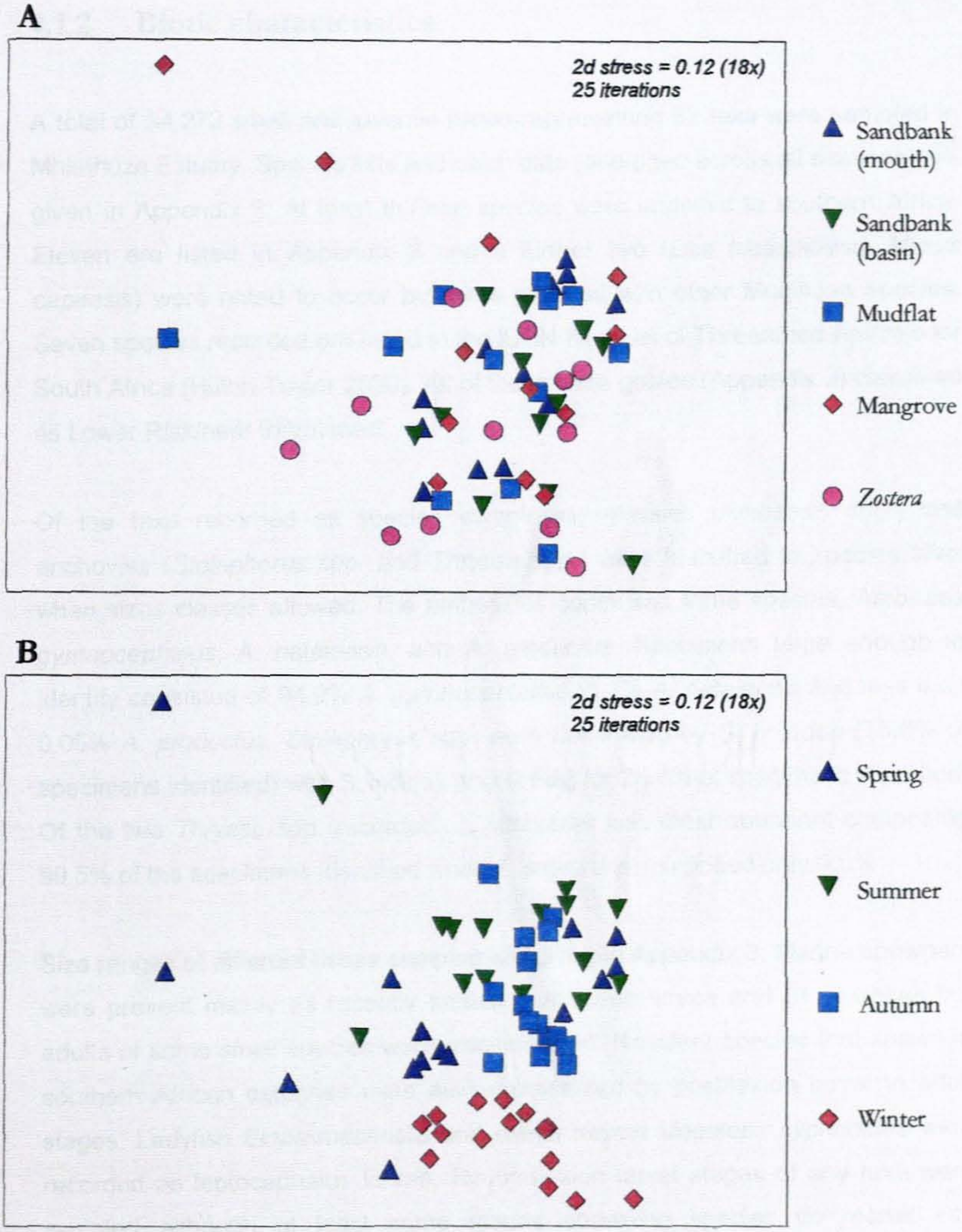


Figure 4.2: MDS ordination of physico-chemical variables (log(x+1) transformed) recorded in Mhlathuze Estuary during twelve months (September 1997 - August 1998). Samples labelled by A) habitat and B) season.

4.1.2 Biotic characteristics

A total of 34,272 small and juvenile fishes representing 82 taxa were sampled in Mhlathuze Estuary. Species lists and catch data (averaged across all seasons) are given in Appendix 3. At least thirteen species were endemic to southern Africa. Eleven are listed in Appendix 3 and a further two (*Liza tricuspidens, Myxus capensis*) were noted to occur but were grouped with other Mugilidae species. Seven species recorded are listed in the IUCN Red List of Threatened Animals for South Africa (Hilton-Taylor 2000). All of these were gobies (Appendix 3) classified as Lower Risk/near threatened.

Of the taxa recorded as species complexes, glassies (*Ambassis* spp.) and anchovies (*Stolephorus* spp. and *Thryssa* spp.) were identified to species level when sizes classes allowed. The ambassids comprised three species, *Ambassis gymnocephalus*, *A. natalensis*, and *A. productus*. Specimens large enough to identify consisted of 94.9% *A. gymnocephalus,* 5.1% *A. natalensis* and less than 0.05% *A. productus*. *Stolephorus* spp. were dominated by *S. holodon* (75.6% of specimens identified) with *S. indicus* accounting for 24.4% of specimens identified. Of the two *Thryssa* spp. recorded, *T. vitrirostris* was most abundant comprising 99.5% of the specimens identified while *T. setirostris* comprised only 0.5%.

Size ranges of different fishes sampled are given in Appendix 3. Marine spawners were present mainly as recently settled postflexion larvae and 0+ juveniles but adults of some small species were also sampled. Resident species that spawn in southern African estuaries were also represented by postflexion larval to adult stages. Ladyfish *Elops machnata* and oxeye tarpon *Megalops cyprinoides* were recorded as leptocephalus larvae. No preflexion larval stages of any taxa were sampled although at least some marine spawning species do recruit into Mhlathuze Estuary at such young developmental stages (Viljoen & Cyrus 2002).

Despite the high number of species sampled over 90% of the total catch (standardised to area seined) comprised of 12 taxa, and only 13 taxa made contributions of over 1% (Table 4.3). Every habitat sampled contributed to supporting one, or several of the most abundant species. Cape stumpnose *Rhabdosargus holubi* contributed 26.2% of the total catch. Over 65% of these fish were sampled as new recruits during August 1998, the last month of the austral winter. Mullet and ambassids accounted for 21.3 and 20.7% of the total catch respectively. Seventy-six percent of all mullet sampled were recorded in spring and winter, while ambassids were more plentiful in summer and autumn (75%). No other taxon contributed more than 10% of the total catch (Table 4.3).

Table 4.3: Fish taxa contributing >1% to the total standardised catch in Mhlathuze Estuary. Percentage contribution from each habitat indicated.

Taxon	total catch		% contribution to taxon total				
	%	Σ %	sandbank (mouth)	sandbank (basin)	mudflat	mangrove	*Zostera*
Rhabdosargus holubi	26.2	26.2	1.7	1.4	1.1	1.9	94.0
Mugilidae spp.	21.3	47.5	58.1	23.2	0.4	12.8	5.5
Ambassis spp.	20.7	68.3	14.3	27.0	0.2	51.1	7.3
Oligolepis keiensis	5.3	73.5	19.3	0.3	20.9	28.0	31.5
Stolephorus spp.	3.5	77.1	26.4	8.4	38.7	20.1	6.4
Arothron immaculatus	2.4	79.5	0.8	0.1	0.6	0.7	97.9
Thryssa spp.	2.3	81.8	15.0	2.5	72.6	9.6	0.2
Rhabdosargus sarba	2.1	83.9	0.3	4.3	1.4	1.1	92.9
Glossogobius callidus	1.8	85.6	15.6	3.4	53.3	18.6	9.1
Diplodus sargus capensis	1.7	87.3	0.2	1.4	0.8	5.2	92.4
Sillago sihama	1.5	88.8	48.9	44.0	2.5	1.3	3.2
Lutjanus fulviflamma	1.2	90.0	0.7	0.2	0.0	0.2	98.8
Favonigobius reichei	1.1	91.1	53.1	29.7	0.3	0.0	16.9

Ninety-four percent of *R. holubi* were sampled in *Zostera*. Several other abundant species occurred almost exclusively in this habitat including *Rhabdosargus sarba* (Natal stumpnose), *Diplodus sargus capensis* (blacktail), *Arothron immaculatus* (blackedged puffer) and *Lutjanus fulviflamma* (Dory snapper). While 58.1% of mullet were sampled from the sandbank at the estuary mouth, ambassids occurred most abundantly in close proximity to mangroves (51.1%) and to a lesser degree at the sandbank in the estuary basin (27.0%). Both sandbanks were important habitats for silver sillago *Sillago sihama*, and tropical sand goby *Favonigobius reichei*. The mudflat supported the highest densities of *Stolephorus* spp. (38.7%), *Thryssa* spp. (72.6%) and river goby *Glossogobius callidus* (53.3%).

Speartail goby *Oligolepis keiensis* occurred fairly consistently at all habitats with the exception of the estuary basin sandbank where it was sampled infrequently in low numbers.

No freshwater species were collected in the estuary. On average, estuarine-dependent marine fishes comprised 76.5% of the fish fauna sampled from different habitats. Marine species constituted 13.1% and estuarine spawning fishes 10.4%. There were no strong deviations from this global average in data averaged across habitats or seasons, with the notable exceptions of fish catches at the mudflat averaged over all seasons, and those in autumn averaged over all habitats (Figure 4.3). The mudflat was distinct in that an average of over 30% of fishes sampled there were estuarine spawners compared with less than 15% at all other habitats. The contribution of these estuarine fishes was most significant in autumn and winter, and was largely brought about by a relative abundance of *G. callidus*. The mudflat also supported a greater proportion of marine species than other habitats, mainly due to the shoaling engraulids, *Thryssa* spp. and *Stolephorus* spp. However, low fish densities at this habitat exaggerated the influence of these fishes. Differences in actual densities of the groups mentioned above from those at other habitats were less significant than differences in their relative contributions. At mangrove fringes, marine species contributed less than 5% of the catch in all seasons except in winter, when over 40% of the fishes sampled were marine. In the *Zostera* beds, catches in summer and autumn comprised over 30% marine species (Figure 4.3).

Autumn was the only season when average contributions of estuarine species and marine species exceeded 20%. During this season there was a general pattern amongst habitats of higher relative proportions of estuarine species, while higher proportions of estuarine-dependent marine species were evident in winter. This was also reflected in absolute densities of fishes in these groups. Mangrove fringes were an exception. Total catches at this habitat in winter were very low, and the relative and actual density of estuarine-dependent marine fishes in winter was far lower than that recorded during any other season.

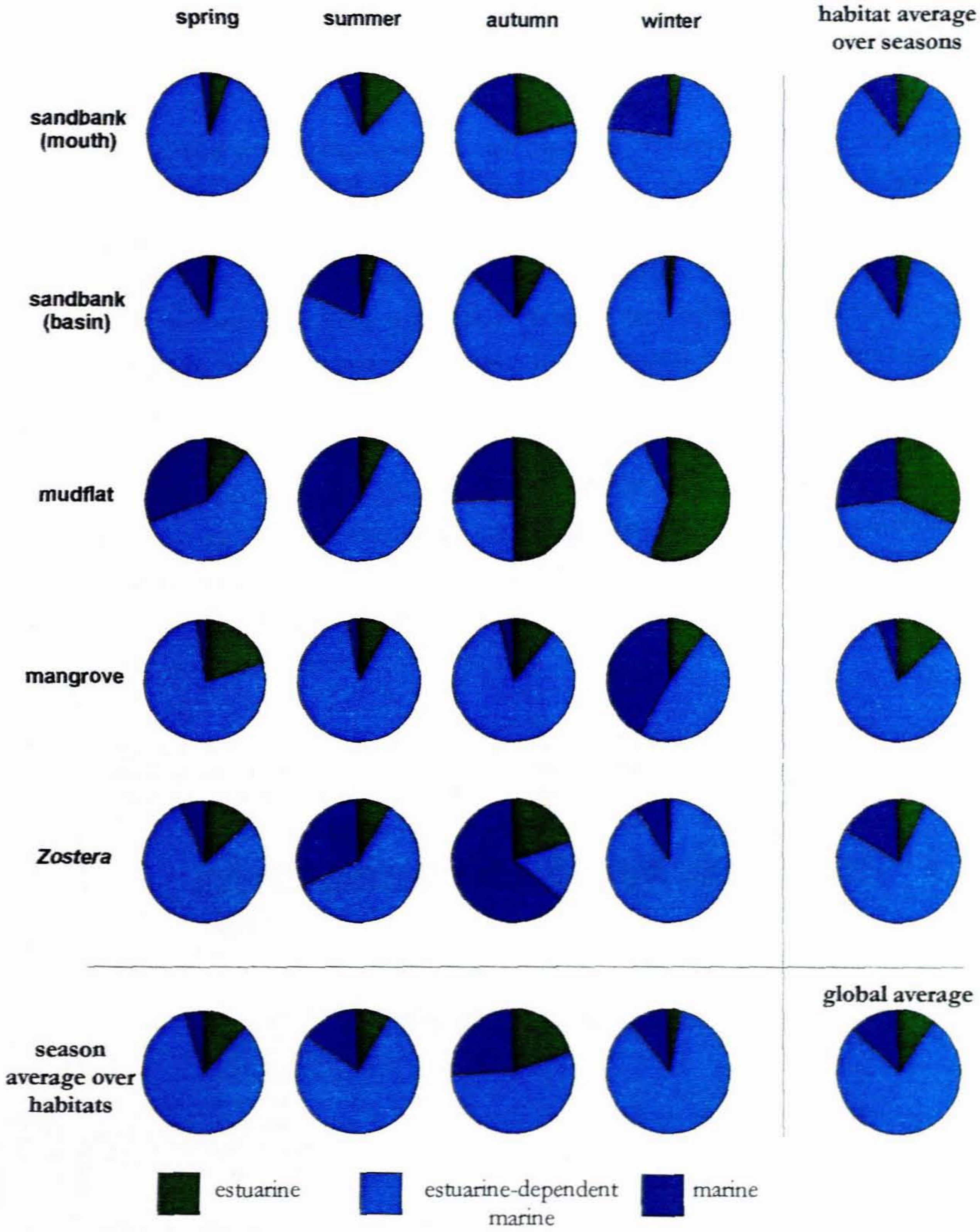


Figure 4.3: Relative contributions of different categories of fishes to different habitats, during different seasons in Mhlathuze Estuary.

Analysis of similarities of species indicated that neither habitat nor season influenced fishes from the same estuarine association group in a comparable manner. In addition, there was no evidence to suggest that fishes with different geographical ranges displayed different trends in their use of habitats or seasonal abundance. The fact that all species recorded in Mhlathuze Estuary had geographical ranges that included subtropical areas rendered the latter analysis largely redundant. Ninety-three percent of taxa were divided equally between fishes known only from subtropical regions and fishes that occur in both subtropical and warm-temperate waters. Seven percent occur around the length of the South African coastline, including the cool-temperate region. In the MDS ordination, with an acceptable stress of 0.13, no patterns were detected in either categorisation of species into estuarine association classes or distribution ranges (Figure 4.4). This was statistically confirmed using ANOSIM. Therefore, amongst different habitats, and across the seasons investigated, there was no difference in abundance of groups of fishes classified either according to estuarine association or biogeographic affinities (Table 4.4).

Table 4.4: Analysis of similarities amongst different categories of fishes sampled from different habitats in Mhlathuze Estuary from September 1997 to August 1998. * = P <0.05, ** = P <0.01, *** = P <0.001, ns = P >0.05, not significant, s = subtropical, w = warm-temperate, c = cool-temperate.

Test	R statistic	P
Differences amongst estuarine association classes		
Global Test	-0.034	ns
Pairwise Tests		
estuarine vs. estuarine-dependent marine	0.042	ns
estuarine vs. marine	-0.056	ns
estuarine-dependent marine vs. marine	-0.048	ns
Differences amongst geographical range classes		
Global Test	0.003	ns
Pairwise Tests		
s vs. sw	0.016	ns
s vs. swc	-0.110	ns
sw. vs swc	-0.033	ns

Means and 95% confidence limits of univariate community indices averaged between habitats and seasonal months are shown in Figure 4.5. Results of ANOVA of these data are given in Table 4.5. The fish assemblage associated with *Zostera* in the estuary had significantly more species and higher species richness

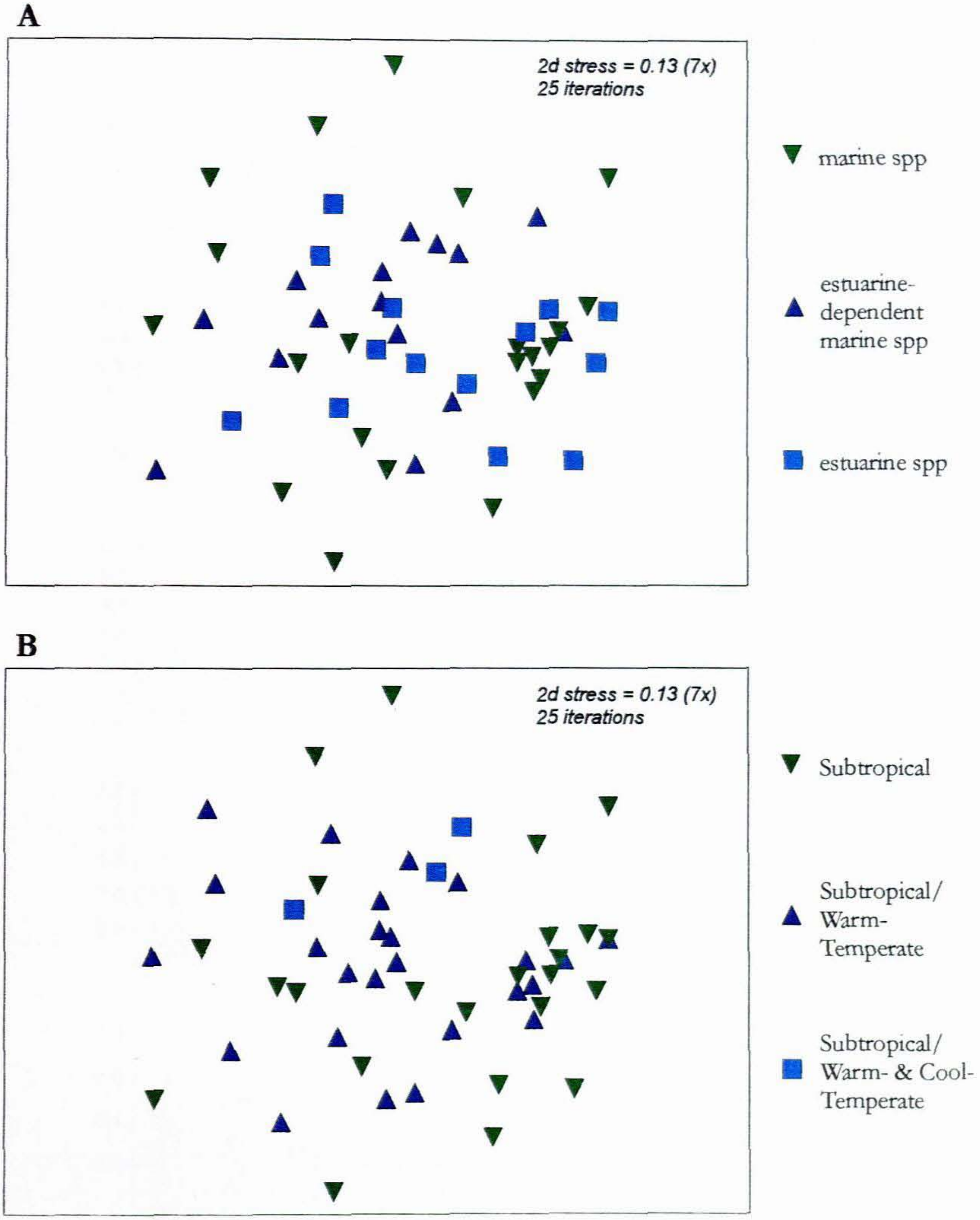


Figure 4.4: MDS ordination of species similarities amongst fishes from habitats in Mhlathuze Estuary during twelve months (September 1997 - August 1998). Samples labelled by A) estuarine association and B) geographical range classes.

S = Number spp. N = Fish Abundance d = Spp. Richness H' = Spp. Diversity J'= Spp. Evenness

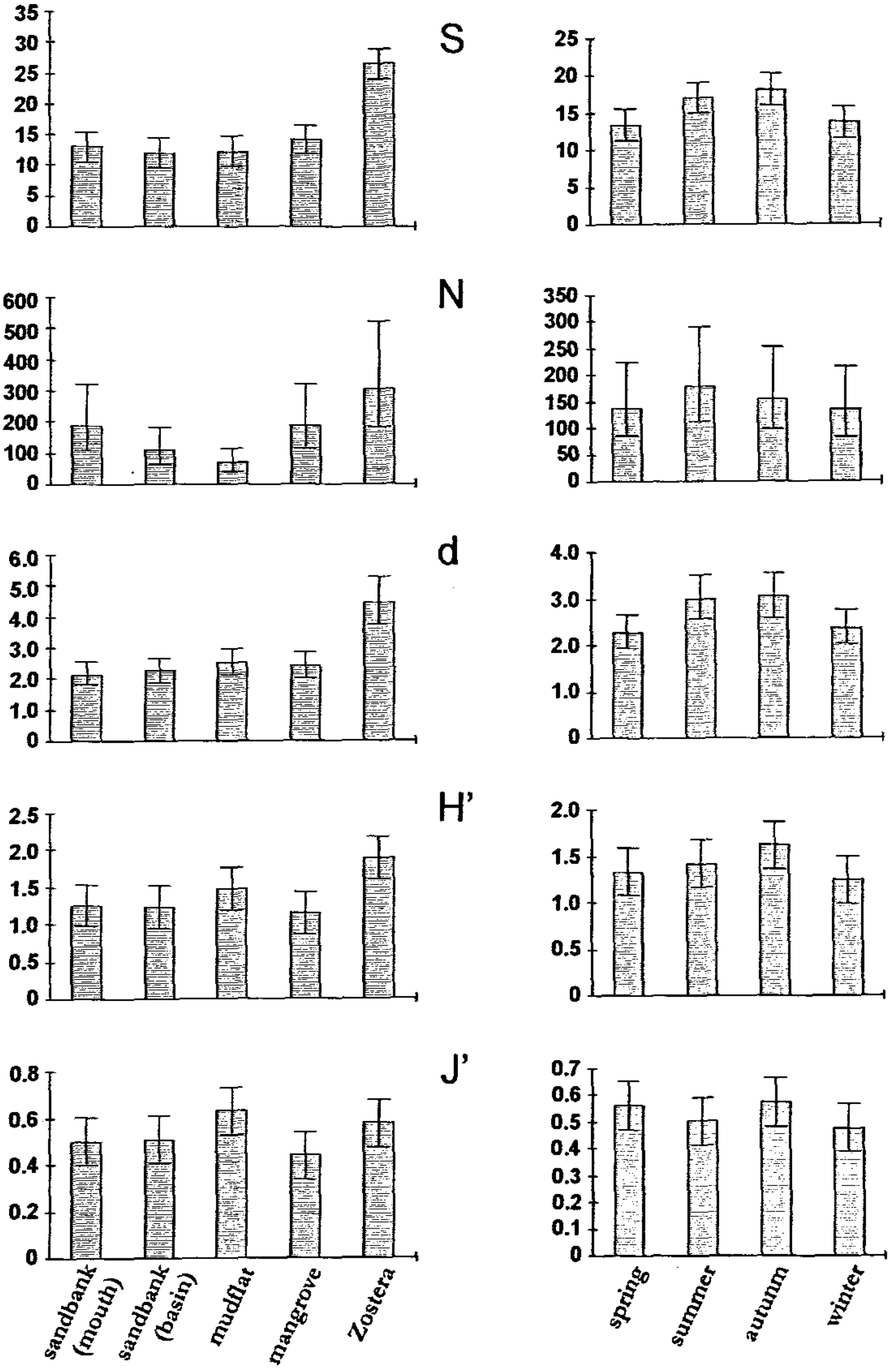


Figure 4.5: Means and 95% confidence limits of fish community indices from assemblages sampled at different habitats and during different seasons in Mhlathuze Estuary.

than all other habitats sampled. Shannon-Wiener diversity in the eelgrass was also significantly higher than at all other habitats with the exception of the mudflat. In terms of numbers of individuals, only *Zostera* and mudflat habitats differed with significantly higher densities sampled in the *Zostera*. There were no significant differences in Pielou's evenness index amongst habitats. Low values of Shannon-Wiener diversity and Pielou's evenness indicated that few species dominated the assemblages at all habitats and seasons. Seasonal differences were only apparent with more species being sampled in autumn than spring and winter, and higher species richness in autumn than spring (Table 4.5).

Table 4.5: F-ratios and significance levels for ANOVA of selected univariate indices of fish assemblages sampled in Mhlathuze Estuary. F-ratios based on the residual mean square error, multiple range analysis based on Tukey's honest significant differences, S = number of species, N = abundance (density), d = Margalef's species richness, H′ = Shannon-Wiener diversity, J′ = Pielou's evenness index, df = degrees of freedom, * P = <0.05, ** = P <0.01, *** = P <0.001, ^{ns} = P >0.05, not significant.

Effect	Index	F statistic	Multiple range tests
Habitat (4 df)	S	27.780^{***}	[*Zostera* > sandbank (basin & mouth), mudflat, mangrove]
	N ($\log_{10}$)	4.967^{**}	[*Zostera* > mudflat]
	d ($\log_{10}$)	11.956^{***}	[*Zostera* > sandbank (basin & mouth), mudflat, mangrove]
	H′	4.564^{**}	[*Zostera* > sandbank (basin & mouth), mangrove]
	J′	2.107^{ns}	
Season (3 df)	S	5.145^{**}	[autumn > spring, winter]
	N ($\log_{10}$)	0.314^{ns}	
	d ($\log_{10}$)	4.076^{*}	[autumn > spring]
	H′	1.661^{ns}	
	J′	1.068^{ns}	
Habitat x Season (12 df)	S	1.317^{ns}	
	N ($\log_{10}$)	1.891^{ns}	
	d ($\log_{10}$)	1.207^{ns}	
	H′	1.468^{ns}	
	J′	1.710^{ns}	

Results of multivariate analyses are presented as the cluster dendrogram (Figure 4.6) and MDS ordination (Figure 4.7) of root transformed fish densities from averaged replicate seine hauls at different habitats. As with the physico-chemical analysis the ordination is presented separately with samples labelled according to habitat (Figure 4.7A) and season (Figure 4.7B). The two-dimensional sample ordination exhibited high stress of 0.24. However, the three-dimensional ordination

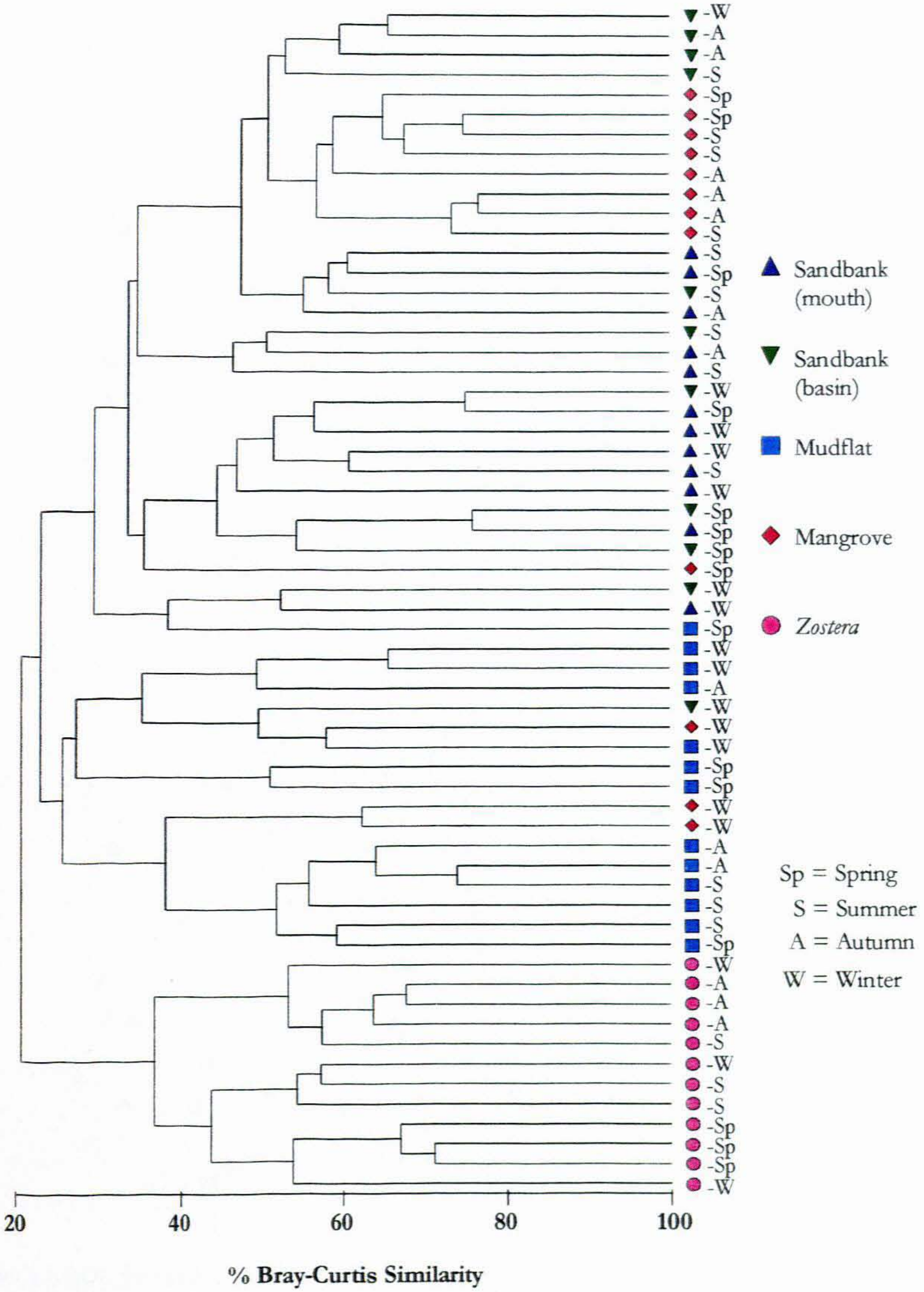


Figure 4.6: Dendrogram of Bray-Curtis similarities in densities of fishes (square root transformed) from habitats sampled during different seasons in Mhlathuze Estuary.

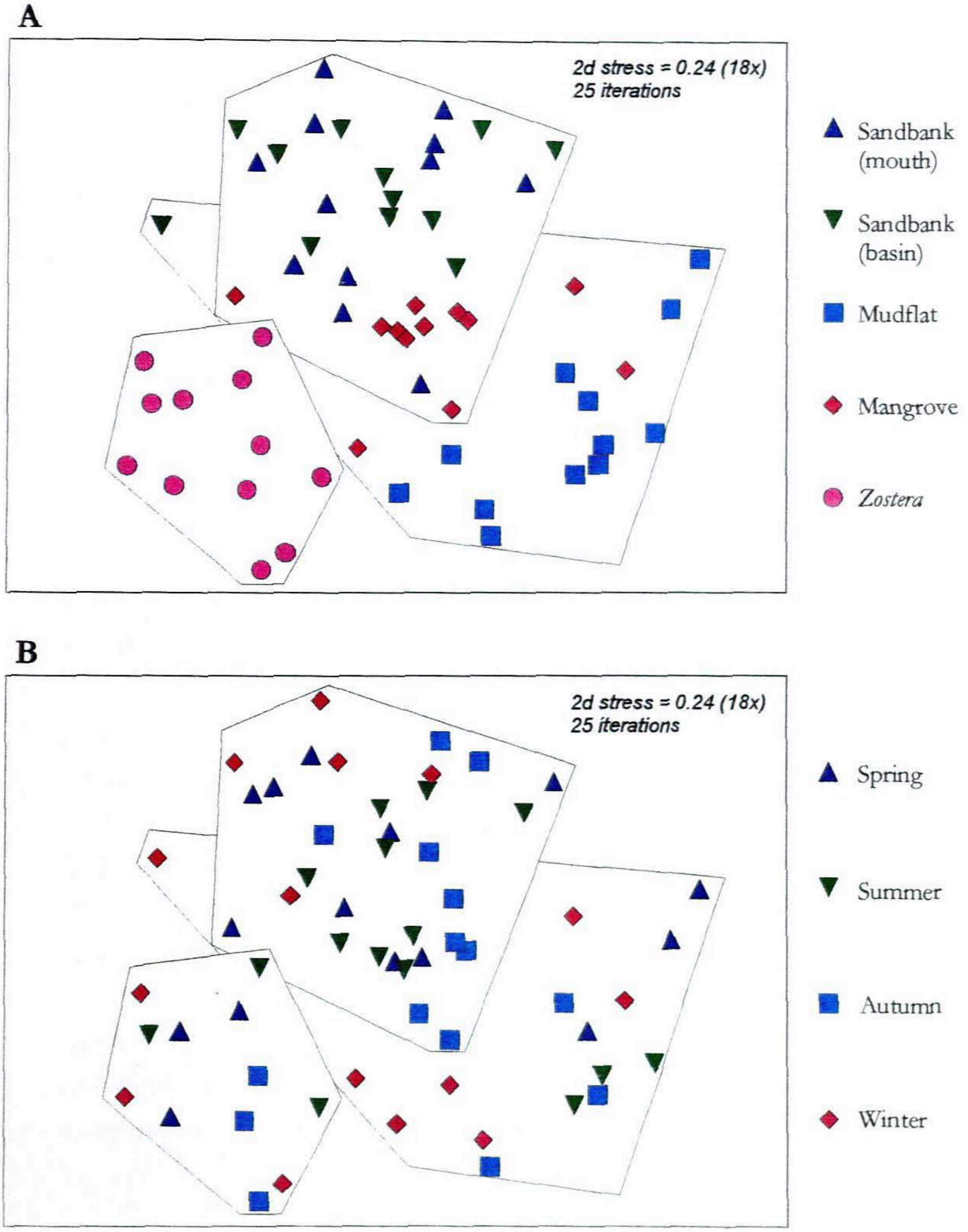


Figure 4.7: MDS ordination of densities of fishes (square root transformed) sampled in Mhlathuze Estuary. Samples labelled by A) habitat and B) season. Blocks represent separate groups at 25% Bray-Curtis Similarity.

was found to be potentially useful with a stress of 0.16 and was examined in order to check the adequacy of the lower-dimensional plot. Groups from cluster analysis (Figure 4.6) were superimposed onto the two-dimensional MDS ordination (Figure 4.7) as suggested by Clarke and Warwick (1994) to check the adequacy and consistency of both representations.

At a low level of similarity (± 25%) three broad groups of fish assemblages were sampled based primarily on habitat rather than seasonal influences (Figure 4.6, Figure 4.7). The fish community in the *Zostera* was distinct from that of any other habitat in the estuary. Amongst the remaining assemblages those associated with the mudflat were different from communities which occurred over the sandbanks or along the mangrove fringes. Although the latter communities appeared broadly similar, those from the mangroves in spring, summer and autumn months formed a distinct cluster within a larger group comprising mainly of fishes sampled from the basin sandbank (Figure 4.6). A seasonal influence was also noted at other habitats. Winter, spring and summer assemblages were distinct from those in autumn in the *Zostera*, and summer and autumn assemblages from the mudflat were distinct from those sampled in winter and spring (Figure 4.6).

Analysis of similarities confirmed that the influence of both habitat and season were significant. A higher Global R statistic for the test of differences amongst habitats indicated that habitat was the principal determinant of the noted differences amongst samples (Table 4.6). There was no significant difference between the fish assemblages sampled over the different sandbanks, and the fish community sampled in the estuary in summer months did not differ from that sampled in autumn months (Table 4.6).

Table 4.6: Analysis of similarities amongst fish assemblages sampled from different habitats in Mhlathuze Estuary. * = P <0.05, ** = P <0.01, *** = P <0.001, ns = P >0.05, not significant.

Test	R statistic	P
Differences amongst habitats		
Global Test	0.723	***
Pairwise Tests		
sandbank (mouth) vs. sandbank (basin)	0.074	ns
sandbank (mouth) vs. mudflat	0.824	***
sandbank (mouth) vs. mangrove	0.528	**
sandbank (mouth) vs. *Zostera*	0.907	***
sandbank (basin) vs. mudflat	0.815	***
sandbank (basin) vs. mangrove	0.528	***
sandbank (basin) vs. *Zostera*	0.843	***
mudflat vs. mangrove	0.730	**
mudflat vs. *Zostera*	0.944	***
mangrove vs. *Zostera*	0.917	**
Differences amongst seasons		
Global Test	0.389	***
Pairwise Tests		
spring vs. summer	0.326	**
spring vs. autumn	0.467	***
spring vs. winter	0.363	**
summer vs. autumn	0.185	ns
summer vs. winter	0.407	**
autumn vs. winter	0.637	***

Exploratory analysis using SIMPER to determine discriminating species indicated that average similarity amongst mangrove samples was the highest of all habitats. In three of the five habitats investigated, contributions from only two taxa accounted for approximately 80% of the similarity within the fish assemblages (Table 4.7). Mullet and ambassids were typical of both sandbanks. Ambassids also typified mangrove fringes in the estuary and at the mudflat engraulids, *Stolephorus* spp. and *Thryssa* spp., and gobies, *G. callidus* and *O. keiensis*, all contributed over 15% to the assemblage similarly. Fishes important in clustering samples from the *Zostera* were *R. holubi*, *A. immaculatus* and *L. fulviflamma*. All of these fishes occurred abundantly as very young juveniles and were not typically associated with any other habitat (Table 4.7).

Table 4.7: Average similarity within groups and species contribution >10% similarity. Average abundance as fish per 100 m^2.

Taxon	average abundance	% contribution	cumulative %
sandbank (mouth) **average similarity = 27.87**			
Mugilidae spp.	162.08	66.65	66.65
Ambassis spp.	38.89	12.77	79.42
sandbank (basin) **average similarity = 26.24**			
Ambassis spp.	73.40	65.87	65.87
Mugilidae spp.	64.65	16.81	82.68
mudflat **average similarity = 25.51**			
Stolephorus spp.	17.95	22.69	22.69
Thryssa spp.	21.94	20.77	43.46
Glossogobius callidus	12.45	19.92	63.38
Oligolepis keiensis	14.40	16.60	79.98
mangrove **average similarity = 33.27**			
Ambassis spp.	138.76	71.91	71.91
Oligolepis keiensis	19.28	10.52	82.44
Zostera **average similarity = 26.03**			
Rhabdosargus holubi	323.26	28.93	28.93
Arothron immaculatus	30.80	17.12	46.05
Lutjanus fulviflamma	16.05	10.29	56.34

Taxa that played important roles in forging dissimilar fish assemblages at different habitats are given in Table 4.8. The relative abundance of mullet and ambassids at sandbanks was important in contributing to dissimilarities amongst fish assemblages from these habitats and those sampled elsewhere in the estuary. Over 40% of the dissimilarity between the basin sandbank and the mangrove, as well as the mudflat and mangrove fishes was due to the relative abundance of *Ambassis* spp. along the mangrove fringes. The paucity of ambassids and abundance of *R. holubi* distinguished *Zostera* samples from those taken at the mangroves. *Rhabdosargus holubi* was also the main discriminating species when assemblages associated with *Zostera* were compared with those from other habitats (Table 4.8).

Table 4.8: Average dissimilarity between groups and species contributing >10% dissimilarity. Average abundance as fish per 100 m^2.

Taxon	average abundance	average abundance	% contribution	cumulative %
Groups compared →	sandbank (mouth)	sandbank (basin)	average dissimilarity 77.15	
Mugilidae spp.	162.08	64.65	41.46	41.46
Ambassis spp.	38.89	73.40	24.96	66.43
Groups compared →	sandbank (mouth)	mudflat	average dissimilarity 90.94	
Mugilidae spp.	162.08	1.06	37.45	37.45
Ambassis spp.	38.89	0.43	11.45	48.90
Groups compared →	sandbank (basin)	mudflat	average dissimilarity 93.36	
Ambassis spp.	73.40	0.43	28.82	28.82
Mugilidae spp.	64.65	1.06	15.44	44.26
Thryssa spp.	0.76	21.94	10.32	54.58
Groups compared →	sandbank (mouth)	mangrove	average dissimilarity 81.36	
Ambassis spp.	38.89	138.76	33.32	33.32
Mugilidae spp.	162.08	35.80	31.99	65.31
Groups compared →	sandbank (basin)	mangrove	average dissimilarity 86.69	
Ambassis spp.	73.40	138.76	43.46	43.46
Mugilidae spp.	64.65	35.80	18.93	62.39
Groups compared →	mudflat	mangrove	average dissimilarity 74.13	
Ambassis spp.	0.43	138.76	43.67	43.67
Groups compared →	sandbank (mouth)	*Zostera*	average dissimilarity 88.60	
Rhabdosargus holubi	5.73	323.26	24.72	24.72
Mugilidae spp.	162.08	15.49	23.37	48.09
Groups compared →	sandbank (basin)	*Zostera*	average dissimilarity 88.16	
Rhabdosargus holubi	4.86	323.26	27.39	27.39
Ambassis spp.	73.40	19.94	16.43	43.82
Mugilidae spp.	64.65	15.49	11.38	55.20
Groups compared →	mudflat	*Zostera*	average dissimilarity 92.67	
Rhabdosargus holubi	3.65	323.26	28.61	28.61
Arothron immaculatus	0.17	30.80	10.97	39.59
Groups compared →	mangrove	*Zostera*	average dissimilarity 86.83	
Ambassis spp.	138.76	19.94	25.95	25.95
Rhabdosargus holubi	6.43	323.26	25.39	51.35

4.2 Discussion

4.2.1 Abiotic characteristics: the role of physico-chemical properties as environmental factors influencing spatial and temporal distribution of fishes in Mhlathuze Estuary

Mhlathuze Estuary is a subtropical estuarine bay in the sense described by Whitfield (1992). Water temperatures in the lower and middle reaches of such bays are strongly influenced by marine conditions (Day 1981d). Temperatures recorded in Mhlathuze Estuary reflected a similar trend to those measured during the same time period in the nearshore marine environment at Richards Bay (Figure 4.1, Figure 2.2). A seasonal influence was clear but there was no difference in water temperature from different sites in the estuary. Dissolved oxygen and pH also displayed little spatial variation, but both were seasonal and inversely related to temperature. Reduced solubility of oxygen may have been a direct result of warmer water temperatures, but consistently low dissolved oxygen in autumn months suggested that some other factor, or set of factors, was also influential in the fluctuation of this variable. A seasonal influence on pH was indicated by statistical analysis and the time series plot suggested a consistent trend in pH variation over the twelve months sampled. As with other variables there was no significant spatial variation in the habitats sampled.

Hypersaline conditions (>40‰) are not recorded in estuarine bays due to the efficient exchange of water between the marine and estuarine environments (Whitfield 1992). Salinities in Mhlathuze Estuary never exceeded 38‰ during this study. The near marine conditions that typified the entire estuary basin were due to the nature of the system as an estuarine bay, as well as the fact that sampling was largely restricted to incoming and high tides when marine influence was greatest. Measurements below 25‰ were made but these were limited to periods of high flow in the Mhlathuze River and, probably to a lesser extent, the Mtantatweni Stream. Over the course of the study there were no significant salinity differences amongst sites but on average, salinities were lowest in summer and spring during months of highest freshwater input. The influence of river flows on estuarine turbidities was less apparent and generated neither a spatial nor a temporal influence.

Multivariate analyses supported the general findings of univariate analyses and indicated no spatial variation. However, a strong seasonal influence in the suite of physico-chemical parameters was measured. In winter, conditions in the estuary were significantly different to all other seasons, but there was little variation amongst spring, summer and autumn. This suggested that in this subtropical system there were really only two seasons, with water being warmer, less saline and closer to neutral pH in the austral spring, summer and autumn than in winter months. Confirmation of this would require similar seasonally replicated data sets over more than a single year.

To summarise, physico-chemical data gathered from September 1997 to August 1998 showed that Mhlathuze Estuary was marine dominated and well mixed with little spatial heterogeneity in water quality variables. This may be attributed to the basin morphology, strong marine influence (large tidal prism) and low discharge volumes of the main inflows during most of the study period. The major area of the estuary comprises a shallow embayment. Tidally driven mixing in the shallow water column prevents any significant vertical stratification. The system does not have the geomorphic characteristics of a drowned river valley or a classic coastal plain estuary in the sense described by Day *et al.* (1989). As a consequence, most of the estuary, and certainly the embayment area sampled in this study, does not exhibit axial gradients in water quality variables. The most significant inflow of freshwater from Mhlathuze River is seasonal but largely bypasses the estuarine embayment via a canal that discharges near the mouth of the system. Although not evident from data here, horizontal salinity gradients are restricted to these canalised sections with the zone of mixing retreating or advancing depending on the state of tide and flow of the river (Wepener & Vermeulen 1998).

Based on the above, it can be concluded that it is unlikely that the water quality variables measured would have markedly influenced spatial distribution of fishes within the estuary, although a temporal (seasonal) effect could not be discounted. Clearly, water quality parameters would have played an important role in influencing what types of fishes were in the estuary to begin with. Salinity ranges in the system were within the tolerance limits of most estuarine and marine spawning fishes which utilise KwaZulu-Natal estuaries (Whitfield *et al.* 1981), but they would have precluded utilisation of the estuary by local freshwater species. Water temperatures recorded were typical of the seasonal range that occurs in KwaZulu-Natal estuaries (Cyrus 1988) and were within the tolerance limits of the juvenile fishes present (Cyrus & Blaber 1987b). There is little published information on the effect of pH or dissolved oxygen on fishes in South African estuaries, but mass fish moralities have been attributed to low concentrations of the latter in some systems (Whitfield 1998, 1999). South African water quality guidelines for estuarine waters have not been determined but those for coastal marine waters set a Target Value of ≥6 mg/l dissolved oxygen for aquatic biota (DWAF 1995). This standard was met in 92% of the measurements in the estuary and it is unlikely that dissolved oxygen was limiting for any of the fishes sampled. Levels of turbidity at which changes in juvenile fish assemblages from KwaZulu-Natal estuaries are most obvious, are 10, 50 and 80 NTU (Cyrus & Blaber 1987a,b). In a regional context Mhlathuze can therefore be regarded as a semi-turbid estuary with a mean turbidity falling within the range of 10 to 50 NTU.

4.2.2 Biotic characteristics: occurrence of fishes in structurally different habitats in Mhlathuze Estuary

4.2.2.1 The fish fauna and the role of Mhlathuze Estuary as a nursery

The fine-mesh seine net was effective in sampling fishes in the size range investigated (<5 to 50 mm SL). Taxa commonly encountered in subtropical estuaries of southern Africa were recorded in Mhlathuze Estuary, including several endemic forms and threatened species worthy of conservation. Newly settled postlarvae and young juveniles of many taxa, as well as the adults of several small-bodied species were sampled. These assemblages are not well sampled using gill nets and larger mesh seine or trawl nets, which have historically been favoured by researchers sampling fish in South African estuaries. Numerical dominance by few species as was recorded here, is characteristic of estuarine ichthyofaunas (Haedrich 1983, Whitfield 1994b), even in assemblages of fishes at early developmental stages (Little *et al.* 1988, Robertson & Duke 1990, Neira *et al.* 1992, Tzeng & Wang 1992, Harris & Cyrus 1995). Results concurred with the findings of Quinn (1980), who noted that fewer than six species often comprise about 70% of fish abundances in temperate and subtropical systems, although many more species than this are usually present.

In recent years the Mhlathuze Estuary has been sampled using more conventional, larger meshed equipment and Weerts and Cyrus (1998) have given an account of the system's ichthyofauna. Over a two-year period in 1996 and 1997, the system was sampled eight times on a seasonal basis. Seventy-two species were recorded which, when grouped into the same family and genus complexes as were necessary for the present study, were reduced to a list of 63 taxa. Forty-two of these were reported from the present study while 21 were not taken from any of the habitats sampled. The absence of four of the latter, two pipefishes, *Microphis fluviatilis* and *Microphis brachyurus*, and two gobies, *Awaous aeneofuscus* and *Glossogobius giuris*, can be attributed to their preferences for less saline habitats than those sampled during this study. These species are largely restricted to estuarine headwaters in the Mhlathuze (*pers. obs*). Forty species sampled during the present study were not recorded in 1996/7. All but seven of these had strong affinities for eelgrass habitat or were small, cryptic fishes such as gobies. Their absence from samples taken in 1996/7 may be ascribed to lack of sampling in the *Zostera*, and inefficiency of the equipment used to sample small and cryptic species.

While discrepancies in the efficiency of gears used for sampling fishes explains some differences in species lists from the two studies, these data also give an indication of the estuarine habitat utilisation of fishes concerned, particularly if consideration is given to the size classes recorded. For example, elf *Pomatomus saltatrix*, mangrove snapper *Lutjanus argentimaculatus* and yellowfin needlefish *Strongylura leiura* are sampled commonly in Mhlathuze Estuary at size classes over 100 mm SL, using gill nets and large seines that target larger fishes (Weerts & Cyrus 1998, CRUZ unpublished data). However, in extensive monthly surveys none of these fishes was recorded with the fine-mesh small seine, which was effective in sampling other species at sizes ≤50 mm SL. This suggests that *P. saltatrix*, *L. argentimaculatus* and *S. leiura* recruit into Mhlathuze Estuary during late juvenile, rather than early developmental stages. The opposite is apparently true of *D. sargus capensis*. The species occurred in high densities in the *Zostera* beds as recently settled postlarvae, but evidently emigrated from the system as very young juveniles at about 20 mm SL. It was not recorded in the eelgrass, or any other habitat as older juveniles in this study and has not been sampled in the system with gear targeting larger fishes (Weerts & Cyrus 1998, CRUZ unpublished data). The biology of *D. sargus capensis* in subtropical estuaries may differ from warm-temperate South African systems where it remains associated with *Zostera* beds, even at sizes above 50 mm SL (Whitfield 1998).

All fishes sampled in Mhlathuze Estuary had distributional ranges which included subtropical waters. A distinct subtropical/warm-temperate boundary appears to separate South African estuarine fish communities between the Great Kei and the Mbashe Estuary (Whitfield 1998, Maree *et al.* 2000). The degree to which warm-temperate species penetrate north into subtropical waters in winter (and subtropical fishes south in summer) is likely to decrease more steeply with increasing distance from this boundary. The fish assemblage of Mhlathuze Estuary, located well within the subtropical bioregion, is therefore less likely to be influenced by high winter abundances of warm-temperate species than subtropical systems further south.

The occurrence of *R. holubi* as the most abundantly sampled species in Mhlathuze Estuary was surprising. Although it occurs commonly in both open and temporarily open/closed systems along the coast of KwaZulu-Natal (Whitfield 1998, CRUZ unpublished data), it is not often found in higher numbers than mullet. The presence of *Zostera* was clearly the reason for the abundance of this species in Mhlathuze Estuary. The strong preference of *R. holubi* for *Zostera* as a nursery habitat has been reported from estuaries over the greater length of the South African coast (Whitfield *et al.* 1989, Ter Morshuizen & Whitfield 1994, Paterson & Whitfield 2000). This would suggest that the absence of submerged aquatic macrophytes from most KwaZulu-Natal estuaries has implications for stocks of *R. holubi* in this region. However, the high abundance of postlarval and juvenile *R. holubi* in Mhlathuze Estuary indicated either very high species fecundity or abundant adult populations in the coastal waters of KwaZulu-Natal. Mass recruitment of *R. holubi* into Mhlathuze eelgrass in late winter and spring was earlier than that into similar habitat in Swartkops Estuary in the eastern Cape, where peak recruitment occurred in spring and summer (Beckley 1983). This is explained by the fact that the species spawns earlier in KwaZulu-Natal coastal waters than in the eastern and western Cape (Wallace 1975b, Whitfield 1998).

Abundances of mullet (0+ juveniles) and ambassids (postflexion larvae to adults) were also seasonal but less so than *R. holubi*. Mullet showed a preference for open water habitat on intertidal sandbanks while ambassids were closely associated with mangrove fringes. These two taxa are the most abundant fishes in many KwaZulu-Natal estuaries (Wallace 1975a) and play important roles in the ecology of these systems as prey species. The separation of their peak abundances in Mhlathuze Estuary into different seasons, with mullet dominating in winter and spring, and ambassids in summer and autumn, is therefore important. Tzeng and Wang (1992) noted similar ecological separation of dominant species by recruitment timing in young juvenile fishes in a Taiwan estuary. Surprisingly, ichthyoplankton surveys at the mouth of Mhlathuze Estuary (Viljoen *et al.* in prep.) and other major systems along the KwaZulu-Natal coast (Kosi Bay, Harris *et al.* 1995; St Lucia Estuary, Harris & Cyrus 1995; Richards Bay Harbour, Harris &

Cyrus 1997; Durban Harbour, Harris & Cyrus 1999) have found mullet and ambassids to be rare, contributing less than 2% of fish larvae. Much higher recruitment of larval and postlarval mullet into eastern Cape estuaries has been reported by Whitfield (1994c). Common species of mullet recruit most abundantly into KwaZulu-Natal estuaries as young juveniles rather than as early postlarvae (Wallace & van der Elst 1975). Standard techniques of sampling ichthyoplankton do not effectively sample juvenile fishes and therefore probably underestimate recruitment by these species in KwaZulu-Natal systems. Ambassids in southern Africa can spawn in estuaries (Whitfield 1998) and may develop without an obligatory marine phase. If eggs and larvae are retained in an estuary, recruitment via the mouth from marine waters need not occur. This might account for the apparent low recruitment of ambassids, but abundance of juveniles and adults, in KwaZulu-Natal estuaries.

Discussion above has focussed on individual habitats and their associated fish faunas, which is the main topic of this study. However, the equal weighting given to habitats skews the true abundances of juvenile fishes in the estuary. Using the average densities of different fishes reported from the study period, and estimates of habitat areas from Chapter 2, an indication could be gained of the numerical standing stock of each species in the estuary. Sandbanks were regarded as a single habitat type and densities of fishes averaged across the two areas sampled. Figures are speculative but are given to illustrate the importance of accounting for habitat preferences in surveying estuarine fish faunas. The ten most numerous taxa and habitat totals for all small fishes are listed in Table 4.9. *Thryssa* spp. did not occur in particularly high densities in seine net catches compared to several other species (Table 4.3). However, they utilised large expanses of estuarine mudflat, and were on average the most numerous small fishes in Mhlathuze Estuary (Table 4.9). Mullet and *Stolephorus* spp. followed as the most plentiful fishes in the system. *Rhabdosargus holubi* was on average the seventh most abundant fish in terms of absolute numbers in the entire estuary. Although this species occurred in higher densities than any other fish, it was largely restricted to *Zostera* beds (Table 4.3) which covered only 5 ha of the estuarine area.

Table 4.9: Standing stock (total numbers) of the ten most abundant fish taxa (≤50 mm SL) in Mhlathuze Estuary, averaged over 12 monthly samples. Total numbers (monthly average) of all small fishes in shallow water habitats indicated. - = <500 fish.

Taxon	Numbers of fish (x 10^6)				
	sandbanks 70 ha	mudflat 450 ha	mangroves 14 ha	*Zostera* 5 ha	Total 539 ha
Thryssa spp.	0.019	0.988	0.004	-	1.010
Mugilidae spp.	0.794	0.048	0.050	0.008	0.899
Stolephorus spp.	0.056	0.808	0.013	0.001	0.879
Oligolepis keiensis	0.047	0.648	0.027	0.011	0.733
Ambassis spp.	0.393	0.019	0.194	0.010	0.617
Glossogobius callidus	0.016	0.560	0.006	0.001	0.583
Rhabdosargus holubi	0.037	0.164	0.009	0.162	0.372
Leiognathus equula	0.011	0.214	0.002	-	0.226
Solea bleekeri	0.003	0.201	0.003	-	0.207
Sillago sihama	0.063	0.022	-	-	0.086
↓	↓	↓	↓	↓	↓
Total	1.570	4.004	0.343	0.264	6.181

Estuarine-dependent marine fishes in Mhlathuze Estuary were more abundant than any other group in terms of numbers of individuals. Salinity preferences of local freshwater species prohibited their utilisation of the estuary embayment throughout the year. All fishes occurred most abundantly as 0+ juveniles, although some small species were also sampled as adults. This was a reflection of the sampling gear which was not efficient in catching large fishes. Nevertheless, based on these results, those of Weerts and Cyrus (1998) and unpublished data (CRUZ), it can be confidently stated that Mhlathuze Estuary plays a primary role as nursery for juveniles of estuarine-dependent marine fishes. Although the properties of water sampled in the embayment are dominated by a strong marine influence, the system is estuarine in character, especially in terms of the structure of habitat available to juvenile fishes.

4.2.2.2 The influence of habitat and season in structuring the fish assemblage

From the above it is apparent that salinity and other physico-chemical determinants in Mhlathuze Estuary were primarily influential in establishing an environment suitable for a wide range of marine and estuarine fish species.

Analysis of these variables indicated they would have little effect in structuring the system's fish assemblage in spatial terms. Within these prevailing conditions the influence of habitat structure on the distribution of these fishes could be investigated, safely assuming that noted patterns would be free from influences of non-structural elements of the environment. Temporal effects could not be discounted as physico-chemical determinants varied seasonally. In this regard the breeding biologies, growth rates, recruitment times and emigration periods of species involved were also expected to be confounding variables.

Groups of fishes with different estuarine dependencies did not have different affiliations for specific habitats. Little variation was noted in the relative contribution of different estuarine categories of fishes at different habitats. Low overall fish densities exaggerated the few exceptions. The distribution of different categories of fishes has been found to vary in estuaries elsewhere (Yáñez-Arancibia *et al.* 1985), but this can usually be related to factors such as salinity or distance from the mouth (Loneragan *et al.* 1989, Loneragan & Potter 1990). Being euryhaline, the estuarine component of the fish fauna is usually less impacted than the marine or freshwater components which are respectively drawn to more saline or fresher waters (Loneragan & Potter 1990).

Salinity is an important factor in the spatial and temporal distribution of fishes in estuaries, either via direct osmoregulatory pressures, or indirectly through its influence on food resources (Whitfield 1998). Although tempting to cite lower river flows and higher salinities as possible reasons for the increased contribution of marine fishes in autumn, this was as much a reflection of decreased abundance of estuarine-dependent marine species as increased abundance of marine fishes. Wallace and van der Elst (1975) noted that peak immigration of postlarval and juvenile estuarine-dependent marine fishes into KwaZulu-Natal estuaries occurred in late winter and spring, a fact verified by results of this study. By the following autumn most of these fishes would have outgrown the size range analysed here, exaggerating the relative contribution of marine (and estuarine) species by their absence. Winter, when salinities in the estuary were highest, saw mass recruitment and marked increases in densities and relative contributions of estuarine-dependent marine fishes, but reductions in densities and percentages of marine species. Species growth rates, emigration and immigration are clearly factors that should be considered when investigating seasonal influences on estuarine fish assemblages. Such influences appear to act at the level of individual species rather than on broad groups of fishes with similar dependencies on estuaries. As noted by Ferrel *et al.* (1993) and Gray *et al.* (1996), it is unlikely that all species display temporally concordant changes in abundances.

Bell *et al.* (1984) reported seasonal peaks in numbers of species and individuals from a temperate mangrove system in Australia and attributed these to recruitment periods of estuarine-dependent marine fishes. However, this is not ubiquitously the case and it is probable that latitude plays a role. Little *et al.* (1988) and Kimani *et al.* (1996) found no significant temporal variation in community diversity indices in Kenyan mangroves. Seasonal differences in the number of species and species richness were observed in Mhlathuze Estuary. However, it was apparent that the influence of season held little sway on community characteristics of fish assemblages compared with that of habitat. Both univariate and multivariate analyses indicated that the fish assemblages were structured primarily on the basis of different habitats within the estuary. Only the two sandbank habitats, which were structurally similar but varied in distance from the estuary mouth, did not support statistically unique fish assemblages. Seasonal variability was displayed within groups of fishes associated with different habitats, but at a broad level it is difficult to explicate the influences of seasonally different environmental conditions and the life cycles of contributing species. Seasonal variations in physical conditions in Mhlathuze Estuary are probably less extreme that those that might be expected in more classical temperate or tropical estuaries. Thus, changing environmental conditions probably play less of a role in influencing the fish assemblage than in temperate estuaries further south. Even within temperate South African estuaries, seasonal temperature variation does not influence fish community structure to the same degree as in boreal systems where marked differences in biological activity occur between summer and winter (Whitfield

1999). Also, Mhlathuze Estuary is not subjected to seasonally consistent extreme summer flows as rainfall along the east coast of South Africa does not approach levels of some of the more tropical areas. The Goedertrouw Dam captures much of the summer flow in the river, especially early in the season, to replenish the reservoir after winter releases for downstream agricultural, industrial and domestic consumption. Furthermore, the bathymetry of Mhlathuze Estuary is such that high river flows largely bypass the estuarine embayment, and the strongly contrasting conditions typical of wet and dry seasons in some parts of the world do not occur here.

4.2.2.3 Small and juvenile fishes associated with different habitats

Four broad habitat types, which can be defined on the basis of structural elements of the estuarine environment, were identified in the embayment section of Mhlathuze Estuary. All contribute to fish taxa that are important in structuring the fish assemblage of the estuary as a whole. The high abundance of relatively few taxa in the estuary masks an added value of these different habitats. Many species that were not important in terms of total abundance in the estuary showed marked preferences for a single habitat type. In most cases these species did not dominate, or characterise the actual habitat that was important to them, and yet they were often almost completely restricted in their distribution to the habitat in question.

Eelgrass

Eelgrass harboured more species and higher species richness than all other habitats in Mhlathuze Estuary. With the exception of the mudflat, which had a low number of species and low fish densities, *Zostera* also supported greater species diversity than other habitats. Hanekom and Baird (1984) found no significant differences in the number of fish species, or overall fish densities between *Zostera* and unvegetated habitats in the eastern Cape Kromme Estuary. In a mangrove ecosystem in southern Florida USA, Sheridan (1992) recorded higher fish densities in unvegetated open water habitats than seagrass beds. However, the importance of different types of seagrasses as nursery habitats has been confirmed by a multitude of other studies from many regions of the world (Pollard 1984). In most systems higher numbers of species and densities utilise seagrass compared with unvegetated habitats (Branch & Grindley 1979, Beckley 1983, Heck *et al.* 1989, Whitfield *et al.* 1989, Costa *et al.* 1994, Edgar & Shaw 1995). While this was true for species numbers in Mhlathuze Estuary, fish densities in the *Zostera* were only significantly higher than those at the mudflat. In a study of eight estuaries within the same 300 km stretch of Australian coastline, Gray *et al.* (1996) consistently found more fish species in seagrass habitats, but larger numbers of individuals were sampled over open sand at some localities. Furthermore, the degree to which vegetated and unvegetated areas support different abundances of fish has been found to vary with distance from seagrass and depth of habitat investigated. Ferrel and Bell (1991) found that *Zostera* supported more individuals than bare sand over 100 m away. However, they recorded no difference in the abundance of fish in *Zostera* and adjacent sandflats within 10 m. Jenkins *et al.* (1997) reported significantly different fish abundances amongst deep subtidal vegetated and unvegetated habitats, but not amongst *Zostera* and unvegetated habitats in shallow subtidal and intertidal zones. At sites sampled in Mhlathuze Estuary distance from *Zostera* was not likely to have played a role in the observed differences or similarities in fish densities at other habitats. However, the nature of substratum sampled, or proximity to shoreline (and therefore depth) may have been influential. Unvegetated, sandy, littoral habitats supported similar fish densities as eelgrass, but unvegetated mudflats a distance from the shoreline supported significantly lower numbers of individuals.

Zostera in Mhlathuze Estuary is an important nursery to a wide range of fishes. Juveniles of most species predominated, as is the case in seagrass habitats around the world. Hypotheses proposed by researchers to explain the apparent preference for seagrasses as nursery habitats have been summarised by Nagelkerken *et al.* (2000). Those most applicable here, are probably the structural complexity of the habitat in providing shelter against predators and an abundance of food organisms for juvenile fishes. In Mhlathuze Estuary, the fish assemblage associated with *Zostera* was the most distinct of all habitats. It comprised over 75% of all species recorded in the system but was typified by juvenile *R. holubi*, *A. immaculatus* and *L. fulviflamma*. Twenty taxa recorded in the *Zostera* did not occur in any other habitat in the estuary. These included *Pelates quadrilineatus*, *Siganus sutor*, *Lethrinus* spp., *Sebastapistes strongia*, *Epinephelus malabaricus*, *Acentrogobius audax*, *Crenidens crenidens*, *Hippichthys cyanospilos*, and several Labridae and Scaridae species. Some of these taxa were forms more typically associated with reef habitat than estuaries, a phenomenon noted to occur in several other regions of the world (Kimani *et al.* 1996, Nagelkerken *et al.* 2000). Many other species, although they also occurred in non-*Zostera* habitats, were found in highest densities in the eelgrass. Of particular interest was the marked preference shown for this habitat by juveniles of families that are highly targeted as recreational angling fishes. This was illustrated by the fact that over 80% of sea breams (Sparidae), emperors (Lethrinidae), snappers (Lutjanidae), rockcods (Serranidae) and barracudas (Sphyraenidae) were sampled in the eelgrass.

Pollard (1984) noted a common suite of fish families that utilise seagrass habitats from studies worldwide. Many of these families were sampled in the *Zostera* bed in the Mhlathuze. Eight of the ten dominant fish families found in Australian seagrass habitats (Pollard 1984) were recorded here including Syngnathidae, Gobiidae, Scorpaenidae, Sparidae, Teraponidae, Apogonidae, Tetraodontidae and Ambassidae. Whitfield *et al.* (1989) compared the dominant fish families in South African *Zostera* beds with those in *Zostera* habitats from south-eastern Australia, north-eastern America and southern Japan. The highest overlap occurred with south-eastern Australia, and was attributed to the distribution pattern of tropical Indo-Pacific fish taxa. Data are reproduced here, and compared with the ten dominant families recorded in this study (Table 4.10). Only four families were shared with south-eastern Australia, but they corresponded closely in their orders of importance. Seven of the ten dominant families in Mhlathuze eelgrass were families noted as important in *Zostera* in South African estuaries, but Mugilidae, Atherinidae and Clupeidae were far less dominant than previously reported, possibly because the information used by Whitfield *et al.* (1989) was predominately drawn from temperate Cape systems.

Table 4.10: Ten numerically dominant fish families associated with eelgrass (*Zostera*) habitats in South Africa and south-eastern Australia. a = data from this study, Mhlathuze Estuary, b = data from various South African systems (Whitfield *et al.* 1989), c = data from south-eastern Australia (references cited in Whitfield *et al.* 1989). Numbers indicate descending order in abundance of individuals.

Family	Mhlathuze Estuary [a]	South Africa [b]	South-eastern Australia [c]
Sparidae	1	4	
Gobiidae	2	5	1
Tetraodontidae	3		3
Ambassidae	4	6	
Lutjanidae	5	10	
Mugilidae	6	1	9
Teraponidae	7	7	8
Monodactylidae	9	8	
Labridae	10		
Syngnathidae	15		2
Scorpaenidae	18		7
Sillaginidae	20		6
Clupeidae	34	3	10
Atherinidae	35	2	4
Clinidae	40	9	5

Data reported by Whitfield *et al.* (1989) from Richards Bay (pre-harbour development) closely matched those from the Mhlathuze in terms of abundant species, although *R. holubi* dominated the assemblage to a far greater degree during the more recent survey. Cluster and ordination analysis comparing fishes associated with South African *Zostera* habitats, indicated that eelgrass in Richards Bay supported an assemblage distinct from those in eelgrasses further south in the subtropics, and in warm- and cool-temperate systems in the Cape (Whitfield *et al.* 1989). This is likely to have been due to zoogeographical factors and places even more emphasis on the importance of *Zostera* in Mhlathuze Estuary as the larger of only two areas of eelgrass remaining in KwaZulu-Natal.

Mangroves

Mangrove habitats have also been widely considered to fulfil an important role as nursery to diverse fish communities, in some areas even more so than seagrass habitats. In tropical Australia, Robertson and Duke (1987) found mangroves to sustain much higher fish densities than nearby seagrasses (although seagrasses sampled were short and sparse and therefore provided poor cover for fishes). Laegdsgaard and Johnson (1995) concluded that while seagrass habitats in subtropical Australia do act as nursery habitats for fishes, the importance of this role is reduced when compared with the gross abundances of juveniles and the number of economically important species found in mangrove habitats. In contrast, Pinto and Punchihewa (1996) found a higher number of species, individuals and diversity in seagrasses than mangroves in a Sri Lankan estuary. In Mhlathuze Estuary, mangrove fringes supported fewer species, lower species richness and lower diversity than eelgrass, but fish densities did not differ significantly between these two habitats. Community attributes were similar amongst mangrove fish assemblages and those from unvegetated habitats. A similar situation was noted in Kenya where catch rate, species richness, diversity and evenness of a mangrove creek fish assemblage were similar to that of the assemblage from a sandy beach at the system's mouth (Little *et al.* 1988).

Blaber *et al.* (1985) reported differences in species composition of fish communities in mangroves and open shore habitats in Australia, and noted that these were probably due to habitat structure since salinities, temperatures and turbidities were similar. Fish assemblages associated with mangrove fringes in Mhlathuze Estuary were different from those elsewhere in the system, although species typical of this habitat (*Ambassis* spp. and *O. keiensis*) were also found in abundance in other areas. Actual densities of *O. keiensis* were higher in the *Zostera* than at mangrove fringes. However, ambassids typically occurred in double the abundance in the mangroves than any other habitat. Mangrove habitats are often dominated by families of small-sized fishes (Blaber *et al.* 1985, Kimani *et al.* 1996). Important groups in the Mhlathuze Estuary were similar to those found in other studies. *Ambassis* spp. (Pinto 1983, Robertson & Duke, 1990, Rönnbäck *et al.* 1999) and gobies (Little *et al.* 1988, Blaber & Milton 1990, Blaber 1997) dominate the fish assemblages of mangrove habitats in many soft-substratum tropical and subtropical estuaries. These small-bodied fishes utilised mangrove habitat as juveniles and adults in the Mhlathuze. Hypotheses proposed to explain the value of mangroves as nurseries are similar to those given for seagrasses (Bell *et al.* 1984, Blaber 1997, Rönnbäck *et al.* 1999, Nagelkerken *et al.* 2000). As with the *Zostera*, those probably most pertinent to the mangroves in Mhlathuze Estuary are based on structural complexity of the habitat in providing shelter against predators and the abundance of food organisms for juvenile fishes.

Mangroves were also important for species less abundant than *Ambassis* spp. and *O. keiensis* in the estuary. Over 92% of the dwarf goby *Pandaka silvana* and 65% of the estuarine bream *Acanthopagrus berda* used mangrove fringe habitat. The former occurred as juveniles and adults and is appropriately named given that it only attains 16 mm. The latter is an important angling species and matures at lengths beyond the size range considered here. It was the only sparid that did not occur most abundantly in the *Zostera*. Degradation of South African estuaries has contributed to the depletion of *A. berda* in the last century and the species therefore also has status as an important 'indicator species' (Whitfield 1998).

Mudflats

Differences between mudflat assemblages and those of *Zostera* or mangroves can be explained in terms of habitat structural complexity. Those noted between mud- and sandflat assemblages need to be explained in terms of the nature of substratum (given the similarity in water parameters) and the influence of proximity to the shore. An extreme example of the influence of substratum on fish distribution is that of reef habitat at the mouth of the Kosi system in northern KwaZulu-Natal. Even though salinities in this area can fall well below that of seawater, typically reef-associated species make significant contributions to the fish fauna as early life stages (Harris *et al.* 1995), juveniles and adults (Blaber 1978, Blaber & Cyrus 1981).

Estuaries are generally dominated by soft substrata (Blaber 1997) and reef areas as those described above are unusual. Blaber (1997) summarised the findings of several studies on Embley Estuary in tropical Australia, which indicated that species composition differed significantly between muddy and sandy sections of the system. However, there are few published quantitative comparisons of fishes associated with mud and sand substrata, especially in the size classes considered in this study. Fishes most likely to exhibit preferences based on sediment structure are those with strong benthic habits, such as some gobies which burrow and flatfishes which bury themselves. Investigations of mud- and sandflats as fish habitats have therefore concentrated on these groups, with most research effort invested in flatfishes, which have greater economic importance to fisheries (e.g. Burke *et al.* 1991, Walsh *et al.* 1999). Studies on whole assemblages are more pertinent to discussion here, but these have often been confounded by differences in water parameters amongst habitats. Muddy areas in estuaries are usually confined to brackish upper reaches and the scoured lower reaches of most systems are significantly more marine (e.g. Weinstein 1979, Ross & Epperly 1985).

In Mhlathuze Estuary, where there was no major difference in the physico-chemical nature of the water at areas characterised by muddy and sandy substrata, fish assemblages were similar in terms of numbers of species, densities and measures of community richness, diversity and evenness. However, they differed when full data sets of species occurrences and densities were compared using multivariate methods. Fish assemblages at the two sand sites were strongly dominated by mullet and *Ambassis* spp. whereas mudflats were characterised by a wider range of species. Paradoxically, these comprised two open water engrualid taxa with pelagic habits and two gobies with benthic affinities. Lengths at sexual maturity have been determined for the engraulid species that dominated the complexes in this study (Blaber 1979 cited in Whitfield 1998) and fish considered here were juveniles. Less is known about the biologies of the two goby species in KwaZulu-Natal but their small sizes at the mudflat indicated that most of these were also juveniles. As with the other habitats, the importance of the mudflat was masked by the fact that species which apparently preferred this habitat were not common in terms of overall abundance in the estuary. Those occurring at densities which indicated more than chance occurrence included *Pomadasys* *olivaceum*, *Pomadasys kaakan*, Sciaenidae sp., *Solea bleekeri* and *Leiognathus equula*.

Sandbanks

Changes in community indices with increasing distance from the estuary mouth have been shown in a number of habitat types from other studies, but generally over spatial scales larger than that considered here in Mhlathuze Estuary. In most cases such studies have involved axial systems or habitats sited across gradients in physico-chemical variables such as salinity or turbidity. Understandably one, or a set, of these variables has most often been cited as influencing community parameters found to change (Blaber *et al.* 1989, Loneragan *et al.* 1989, Loneragan & Potter 1990, Cyrus & Blaber 1992, Winemiller & Leslie 1992, Sheaves 1998, Kupschus & Tremain 2001). Fish assemblages from sandy areas in the Mhlathuze, with no differences in water variables, did not differ with respect to any community diversity index. The two sandbanks were the only areas that were initially regarded as potentially different habitats but did not support distinct juvenile and small fish assemblages. It is probable that the proximity of the sandbanks to one another precluded any potential influence that distance from mouth may have had on fish assemblage via, for example, availability of recruits.

Mugilidae are one of the most characteristic and numerous families of fishes in tropical and subtropical estuaries (Blaber 1997). Although not restricted in their distribution to sandy substrata, juveniles do avoid deepwater areas. This is probably related to predator avoidance as shallow water habitats offer juvenile mullet protection from piscivorous fish (Blaber 1987). Sediment grain size is also a factor in the distribution of these fishes. Blaber (1977) and Blaber and Whitfield (1977) investigated the feeding ecology of juvenile mullet in estuaries and recorded ontogenetic shifts in feeding habits after recruitment which culminated in an iliophagous diet by 50 mm SL. From these studies it emerged that species distributions are influenced by specific foraging preferences for substrata of different particle sizes. The abundance of mullet over sandbanks in the Mhlathuze was therefore likely to have been influenced by the littoral nature of these sites, as well as the nature of the substratum. This is corroborated by the fact that the subtidal mudflat and *Zostera* both supported insignificant densities of mullet in comparison with littoral sites (sandbanks and mangrove fringes). Depth at littoral sites always varied from a maximum where the haul began to a minimum at the shoreline, while at other sites it was largely constant across the length of the haul. Different depths in less than 1 m of water have been shown to influence the distribution of estuarine fishes in unvegetated habitats (Ruiz *et al.* 1993) and a similar influence cannot be discounted here.

Species other than mullet, which also typified sandbanks in Mhlathuze Estuary, may have also favoured the habitat for its littoral, rather than sandy nature. Other abundant fishes were *Ambassis* spp., which were also abundant at littoral mangrove sites. Species that made minor contributions to the fish densities on sandbanks, but nevertheless indicated a strong preference for this habitat rather than others in the estuary were *Terapon jarbua*, *Favonigobius reichei*, *Chanos chanos*, *Sillago sihama*, *Scomberoides* sp., *Chelonodon laticeps* and *Gerres* spp. Several of these are important angling species. *Rhabdosargus holubi* were also noted as being abundant over estuarine sandbanks, but at lengths greater than 50 mm. The relative paucity of *R. holubi* in these larger size classes in the *Zostera* suggests that the species undergoes a shift in habitat requirements and migrates from eelgrass to open sand habitat with growth. Such ontogenetic shifts in habitat preference have been observed in King George whiting *Sillaginodes punctata* in Australia, and also involved a change from vegetated to unvegetated habitat (Jenkins & Wheatley 1998).

As noted above, many studies have recorded more species and higher fish abundances in seagrasses than over open sand substrata, and emphasised the importance of seagrasses as nursery habitat. However, other studies have recorded a relative abundance of some species over bare sand compared with seagrass beds (Ferrel & Bell 1991, Gray *et al.* 1996). Moreover, Gray *et al.* (1998) found that sand habitat was used by several fish species at night that were not observed during the day. Their opinion that the importance of sand habitat was underestimated is therefore valid and of particular reference to the estuaries of KwaZulu-Natal, which have suffered the results of bad catchment management and sedimentation throughout most of last century.

4.3 Conclusions

Juvenile estuarine-dependent marine species dominate the fish fauna of Mhlathuze Estuary and the system has a primary function as a nursery to these fishes. The system offers at least four distinct habitat types. In the relatively homogeneous water body of the estuarine embayment, these habitats can be described in terms of their structural elements without any necessary refinement with respect to water quality parameters. Eelgrass (*Zostera capensis*), mangrove fringes (*Avicennia marina*), subtidal mudflats and intertidal sandbanks support distinct fish assemblages. All habitats are important as nursery areas. Season does influence these assemblages, via changes in water quality parameters and due to the life cycles of component species, involving reproduction, recruitment, growth, and in the case of most species, emigration back into the marine environment. Possible migration between estuarine habitats with growth is also a factor in at least one species.

This study did not take into account the influences of tidal migrations or diel activities which have been shown to be important factors in habitat utilisation by fishes elsewhere (Laegdsgaard & Johnson 1995, Blaber 1997, Gray *et al.* 1998). This may have influenced the results to some extent but the general agreement with findings from other parts of the world indicated that eelgrass supports the most diverse fish assemblage of all habitats. *Zostera* is relatively abundant in systems further south, but indications are that it does not support fish assemblages similar to those in subtropical KwaZulu-Natal (Whitfield *et al.* 1989). *Zostera* beds in Mhlathuze Estuary are presently probably the best representative stand of subtropical eelgrass in South Africa. Given the sensitivity and rarity of this habitat type, this places significant conservation value on the estuary and this habitat in particular. However, this should not distract from the value of other habitats which, based on spatial distributions of juvenile fishes, were all found to be nurseries preferred by different species. No one habitat can adequately be substituted for another in Mhlathuze Estuary.

While the general findings of this work have been found to concur with those from most other studies, the degree to which some systems have been shown to differ indicated that it is of importance. Much of the literature on the subject originates from research conducted out of South Africa, and within this country similar studies have only been conducted on temperate systems. Moreover, there have been few studies looking at the range of habitats investigated here. Most have concentrated on comparisons between vegetated and unvegetated habitats, or between mangroves and seagrasses. Few have considered differences between whole species assemblages from sandy and muddy substrata. Most importantly, there have been very few studies where physico-chemical water characteristics have been rigorously analysed and shown not to be confounding variables in investigating the influence of habitat structure on fish assemblages in estuaries.

CHAPTER 5

Richards Bay Harbour

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Richards Bay Harbour

5.1 Results

5.1.1 Abiotic characteristics

Over the study period the average depth at sites sampled in Richards Bay Harbour was 0.71 m (SD = 0.19 m). Other physico-chemical variables recorded during monthly sampling trips are listed in Appendix 4 and discussed below.

Mean water temperature was 23.07°C (SD = 2.53°C) with a range of 18.51 to 29.22°C. There were no differences in temperatures amongst sites but predictable differences across seasons were significant (Table 5.1). January, February and March had the highest temperatures and the lowest were recorded in winter, from June to August. (Figure 5.1). Dissolved oxygen ranged from 4.84 to 9.16 mg/l, with a mean of 6.50 mg/l (SD = 1.01 mg/l). Water at the Bhizolo Canal mangroves consistently, and significantly, had lower oxygen content than sites elsewhere in the harbour. Seasonal differences in dissolved oxygen content were also apparent with the highest concentrations recorded in spring. Levels in summer were lower than all other seasons (Table 5.1, Figure 5.1). Mean pH was 8.24 (SD = 0.34) with no spatial differences. Seasonal variations were significant, with the highest pH recorded during winter and spring (Table 5.1). The monthly trend in pH in the harbour was similar to that reported from the adjacent Mhlathuze Estuary during the same study (Figure 5.1).

Salinities in the harbour were mostly marine with a mean of 34.09 ‰ (SD = 2.90 ‰). Two habitats had significantly less saline waters than others sampled. The sandbank near the harbour entrance was influenced by freshwater outflow from the Mzingazi Canal (average salinity 31.60 ‰, minimum 22.90 ‰ recorded September 1997), and the canal mangrove fringe had reduced salinities (average 32.00 ‰, minimum 23.60 ‰ recorded November 1997) due to drainage from the papyrus swamps and the small stream flowing into the Manzinyama Canal. A seasonal influence was also detected with reductions from seawater mostly recorded in spring and summer (Table 5.1, Figure 5.1). Turbidity levels were generally lower in the harbour than the adjacent estuary and ranged from 3 to 29 NTU with a mean of 8.4 NTU (SD = 5.27 NTU). Differences amongst habitats were significant (Table 5.1) with the canal mangroves and the mudflat (13.3 and 11.0 NTU respectively) being more turbid than other habitats. The sandbank near the harbour entrance was the least turbid of all sites (3.9 NTU). Variation with seasons was not significant (Table 5.1).

Table 5.1: Nonparametric analysis of variance (Kruskal-Wallis tests) of physico-chemical variables at different habitats in Richards Bay Harbour. N = total number of replicates, df = degrees of freedom, SNK = Student-Newman-Keuls tests, * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$, ^{ns} = $P > 0.05$, not significant.

Source	Variable	H statistic	SNK, comparison of median and percentiles
Habitat N = 12 df = 5	Temperature	2.685^{ns}	
	Dissolved O_2	12.833^{*}	[canal mangrove < mud, sandbank (basin), sandbank (mouth), mangrove, *Bruguiera* mangrove]
	pH	5.898^{ns}	
	Salinity	22.742^{***}	[canal mangrove < mud, sandbank (basin), mangrove, *Bruguiera* mangrove], [sandbank (mouth) < mud, sandbank (basin), mangrove, *Bruguiera* mangrove], [sandbank (mouth) ≠ canal mangrove]
	Turbidity	34.313^{***}	[canal mangrove > sandbank (basin), sandbank (mouth), mangrove, *Bruguiera* mangrove], [mud > sandbank (basin & mouth), mangrove, *Bruguiera* mangrove], [sandbank (basin) > sandbank (mouth)], [mangrove > sandbank (mouth)], [*Bruguiera* mangrove > sandbank (mouth)]
Season N = 18 df = 3	Temperature	42.457^{***}	[summer > winter, spring, autumn], [autumn > winter, spring], [spring > winter]
	Dissolved O_2	14.799^{**}	[spring > summer, autumn, winter], [winter > summer, autumn], [autumn > summer]
	pH	30.195^{***}	[winter > summer, autumn, spring], [spring > summer, autumn], [autumn > summer]
	Salinity	22.553^{***}	[winter > summer, autumn, spring], [autumn > summer, spring]
	Turbidity	0.772^{ns}	

Thus, with the exception of turbidity, a seasonal influence was significant in all measured abiotic variables in Richards Bay Harbour. There were also differences in selected physico-chemical variables amongst habitats. In most case these involved the canal mangrove habitat, which was more turbid, less saline and had

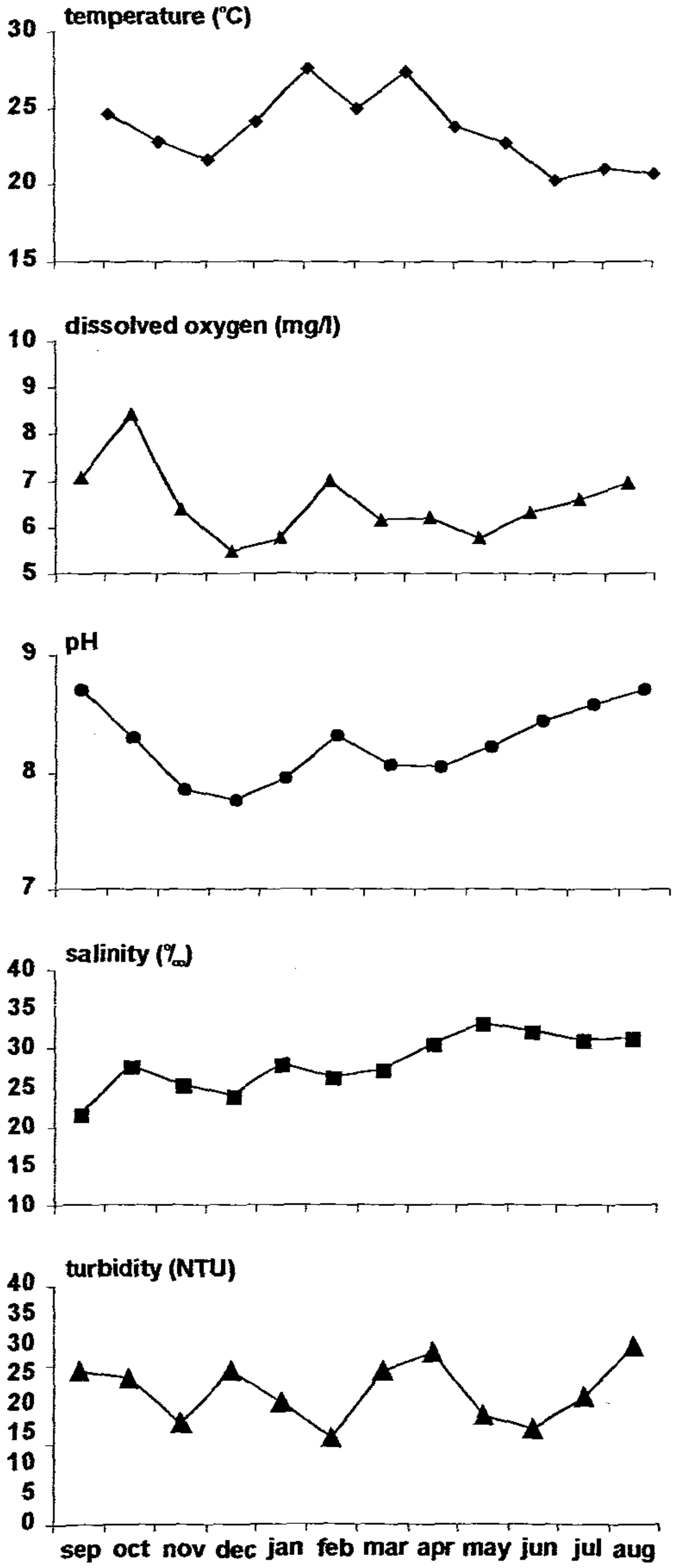

Figure 5.1: Physico-chemical variables recorded on a monthly basis from 1997 to 1998 in Richards Bay Harbour (averaged across habitats).

lower dissolved oxygen concentrations than other sites in the harbour. The sandbank near the entrance was also less saline than other areas and was the least turbid harbour habitat. Lowered salinity at this site seems contradictory given its proximity to the harbour entrance, but was brought about by freshwater discharge from the Mzingazi Canal.

Results of multivariate analysis are shown in the MDS ordinations in Figure 5.2. Samples have been labelled according to habitat (Figure 5.2A) and season (Figure 5.2B). Ordinations of the measured physico-chemical variables indicated distinct groups within habitat and season samples. However, delineation into seasonal groups was clearer in the two-dimensional plot (Figure 5.2B). Analysis of similarity confirmed both habitat and seasonal differences, with a higher R statistic reported for differences amongst seasons (Table 5.2).

Table 5.2: Analysis of similarities of physico-chemical variables recorded from different habitats in Richards Bay Harbour. * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$, ns = $P > 0.05$, not significant.

Test	R statistic	P
Differences amongst habitats		
Global Test	0.184	**
Pairwise Tests		
sandbank (mouth) vs. sandbank (basin)	0.093	ns
sandbank (mouth) vs. mudflat	0.583	**
sandbank (mouth) vs. mangrove	0.204	ns
sandbank (mouth) vs. canal mangrove	0.556	**
sandbank (mouth) vs. *Bruguiera* mangrove	0.176	ns
sandbank (basin) vs. mudflat	0.12	ns
sandbank (basin) vs. mangrove	-0.176	ns
sandbank (basin) vs. canal mangrove	0.259	*
sandbank (basin) vs. *Bruguiera* mangrove	-0.13	ns
mudflat vs. mangrove	0.259	ns
mudflat vs. canal mangrove	0.046	ns
mudflat vs. *Bruguiera* mangrove	0.269	ns
mangrove vs. canal mangrove	0.333	*
mangrove vs. *Bruguiera* mangrove	-0.241	ns
canal mangrove vs. *Bruguiera* mangrove	0.389	**
Differences amongst seasons		
Global Test	0.306	***
Pairwise Tests		
spring vs. summer	0.086	ns
spring vs. autumn	0.167	ns
spring vs. winter	0.309	***
summer vs. autumn	-0.056	ns
summer vs. winter	0.704	***
autumn vs. winter	0.599	***

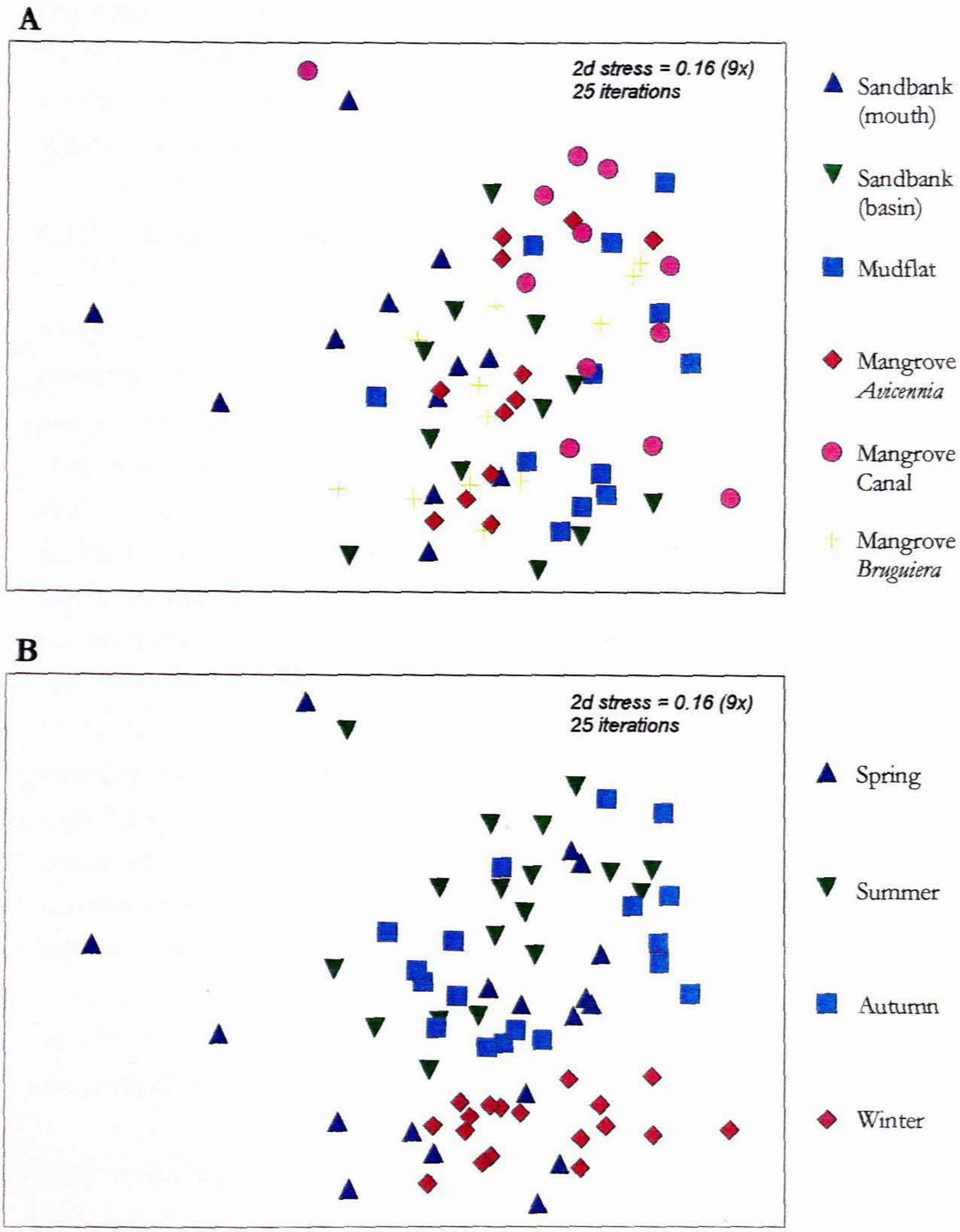

Figure 5.2: MDS ordination of physico-chemical variables (log(x+1) transformed) recorded in Richards Bay Harbour during twelve months (September 1997 - August 1998). Samples labelled by A) habitat and B) season.

The canal mangroves differed from all other habitats, except for the mudflat. Physico-chemical conditions at the mouth sandbank differed from those at the mudflat. Winter measurements differed significantly from spring, summer and autumn (Table 5.2).

5.1.2 Biotic characteristics

A total of 25,874 small and juvenile fishes representing 64 taxa were sampled in Richards Bay Harbour. Species lists and catch data (averaged across all months) are given in Appendix 5. Of the taxa recorded as species complexes, glassies (*Ambassis* spp.) and anchovies (*Stolephorus* spp. and *Thryssa* spp.) were identified to species level when possible. Dominant species in these groups were similar to those at Mhlathuze Estuary. Only two ambassids, *A. gymnocephalus* and *A. natalensis,* were recorded. They comprised 87.7 and 12.3% of specimens identified respectively. *Stolephorus* spp. were dominated by *S. holodon* (78.8% of specimens identified) but *S. indicus* also occurred (12.2% of specimens identified). Of the two *Thryssa* spp. recorded, *T. vitrirostris* was most abundant comprising 96.9% of the specimens identified while *T. setirostris* comprised only 3.1%. Mullet were not identified to species level, but two endemic species *Liza tricuspidens* and *Myxus capensis* were noted. A total of thirteen endemic and six endangered species were collected. All the endangered species were estuarine gobies. Size ranges of different fishes sampled are given in Appendix 5. As was the case in Mhlathuze Estuary, both marine and estuarine spawners were present as recently settled postflexion larvae, 0+ juveniles and adults (in the case of small species). No preflexion larval stages were recorded.

Only seven taxa made individual contributions of over 1% to the total catch in Richards Bay Harbour (standardised to area seined, Table 5.3). *Rhabdosargus holubi*, which dominated Mhlathuze Estuary catches, was not amongst these. Over 50% of the catch comprised of mullet and 30% were Ambassis *spp.* Together with two other taxa, *Sillago sihama* and *Stolephorus* spp., these groups accounted for more than 90% of the fishes caught in the harbour. The *Bruguiera* mangrove fringes supported the highest densities of mullet while *Ambassis* spp. were most abundant at the *Avicennia* mangroves in the Bhizolo Canal. *Sillago sihama* showed a strong preference for sandbanks and *Stolephorus* spp. were most abundant over the mudflat. Over 85% of piggy *Pomadasys olivaceum* were sampled over the hard sand substratum at the *Bruguiera* mangroves. Despite being largely restricted to this habitat, this small marine haemulid was the fifth most abundantly sampled fish in the harbour. Another marine species which is not regarded as estuarine-dependent, the blueline herring *Herklotsichthys quadrimaculatus*, was also abundant at the *Bruguiera* mangroves. Pursemouths *Gerres* spp. dominated catches over the sandbank near the harbour entrance. None of the abundant species in the harbour were recorded in high densities at *Avicennia* mangrove fringes (Table 5.3).

Table 5.3: Fish species contributing >1% to the total standardised catch in Richards Bay Harbour. Percentage contribution from each habitat indicated.

Taxon	total catch		% contribution to taxon total					
	%	Σ %	sandbank (month)	sandbank (basin)	mudflat	mangrove	canal mangrove	*Bruguiera* mangrove
Mugilidae spp.	51.3	51.3	2.5	0.1	0.1	0.1	16.0	81.3
Ambassis spp.	29.9	81.2	2.8	3.5	2.0	5.5	54.9	31.2
Sillago sihama	5.4	86.5	28.8	62.8	0.4	0.0	0.2	7.9
Stolephorus spp.	3.6	90.1	0.0	39.8	57.9	0.3	0.4	1.6
Pomadasys olivaceum	1.7	91.8	3.2	11.2	0.1	0.0	0.0	85.4
Herklotsichthys quadrimaculatus	1.6	93.5	32.9	2.8	5.4	0.0	0.3	58.7
Gerres spp.	1.1	94.6	65.9	19.2	1.2	0.0	0.0	13.8

No freshwater species were sampled in Richards Bay Harbour and less than 2% of individuals were estuarine species (Figure 5.3). Estuarine-dependent marine species dominated fish abundance and comprised 83.7% of the total catch with marine fishes contributing 14.4%. There were strong habitat and seasonal influences on the proportions of fishes with different estuarine affinities (Figure 5.3). Marine species were important components of fish assemblages at the sandbanks and the mudflat, but did not contribute more than 7% to average abundances at any of the three sites fringed with mangroves. Conversely, estuarine–dependent marine species were relatively more abundant at mangrove sites than unvegetated habitats. Estuarine spawning species only accounted for

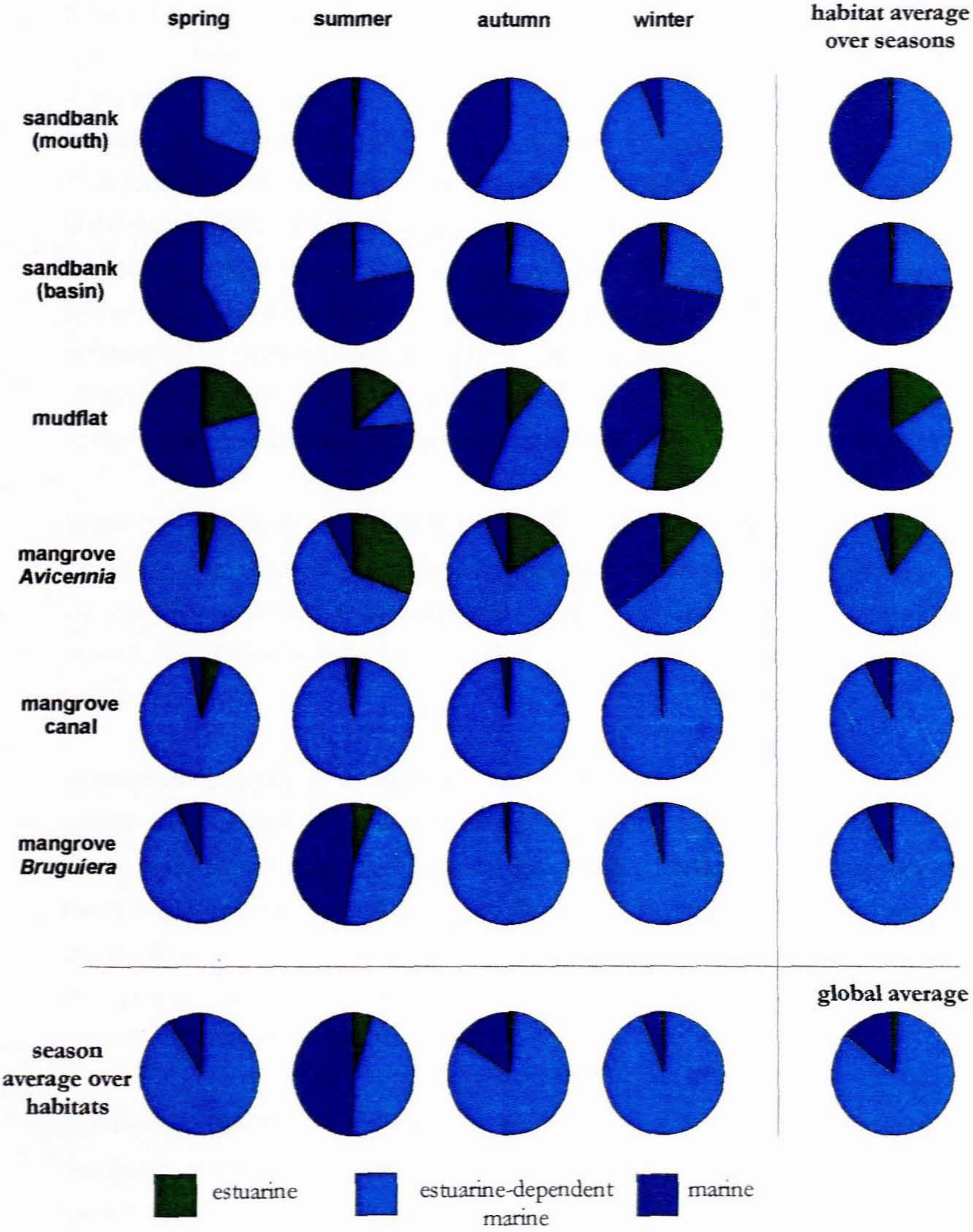

Figure 5.3: Relative contributions of different categories of fishes to different habitats, during different seasons in Richards Bay Harbour.

more than 10% of fish densities at the mudflat and *Avicennia* mangrove fringes in the small, calm water embayment. The low density of fishes sampled at these *Avicennia* mangroves compared with other habitats was influential in this regard. Individuals of estuarine species were actually more abundant at the canal *Avicennia* mangroves than at the embayment *Avicennia* mangroves in spring, autumn and winter, despite percentage contributions of these fishes being lower at the canal mangrove in the latter two seasons. Similarly, higher densities of estuarine fishes were recorded at the *Bruguiera* mangroves than at the *Avicennia* embayment in both summer and winter. Very abundant marine and estuarine-dependent marine species at these sites masked the presence of estuarine species and reduced their relative contribution to all fishes sampled.

Seasonal variation in catch contributions of different categories of fishes was most distinct in the reduced proportions of estuarine-dependent marine species in summer. This was most evident at the *Bruguiera* mangroves, and also influenced by high densities of marine fishes sampled at the sandbanks and mudflat during summer.

Multivariate analysis of similarities amongst different estuarine associated fish categories indicated no consistent pattern amongst species in their distribution across habitats and seasons (Table 5.4, Figure 5.4). This appears contradictory to the findings discussed above based on Figure 5.3. Clearly, the patterns noted in the distribution of marine and estuarine-dependent marine fishes across habitats and seasons were driven by few species, and were not consistent amongst the majority of fishes in either group.

Significant differences were noted in the analysis of fishes according to their distributional ranges. Forty-eight percent of taxa in Richards Bay Harbour were known only from subtropical regions and 42% were species which occur in both subtropical and warm-temperate waters. Less than 8% occur around the South African coastline, including the cool-temperate region. One species caught in the harbour, the white stumpnose *Rhabdosargus globiceps*, has a warm-temperate/

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more than 10% of fish densities at the mudflat and *Avicennia* mangrove fringes in the small, calm water embayment. The low density of fishes sampled at these *Avicennia* mangroves compared with other habitats was influential in this regard. Individuals of estuarine species were actually more abundant at the canal *Avicennia* mangroves than at the embayment *Avicennia* mangroves in spring, autumn and winter, despite percentage contributions of these fishes being lower at the canal mangrove in the latter two seasons. Similarly, higher densities of estuarine fishes were recorded at the *Bruguiera* mangroves than at the *Avicennia* embayment in both summer and winter. Very abundant marine and estuarine-dependent marine species at these sites masked the presence of estuarine species and reduced their relative contribution to all fishes sampled.

Seasonal variation in catch contributions of different categories of fishes was most distinct in the reduced proportions of estuarine-dependent marine species in summer. This was most evident at the *Bruguiera* mangroves, and also influenced by high densities of marine fishes sampled at the sandbanks and mudflat during summer.

Multivariate analysis of similarities amongst different estuarine associated fish categories indicated no consistent pattern amongst species in their distribution across habitats and seasons (Table 5.4, Figure 5.4). This appears contradictory to the findings discussed above based on Figure 5.3. Clearly, the patterns noted in the distribution of marine and estuarine-dependent marine fishes across habitats and seasons were driven by few species, and were not consistent amongst the majority of fishes in either group.

Significant differences were noted in the analysis of fishes according to their distributional ranges. Forty-eight percent of taxa in Richards Bay Harbour were known only from subtropical regions and 42% were species which occur in both subtropical and warm-temperate waters. Less than 8% occur around the South African coastline, including the cool-temperate region. One species caught in the harbour, the white stumpnose *Rhabdosargus globiceps*, has a warm-temperate/

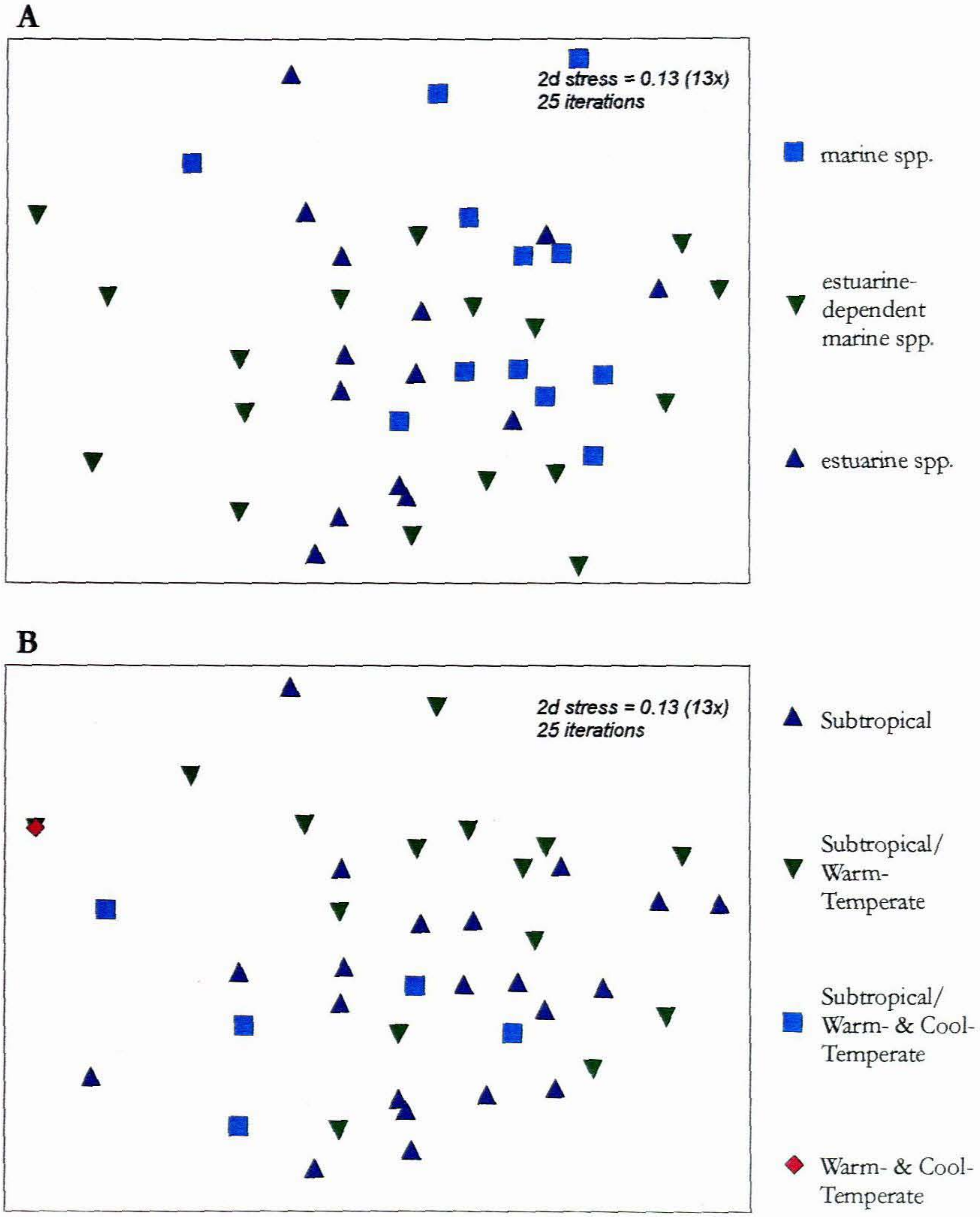


Figure 5.4: MDS ordination of species similarities amongst fishes from habitats in Richards Bay Harbour during twelve months (September 1997 - August 1998). Samples labelled by A) estuarine association and B) geographical range classes.

cool-temperate distribution and is rarely recorded in subtropical waters. Comparisons involving this single species (wc in Table 5.4) should be ignored. Groups indicated as significantly different from one another tended to occur in distinct samples, although the low R statistics (Table 5.4) suggest that this tendency was not marked. Groups with the closest zoogeographical affinities occurred more commonly in the same samples than groups with distinct geographical ranges.

Table 5.4: Analysis of similarities amongst different categories of fishes sampled from different habitats in Richards Bay Harbour. * = P <0.05, ** = P <0.01, *** = P <0.001, ns = P >0.05, not significant, s = subtropical, w = warm-temperate, c = cool-temperate.

Test	R statistic	P
Differences amongst estuarine association classes		
Global Test	0.021	ns
Pairwise Tests		
estuarine vs. estuarine-dependent marine	0.064	ns
estuarine vs. marine	0.040	ns
estuarine-dependent marine vs. marine	-0.020	ns
Differences amongst geographical range classes		
Global Test	0.147	**
Pairwise Tests		
s vs. sw	0.097	*
s vs. swc	0.149	ns
s vs. wc	0.226	ns
sw vs. swc	0.209	*
swc vs. wc	0.200	ns
sw vs. wc	0.451	*

Results of ANOVA of univariate community indices are given in Table 5.5. Means and 95% confidence limits of these indices, averaged between habitats and across seasonal months, are shown in Figure 5.5. Means and confidence limits were back-transformed for presentation of indices that had been transformed to conform to the assumptions of ANOVA. None of these parameters differed significantly amongst sandbank or mudflat samples but species numbers, richness and diversity were often higher in open water areas than at mangrove fringes. Although fewer species occurred at the *Bruguiera* and canal mangroves than sandbank or mudflat habitats, they were present in very high densities. This had predictable ramifications for diversity and evenness indices. Shannon-Wiener diversity at these two sites was significantly lower than at other habitats in several of the

S = Number spp. **N** = Fish Abundance **d** = Spp. Richness **H'** = Spp. Diversity **J'**= Spp. Evenness

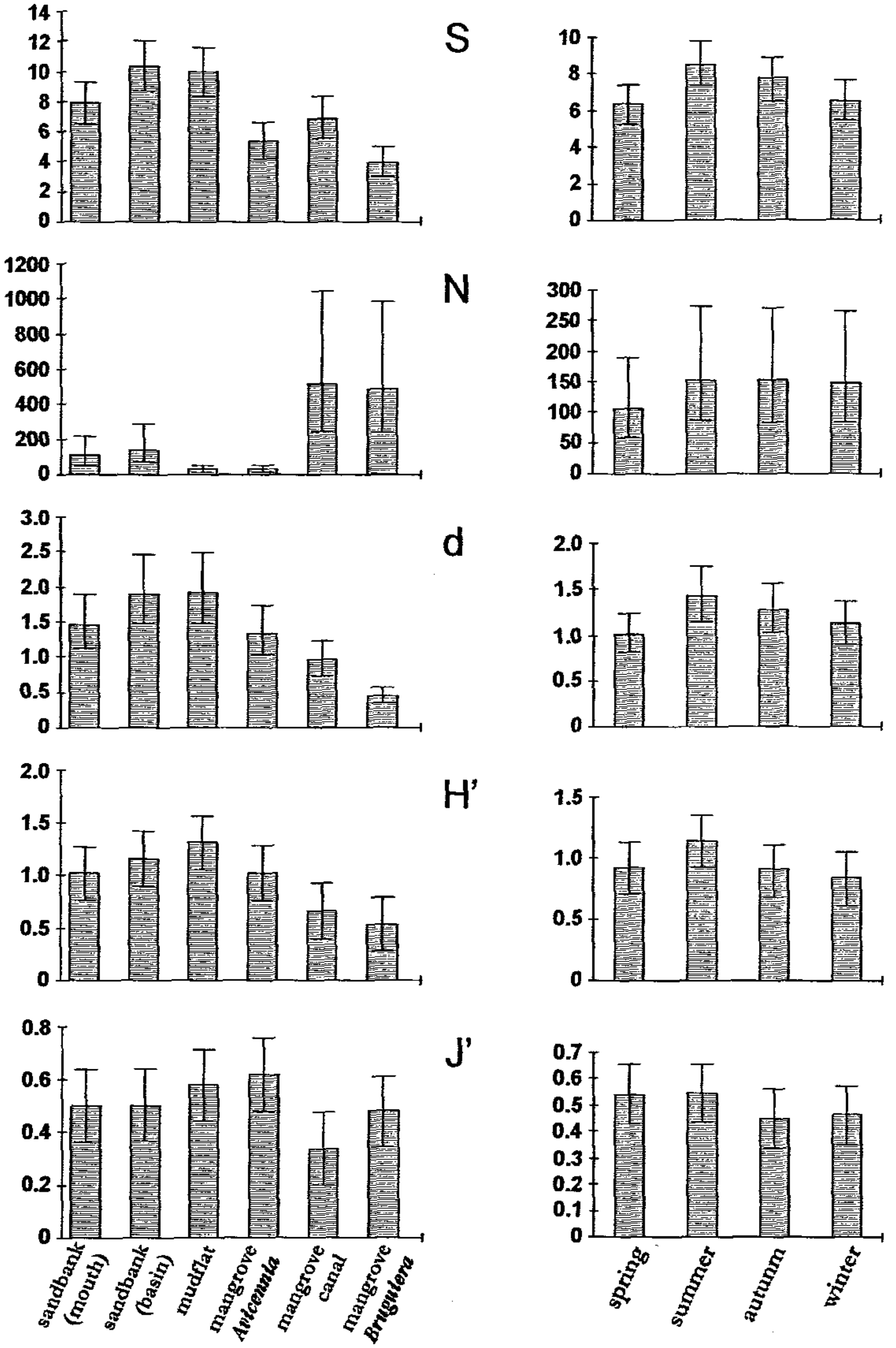


Figure 5.5: Means and 95% confidence limits of fish community indices from assemblages sampled at different habitats and during different seasons in Richards Bay Harbour.

pairwise tests (Table 5.5). The only significant seasonal difference in any index used to characterise the fish communities, was a higher number of species sampled at sites during summer months than winter months. None of the interaction terms were significant (Table 5.5).

Table 5.5: F-ratios and significance levels for ANOVA of selected univariate indices of fish assemblages sampled at Richards Bay Harbour. F-ratios based on the residual mean square error, multiple range analysis based on Tukey's honest significant differences, S = number of species, N = abundance (density), d = Margalef's species richness, H′ = Shannon-Wiener diversity, J′ = Pielou's evenness index, df = degrees of freedom, * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$, ^{ns} = $P > 0.05$, not significant.

Effect	Index	F statistic	Multiple range tests
Habitat (5 df)	S	14.269^{***}	[*Bruguiera* mangrove < sandbank (basin & mouth), mudflat, canal mangrove], [sandbank (basin), mudflat > mangrove, canal mangrove], [mudflat > mangrove]
	N ($\log_{10}$)	10.325^{***}	[*Bruguiera* mangrove, canal mangrove > sandbank (mouth), mudflat], [sandbank (basin > mangrove]
	d ($\log_{10}$)	19.074^{***}	[*Bruguiera* mangrove < sandbank (basin & mouth), mudflat, mangrove, canal mangrove], [canal mangrove < sandbank (basin & mouth), mudflat]
	H′	5.345^{***}	[sandbank (basin), mudflat > *Bruguiera* mangrove,] [mudflat > canal mangrove]
	J′	2.048^{ns}	
Season (3 df)	S	3.159^{*}	[summer > spring]
	N ($\log_{10}$)	0.384^{ns}	
	d ($\log_{10}$)	2.119^{ns}	
	H′	1.564^{ns}	
	J′	0.819^{ns}	
Habitat x Season (15 df)	S	1.240^{ns}	
	N ($\log_{10}$)	1.537^{ns}	
	d ($\log_{10}$)	1.197^{ns}	
	H′	0.928^{ns}	
	J′	0.856^{ns}	

The two-dimensional MDS ordination of similarities amongst root transformed fish densities was highly stressed (0.22). The three-dimensional plot was only moderately stressed (0.15) but its presentation on paper would not have given a clear indication of the relationship amongst samples. For this reason the cluster dendrogram (Figure 5.6) and the two-dimensional MDS ordination are presented (Figure 5.7). Comparison of the two representations is difficult but at approximately 20% similarity four broad groups were distinguished. The ordination plot is presented twice and labelled according to habitat (Figure 5.7A) and season (Figure 5.7B) to illustrate the relative influence of these two factors. That of habitat

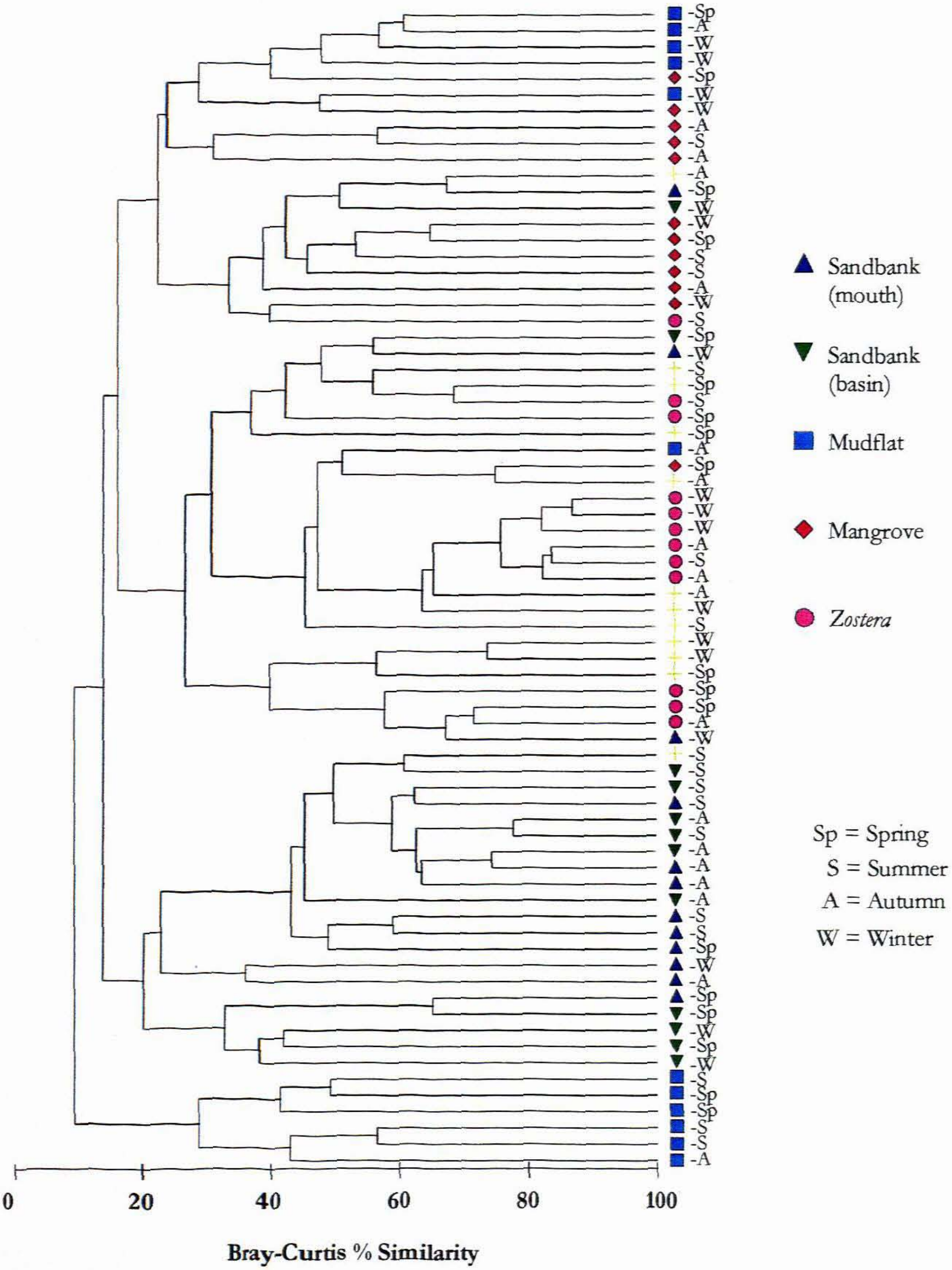


Figure 5.6: Dendrogram of Bray-Curtis similarities in densities of fishes (square root transformed) from habitats sampled during different seasons in Richards Bay Harbour.

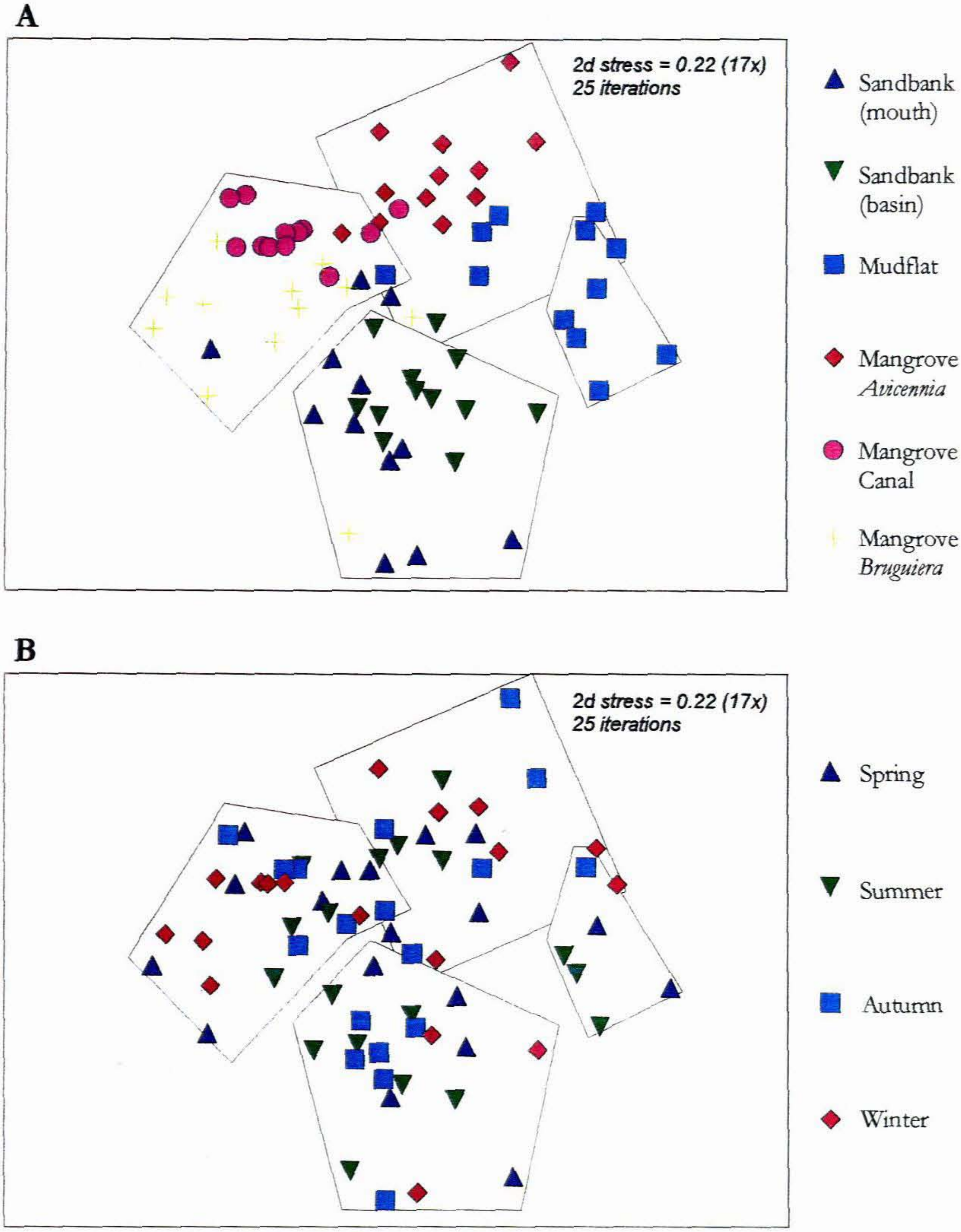


Figure 5.7: MDS ordination of densities of fishes (square root transformed) sampled in Richards Bay Harbour. Samples labelled by A) habitat and B) season. Blocks represent separate groups at 20% Bray-Curtis Similarity.

is quite clear but a seasonal effect is more difficult to discern at a low level of similarity. The dendrogram (Figure 5.6) indicated that the high degree of overlap amongst the sandbank samples was not an artefact of a forced two-dimensional representation of the MDS ordination (Figure 5.7A). These samples did group together in a largely distinct cluster. Mud, mangrove and canal mangrove samples were all distinct. Some overlap was indicated amongst mud and *Avicennia* embayment mangrove samples, and also amongst *Avicennia* canal mangrove and *Bruguiera* mangrove samples. In some cases this appeared to have been influenced by season. For example, mudflat samples from spring and summer were distinct from all other samples, but those from the mudflat in winter shared some similarity with samples collected at mangrove fringes (Figure 5.6). Analysis of similarities was used to check for significance of differences amongst habitats and seasons (Table 5.6).

Table 5.6: Analysis of similarities of fish assemblages sampled from different habitats in Richards Bay Harbour. * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$, ns = $P > 0.05$, not significant.

Test	R statistic	P
Differences amongst habitats		
Global Test	0.576	***
Pairwise Tests		
sandbank (mouth) vs. sandbank (basin)	0.05	ns
sandbank (mouth) vs. mudflat	0.861	***
sandbank (mouth) vs. mangrove	0.625	***
sandbank (mouth) vs. canal mangrove	0.667	***
sandbank (mouth) vs. *Bruguiera* mangrove	0.361	**
sandbank (basin) vs. mudflat	0.815	***
sandbank (basin) vs. mangrove	0.731	***
sandbank (basin) vs. canal mangrove	0.963	***
sandbank (basin) vs. *Bruguiera* mangrove	0.556	***
mudflat vs. mangrove	0.491	**
mudflat vs. canal mangrove	0.796	***
mudflat vs. *Bruguiera* mangrove	0.694	***
mangrove vs. canal mangrove	0.616	**
mangrove vs. *Bruguiera* mangrove	0.556	***
canal mangrove vs. *Bruguiera* mangrove	0.296	*
Differences amongst seasons		
Global Test	0.239	***
Pairwise Tests		
spring vs. summer	0.275	**
spring vs. autumn	0.216	*
spring vs. winter	0.265	*
summer vs. autumn	0.065	ns
summer vs. winter	0.423	***
autumn vs. winter	0.343	**

All habitats were significantly different in the fish assemblages they supported, with the exception of the two harbour sandbanks (Table 5.6). Similar comparisons amongst seasons indicated that only the summer and autumn fish faunas did not differ from one another. Differences in samples taken during other seasons were not apparent in Figure 5.6B. However, ANOSIM indicated that summer, autumn and winter samples differed from those taken during spring months, and winter samples differed from those in summer and autumn (Table 5.6). A higher Global R statistic indicated that habitat was a more influential factor in the distribution of samples, and was the principal determinant of similarities and differences amongst the respective samples.

Average similarity amongst canal mangrove samples was the highest of all habitats and that amongst the *Bruguiera* samples was the lowest (Table 5.7). Eighty percent of the similarity amongst samples within each habitat was generally accounted for by three or fewer taxa. In the mangrove canal and the *Bruguiera* habitats only two taxa (mullet and ambassids) contributed over 95% of the similarity amongst samples (Table 5.7). Higher densities of mullet were recorded at the *Bruguiera* mangrove than any other harbour habitat. *Ambassis* spp. were most abundant at the canal mangroves. The abundance of these fishes, rather than their presence, typified these mangrove assemblages. Although the fish communities at the sandbanks were not significantly different from one another, mullet, *Ambassis* spp. and *Gerres* spp. were important at the sandbank near the harbour entrance. *Sillago sihama*, *Ambassis* spp. and *Stolephorus* spp better typified the basin sandbanks (Table 5.7). The consistent abundance of *Stolephorus* spp. and *Leiognathus equula* at the mudflat distinguished this habitat. *Avicennia* mangrove fringes in the small mangrove embayment were typified by low abundances of *Ambassis* spp. and the goby *Oligolepis keiensis*. Mullet played very little role in accounting for similarity amongst samples taken at this site as they occurred infrequently and in very low densities.

Table 5.7: Average similarity within groups and species contribution >10% similarity. Average abundance as fish per 100 m^2.

Taxon	average abundance	% contribution	cumulative %
sandbank (mouth) average similarity = 16.34			
Mugilidae spp.	46.73	51.40	51.40
Ambassis spp.	25.22	17.90	69.30
Gerres spp.	21.62	12.47	81.77
sandbank (basin) average similarity = 22.39			
Sillago sihama	102.01	52.63	52.63
Ambassis spp.	31.53	23.29	75.92
Stolephorus spp.	43.23	12.32	88.23
mudflat average similarity = 18.86			
Stolephorus spp.	62.91	37.17	37.17
Leiognathus equula	16.98	25.79	62.96
Oligolepis keiensis	7.92	14.74	77.70
mangrove average similarity = 20.70			
Ambassis spp.	49.97	65.05	65.05
Oligolepis keiensis	2.65	11.95	77.01
canal mangrove average similarity = 31.99			
Ambassis spp.	495.36	53.86	53.86
Mugilidae spp.	247.14	42.56	96.42
Bruguiera mangrove average similarity = 14.08			
Mugilidae spp.	1257.92	66.01	66.01
Ambassis spp.	281.70	31.44	97.45

Taxa that played important roles in forging dissimilar fish assemblages at different habitats are given in Table 5.8. *Sillago sihama* consistently distinguished sandbank sites (where they occurred abundantly) from other habitats in the harbour. High abundances of mullet and *Ambassis* spp. contributed to the dissimilarity between canal mangroves and other habitats. This was also true of the *Bruguiera* mangroves. *Stolephorus* spp. played a role in distinguishing mudflat samples from habitats other than canal and *Bruguiera* mangroves where relative differences in abundances of mullet and *Ambassis* spp. were the main reasons for sample dissimilarities. Differences between the *Avicennia* mangrove habitats in the small embayment and the canal were most notable in different abundances of ambassids and mullet (Table 5.8).

Table 5.8: Average dissimilarity between groups and species contributing >10% dissimilarity. Average abundance as fish per 100 m^2.

Taxon	average abundance	average abundance	% contribution	cumulative %
Groups compared ->	sandbank (mouth)	sandbank (basin)	average dissimilarity 81.70	
Sillago sihama	46.73	102.01	31.18	31.18
Ambassis spp.	25.22	31.53	13.76	44.94
Stolephorus spp.	0.00	43.23	13.63	58.56
Mugilidae spp.	38.06	2.12	11.69	70.25
Groups compared ->	sandbank (mouth)	mudflat	average dissimilarity 96.02	
Sillago sihama	46.73	0.66	16.17	16.17
Stolephorus spp.	0.00	62.91	15.22	31.39
Ambassis spp.	25.22	18.09	13.69	45.08
Mugilidae spp.	38.06	1.11	11.66	56.74
Groups compared ->	sandbank (basin)	mudflat	average dissimilarity 90.95	
Sillago sihama	102.01	0.66	26.76	26.76
Stolephorus spp.	43.23	62.91	22.84	49.60
Ambassis spp.	31.53	18.09	13.37	62.96
Groups compared ->	sandbank (mouth)	mangrove	average dissimilarity 92.23	
Ambassis spp.	25.22	49.97	25.14	25.14
Sillago sihama	46.73	0.00	21.76	46.89
Mugilidae spp.	38.06	1.01	14.87	61.76
Groups compared ->	sandbank (basin)	mangrove	average dissimilarity 90.00	
Sillago sihama	102.01	0.00	32.10	32.10
Ambassis spp.	31.53	49.97	21.57	53.67
Stolephorus spp.	43.23	0.30	16.22	69.89
Groups compared ->	mudflat	mangrove	average dissimilarity 88.54	
Ambassis spp.	18.09	49.97	27.84	27.84
Stolephorus spp.	62.91	0.30	24.12	51.97
Leiognathus equula	16.98	2.10	14.69	66.66
Groups compared ->	sandbank (mouth)	canal mangrove	average dissimilarity 90.57	
Ambassis spp.	25.22	495.36	44.60	44.60
Mugilidae spp.	38.06	247.14	32.62	77.22
Groups compared ->	sandbank (basin)	canal mangrove	average dissimilarity 94.23	
Ambassis spp.	31.53	495.36	40.77	40.77
Mugilidae spp.	2.12	247.14	29.72	70.49
Sillago sihama	102.01	0.26	12.69	83.18
Groups compared ->	mudflat	canal mangrove	average dissimilarity 95.34	
Ambassis spp.	18.09	495.36	43.69	43.69
Mugilidae spp.	1.11	247.14	32.98	76.67
Groups compared ->	mangrove	canal mangrove	average dissimilarity 91.14	
Ambassis spp.	49.97	495.36	50.00	50.00
Mugilidae spp.	1.01	247.14	38.44	88.44

Table 5.8 continued...

Table 5.8 continued

Taxon	average abundance	average abundance	% contribution	cumulative %
Groups compared →	sandbank (mouth)	*Bruguiera* mangrove	average dissimilarity 88.94	
Mugilidae spp.	38.06	1257.92	43.43	43.43
Ambassis spp.	25.22	281.70	24.31	67.74
Sillago sihama	46.73	12.85	11.51	79.25
Groups compared →	sandbank (basin)	*Bruguiera* mangrove	average dissimilarity 90.84	
Mugilidae spp.	2.12	1257.92	38.39	38.39
Ambassis spp.	31.53	281.70	21.92	60.31
Sillago sihama	102.01	12.85	16.26	76.57
Groups compared →	mudflat	*Bruguiera* mangrove	average dissimilarity 96.26	
Mugilidae spp.	1.11	1257.92	39.57	39.57
Ambassis spp.	18.09	281.70	24.18	63.75
Groups compared →	mangrove	*Bruguiera* mangrove	average dissimilarity 92.34	
Mugilidae spp.	1.01	1257.92	45.43	45.43
Ambassis spp.	49.97	281.70	31.49	76.93
Groups compared →	canal mangrove	*Bruguiera* mangrove	average dissimilarity 81.49	
Mugilidae spp.	247.14	1257.92	46.23	46.23
Ambassis spp.	495.36	281.70	40.17	86.40

5.2 Discussion

5.2.1 Abiotic characteristics: the role of physico-chemical properties as environmental factors influencing spatial and temporal distribution of fishes in Richards Bay Harbour

Water temperatures in the harbour displayed a similar trend to those in the estuary and the nearshore marine environment at Richards Bay. A seasonal influence was clear but habitats within the system did not differ in terms of this variable. Dissolved oxygen and pH also displayed seasonal variation and were inversely related to temperature. Although pH did not differ spatially, oxygen concentrations at the canal mangroves were lowest of all areas in the harbour. This may be the result of runoff water from industry or the influence of highly organic sediments in this canal. During eight of the twelve months sampled, oxygen tensions in the canal were below 6 mg/l and in May 1997 they dropped below 5 mg/l. This did not appear to influence the fish community structure at the canal as both the number of species and individuals during May were higher than average.

Efficient exchange of water between the marine and harbour environments precluded the possibility of hypersaline conditions developing. As with the estuary near marine conditions typified the whole embayment. There were influences of freshwater input into the system, which were most notable in seasons of highest rainfall and runoff. These influences were only significant at the outflow areas of drainage canals at the Bhizolo mangroves and the sandbank near the harbour mouth. The spatial variation in turbidity in the harbour can be ascribed to the nature of the sediments with muddy sites more likely to be turbid than sandy ones. Water currents and wind probably also influenced turbidity at some sites more than others. The mudflat and mangrove canal both had muddy substrata that were exposed to wind and current induced turbulence, which entrained fine sediments. The *Avicennia* mangroves, although also on a muddy substratum, were located in a calm water embayment where the influence of tidal currents and wind was reduced.

The canal mangroves and sandbanks near the harbour mouth were distinct habitats in terms of the physical properties of their waters. These differences would have to be taken into account if these habitats were to be defined as different fish habitats. Investigation of the full set of water variables measured indicated that winter conditions in the harbour were significantly different from all other seasons, with little variation amongst spring, summer and autumn. As was the case in the estuary, this suggested that the harbour was actually only subject to the influence of two seasons, with water in the austral spring, summer and autumn being warmer, less saline and closer to neutral pH than in winter months. Confirmation of this would require seasonally replicated data sets over more than a single year.

In summary, Richards Bay Harbour is a marine dominated embayment. This was expected given the deep, wide mouth and limited volumes of freshwater inflow. Freshwater influence was significant in localised areas at the outflow points of two canals that drain into the harbour. This accounted for some spatial heterogeneity in the water from different habitats investigated. While estuarine mixing can lead to sharp gradients in variables (Turner *et al.* 1991) marine environments tend to be more homogeneous (Lusher 1985). The low discharge volumes of these canals, together with the morphology of the embayment and large tidal prism prevented the establishment of gradients across any significant distance. However, it cannot be assumed that the water quality variables measured did not influence the spatial distribution of fishes within the harbour. The influence of seasonal changes in water parameters could also not be discounted.

While the marine characteristics of harbour waters prevented their use by most local freshwater species, it was unlikely to have precluded the occurrence of any of the estuarine and marine spawning fishes that utilise KwaZulu-Natal estuaries on the basis of their osmoregulatory capabilities (Whitfield *et al.* 1981). Temperature was also unlikely to have been a factor for these fishes (Cyrus & Blaber 1987b, Cyrus 1988). As discussed above, relatively low oxygen concentrations at the canal mangroves appeared to have little influence on the fish community. Mass fish moralities attributed to low oxygen have occurred at much lower concentrations than the minimum values measured here (Whitfield 1998, 1999). Thus, while it cannot be completely discounted that dissolved oxygen influenced fish distribution, it was unlikely that this variable was limiting for most of the fishes sampled in the harbour. Although the harbour as a whole is a clear water system with an average turbidity of less than 10 NTU, deviations from this were noted at specific habitats and this might have influenced the species composition at these areas.

5.2.2 Biotic characteristics: occurrence of fishes in structurally different habitats in Richards Bay Harbour

5.2.2.1 The fish fauna and the role of Richards Bay Harbour as a nursery

The physical natures of habitats sampled in the harbour were similar to most of those sampled in the estuary. In these habitats the fine-mesh seine net was effective in sampling fishes in the size range investigated (<5 to 50 mm SL). The list of species recorded was dominated by taxa with subtropical affinities that are known to occur in KwaZulu-Natal estuaries. Several endemic and rare species were recorded. Some forms more typical of shallow marine environments were noted on a fairly regular basis. These included the blueline herring *Herklotsichthys quadrimaculatus* and the sand dragonet *Callionymus marleyi.*

Whilst the distribution of fish densities is characteristically dominated by few species in both embayments (Quinn 1980) and estuaries (Haedrich 1983, Whitfield 1994b) it was particularly skewed in Richards Bay Harbour, mainly due to the high densities of mullet and ambassids. These taxa were not recorded in high densities in a study of larval fishes in the harbour, and contributed less than 0.1% to total catches (Harris & Cyrus 1997). Possible reasons for this have been expounded upon in Chapter 4, and involve size at recruitment and the reproductive biologies of the species concerned. Mullet and *Ambassis* spp. typically dominate KwaZulu-Natal estuaries (Wallace 1975a) and their great abundance in the harbour is a reflection of the estuarine nature of the system, not in terms of water characteristics but rather as a shallow body of water protected from wave action. These are key properties of nursery habitat utilised by juveniles of many estuarine-dependent marine fish (Blaber 1974).

The ichthyofauna of Richards Bay Harbour has been sampled most recently by the Coastal Research Unit of Zululand. Seasonal sampling was undertaken similar to that conducted at Mhlathuze Estuary during 1996 and 1997 (Chapter 4). Results have yet to be published but analysis of the database indicated that estuarine-dependent marine fishes dominated the species composition. A single record of the freshwater cichlid *Oreochromis mossambicus* was noted. This species shows remarkable tolerance to salinity and is common in southern African coastal lakes and temporary open/closed estuaries (Whitfield 1990, 1998). In the harbour, it occurs in large numbers at the head of the Mzingazi Canal (*pers. obs*) and despite its salinity tolerance capabilities, apparently seldom ventures into areas of marine salinities. This concurs with the findings of Whitfield and Blaber (1979) who noted that the species is normally absent from the lower reaches of open estuaries.

Published data from fish surveys of the harbour are limited to sampling conducted in 1991, and these are reported by Cyrus and Forbes (1996). Species sampled were not listed, but the dominance of juveniles of estuarine-dependent marine fishes in terms of species numbers and abundances was highlighted, as was the role of the harbour as a nursery for these fishes. Results from the present study substantiate these findings but also indicate that the harbour is well utilised by typically marine species in their juvenile life stages. *Pomadasys olivaceum* and *H. quadrimaculatus* are marine stragglers in South African estuaries (Whitfield 1998) but were abundant in the system as early juveniles.

Estimates of numerical standing stocks of species (averaged over all months) were gained by multiplying average fish densities by approximate areas of habitat in the harbour (Table 5.9). Sandbanks were considered a single habitat type and species densities were averaged across the two sandy areas sampled. Several assumptions were made regarding densities of fishes from mangrove sites in the harbour. Densities of fishes from canal *Avicennia* fringes were regarded as more representative of *Avicennia* fringes in the main harbour basin than densities recorded in the small embayment mangroves. This assumption is substantiated in the following section, but it remains untested conjecture. *Sillago sihama*, which occurred in high densities over large sandbank areas, were the most abundant fish in the system followed by *Stolephorus* spp., *Ambassis* spp. and mullet (Table 5.9).

Areas of *Avicennia* and *Bruguiera* fringe habitat in Richards Bay Harbour are small compared to sand- and mudflats. However, these mangrove fringes contributed large numbers of ambassids and mullet to the total abundances of these taxa in the harbour (Table 5.9). The high densities of these fishes at mangrove fringes suggest that the relatively small area of this habitat may limit numbers of these species in the harbour.

Table 5.9: Standing stock (total numbers) of the ten most abundant fish taxa (≤50 mm SL) in Richards Bay Harbour, averaged over 12 monthly samples. Total numbers (monthly average) of all small fishes in shallow water habitats indicated. - = <500 fish.

	Numbers of fish (x 10^6)					
Taxon	sandbanks 190 ha	mudflat 125 ha	*Avicennia* embayment 3 ha	*Avicennia* canals & main basin 6.5 ha	*Bruguiera* fringes 0.5 ha	Total 325 ha
Sillago sihama	1.413	0.008	-	-	0.001	1.422
Stolephorus spp.	0.411	0.786	-	-	-	1.197
Ambassis spp.	0.539	0.226	0.015	0.322	0.014	1.116
Mugilidae spp.	0.382	0.014	-	0.161	0.063	0.619
Gerres spp.	0.265	0.005	-	-	-	0.270
Leiognathus equula	-	0.212	0.001	0.004	-	0.217
Herklotsichthys quadrimaculatus	0.167	0.033	-	-	0.001	0.201
Oligolepis keiensis	-	0.099	0.001	-	-	0.100
Thryssa spp.	-	0.097	-	-	-	0.098
Gobiidae sp. 1	-	0.089	-	-	-	0.090
↓	↓	↓	↓	↓	↓	↓
Total	3.516	1.718	0.018	0.499	0.084	5.836

5.2.2.2 The influence of habitat and season in structuring the fish assemblage

Based on the physico-chemical properties of the water in Richards Bay Harbour, a wide range of marine and estuarine fish species could potentially make use of habitats offered in the system. These habitats were not characterised by similar water properties and differences in physico-chemical conditions could not be discounted as potential influences on the spatial distribution of small fishes. Similarly, temporal influences by seasonally changing water properties may have influenced the movement of fishes between the harbour and the marine environment, as well as amongst habitats. Influences other than habitat preferences were expected to confound this, as most species recorded in the harbour were known to have life cycles that involve movement between marine and estuarine environments.

Groups of fishes with different estuarine dependencies showed marked differences in their association with different habitats. The clearest distinction was evident in the fact that estuarine-dependent marine species strongly dominated all mangrove habitats (on average contributing >80% of fish densities) but occurred in much reduced proportions in open water areas (average contribution <35%). This could not be explained in terms of water variables. Only at the canal mangroves was water more characteristically estuarine than at sand and mud sites in the main harbour basin. Water at the sandbank near the harbour entrance was actually more estuarine than that at either the *Avicennia* or *Bruguiera* mangroves. The influence of depth could also not be cited, as both sandbanks and mangrove sites were littoral and nets were hauled to the shoreline. As proportions of estuarine spawning species were similar between open habitats and mangroves these differences originated in the relative abundances of marine and estuarine-dependent marine fishes. They may therefore have been caused by various permutations of:

1. marine species avoiding mangrove habitats
2. estuarine-dependent marine species preferring mangrove habitats
3. estuarine-dependent marine species avoiding open water habitats
4. marine species preferring open water habitats

Actual, rather than relative densities of different groups suggested that all four factors were influential, but that (2) and (3) were probably the main reasons for the observed pattern. Densities of marine fishes from these two broad groups of habitats differed (average 43 per 100 m^2 in mangroves vs. 102 per 100 m^2 at open sites), but much less conspicuously than estuarine-dependent marine fishes (785 fish per 100 m^2 in mangroves vs. 58 per 100 m^2 at open sites). Mullet and ambassids were the dominant taxa at these mangrove sites. Given that differences in measured physico-chemical conditions did not explain their high preference for mangrove fringes, and that these species have wide tolerances for variation in water variables anyway, some other reason must have been responsible. Moreover, in the light of findings at the nearby Mhlathuze Estuary, where mullet especially were most abundant over open sandbanks (Chapter 4), some specific difference between the harbour and estuarine environments was likely to be responsible. Abundant food is one option. However, these fishes rely on different nutritional resources (Whitfield 1998) and marked differences in the ability of similar habitats from the estuary and harbour to meet these different dietary requirements seem unlikely. A greater preference for structurally complex habitat in the harbour as a means of avoiding predators is a plausible alternative, particularly if predation pressure in the harbour was shown to be greater than in the adjacent estuary. Circumstantial evidence suggests that this is the case, at least in the terms of predation pressure from piscivorous fish.

Data from fish surveys conducted in 1996 and 1997 indicate that more piscivorous fishes use the harbour than the estuary (CRUZ unpublished data). Species sampled in the harbour included several kingfish *Caranx* spp., garrick *Lichia amia*, three species of queenfish *Scomberoides* spp., barracuda *Sphyraena jello*, king mackerel *Scomberomorus commerson* and queen mackerel *Scomberomorus plurilineatus*. All these fishes occurred at large sizes and were taken in gill nets. Several were also sampled at shallow sandbanks using large seines. Gill and seine net catches from Mhlathuze Estuary during the same period were comparatively devoid of large piscivores. The harbour is characteristically less turbid than the estuary and differences in the abundance of predatory fishes may be ascribed to the preference of visual predators for clear waters. Blaber *et al.* (1985) also observed an abundance of piscivorous fish species in a mangrove system that receives no runoff, and therefore has clear waters. It is widely held that a major attraction of estuaries as nurseries is that large predatory marine fishes do not frequent them (Blaber 1980, Potter *et al.* 1990). This may not hold true for Richards Bay Harbour. The presence of two large mackerel species in particular, suggests that marine predators can, and do, enter the harbour where they probably forage on smaller (juvenile) fishes. As the harbour is less turbid than the estuary, it offers less protection against visual predators, especially in open water. In the absence of this protection, fishes susceptible to predation may limit their distribution to habitats with high structural complexity, even if this is at the cost of increased competition for food and other resources.

Presumably, the marine species that assumed greater importance in the absence of high densities of mullet and ambassids at sandbanks are better adapted to the predation threat than these estuarine-dependent marine fishes. This is likely, given their stronger affiliation for the marine environment where predation threats of this nature might be more common than in estuaries. *Sillago sihama* occurred abundantly over both sandbanks. In these habitats and in clear water, the coloration and benthic habit of this species afford it greater protection from pelagic predators that would plunder shoals of mullet. If so, it suggests that the distribution of different categories of estuarine associated fishes can vary in their distribution in an estuarine environment depending on predation pressure. In turbid systems with few predators, there may be a reduced tendency to restrict distribution to refugia in the form of structurally complex habitats. This would afford fishes the opportunity to benefit from reduced competition by spreading out over greater areas to feed or, in the case of shoaling species, to forage more widely across such systems.

High densities of marine fishes at sandbanks in summer were due to recruitment of young *S. sihama*. With the reduced number of mullet recorded at the *Bruguiera* and canal mangroves, this accounted for the most noted seasonal variation in contributions of fishes from different estuarine association categories. Mullet and ambassid abundances showed similar trends to those noted in the estuary, with highest densities of the former in spring and winter, and *Ambassis* spp. more abundant in summer and spring. These species therefore alternated in ensuring high utilisation of the system by estuarine-dependent marine species throughout the year. This contributed to the fact that very little seasonal variation was evident in community indices.

Indices characterising fish assemblages from different habitats may have been affected by the distinct marine, or estuarine-dependent marine natures of the communities recorded. Vegetated habitats usually support greater fish species richness and abundance than open water habitats (Laegdsgaard & Johnson 1995 and references therein). Mangrove habitats in the present study generally supported greater numbers of individuals but their assemblages comprised fewer species and were lower in species richness. Fish communities at these habitats were dominated by estuarine-dependent marine species and were typically low in diversity with high abundances contributed by few species. The open water assemblages (sand- and mudflats) were influenced by marine species to a greater degree, and were more diverse with higher evenness in abundances of individuals amongst species. Mullet and ambassids were the two main taxa influencing species richness, diversity and evenness of assemblages sampled in both the harbour and the estuary. Habitats not preferred by either of these taxa were generally characterised by lower fish densities, but high species diversity and evenness indices.

Reasons for significantly lower fish densities at *Avicennia* mangroves in the small embayment compared with those at the Bhizolo Canal are speculative. A single beam trawl sample taken in this embayment during pilot studies for this project yielded no crustaceans (*pers. obs*) although other mangrove sites nearby are productive in terms of prawns and crabs (Forbes *et al.* 1996, Weerts *et al.* in prep.). Computer models generated by the Centre for Scientific and Industrial Research (CSIR 2000) have indicated that this small *Avicennia* embayment may be subject to relatively high deposition of fine coal particles originating from a nearby coal storage facility. Therefore, this site may be impacted by harbour activities. An alternative postulate is related to the calm nature of the embayment that renders it susceptible to deposition of coal fines. Mullet comprised less than 2% of the fishes sampled at these *Avicennia* fringes, which was remarkable given the abundance of this taxon at other mangrove sites in the harbour and estuary. This low abundance could not readily be explained in terms of water quality or a higher predation risk at the mangrove embayment. However, this site did differ from other mangrove habitats in being almost completely enclosed within a stand of trees. Only a series of narrow inlet channels joins this area with the main harbour embayment. Leaf fall in *Avicennia marina* occurs throughout the year (Steinke 1999). Due to the calm nature of this small embayment the substratum was carpeted with mangrove leaf litter which was not washed away by the weak currents. Mullet species ingest large quantities of sediment particles together with food items (Blaber 1977, Blaber & Whitfield 1977) and it has been suggested that the grinding action of a muscular gizzard acts with these particles to aid in the digestive process (Payne 1978). Continual blanketing of the substratum with relatively fresh leaf litter may deny young mullet the opportunity to ingest sediment and, in so doing, impede rates of digestion. This may have rendered the *Avicennia* embayment site unattractive as a nursery habitat for mullet species.

Multivariate analysis indicated that habitat and season influenced species assemblages in Richards Bay Harbour, but as was the case in the estuary, groups of samples clustered most distinctly according to the habitat they were sampled in, rather than the season in which they were caught. Thus, habitat was more influential in forging distinct fish assemblages. Only the two sandbanks were found to support communities that were similar. Other habitats all supported different assemblages although some overlap of samples was indicated across the clusters. This occurred along seasonal lines in some cases. In the harbour, fish assemblages in summer and autumn did not differ, but those from other seasons did. This was also noted in Mhlathuze Estuary. Fish communities in KwaZulu-Natal estuarine environments are likely to be least stable in late winter and spring when peak recruitment occurs. Similarities in assemblages between months during these seasons are therefore likely to be low, especially if juvenile fish components are compared.

As was the case in the estuary, it is difficult to distinguish between the effects of physico-chemical and biological factors in determining the effect of season on the harbour fish fauna. The role of habitat in influencing the spatial distribution of small fishes was illustrated, but cognisance needs to be taken of the fact that habitat structure and water quality potentially both played a role in at least some of the differences amongst habitats. Influences of water characteristics were not consistent across broad groups of fishes, but were likely to have affected individual species that have different requirements and display distinct physiological capacities for different ranges in water variables. Preferences in habitat utilisation were also likely to have been influenced by biological factors, such as the threat of predation, foraging efficiency and inter- and intra-species competition.

5.2.2.3 Small and juvenile fishes associated with different habitats

Five distinct habitat types were identified in the harbour. Fishes at these habitats differed in terms of their constituent species or the abundances in which those species occurred. Similar fish assemblages were associated with sandbanks in the harbour. These areas can be considered as the same habitat type despite the difference in their distances from the system's mouth. Mudflats, *Bruguiera* mangrove fringes and *Avicennia* mangroves at two different locations all supported distinct fish communities. As was the case in the estuary the high abundance of relatively few taxa often masked important contributions of habitats to less abundant species. In most cases these species did not dominate, or characterise the actual habitat that was important to them, and yet they were often almost completely restricted in their distribution in the system to specific habitat types.

Bruguiera mangroves

The *Bruguiera* mangroves supported higher fish densities than any other habitat in Richards Bay Harbour. This was brought about chiefly by very high densities of juvenile mullet which typified this habitat. In Mhlathuze Estuary mullet showed a preference for sandbanks, possibly favouring these areas as foraging grounds. This postulate was confounded by the possible influence of the littoral nature of these habitats (Chapter 4). However, a case has been made above for mullet selecting more structurally complex habitat in Richards Bay Harbour due to higher predation pressure. It was notable that the *Bruguiera* mangrove was the only harbour area that was both structurally complex and had a sandy substratum. For mullet this habitat may have been attractive as a unique area in the harbour where good refuge from predators could be gained without forfeiting favoured forage substratum.

Structural complexity and predator avoidance is frequently cited as a reason for the high value of mangrove habitats as nurseries (Bell *et al.* 1984, Blaber 1997, Rönnbäck *et al.* 1999). Furthermore, turbidity is often cited as contributing to protective isolation in estuarine waters in general (Blaber 1980, Blaber & Blaber 1980, Cyrus & Blaber 1987c). Despite this few published studies from elsewhere make specific reference to differences in mangrove utilisation due to differences in turbidity. Seasonal fluctuations in densities of juvenile fishes in mangroves have been noted, but these have been readily explained in terms of life history cycles and recruitment patterns of species involved (e.g. Robertson & Duke 1990). Robertson and Duke (1987) recorded significant and substantial differences in the numbers of individuals sampled in mud and sand mangroves during an Australian dry season, but not in the wet season. However, these were caused by abundances of anchovies that moved between habitats in large schools, and these authors concluded that water clarity was unlikely to explain differences in fish densities amongst mangrove habitats. Several studies have demonstrated that differences between mangroves and nearby habitats occur in the absence of turbidity gradients (Robertson & Duke 1987, Laegdsgaard & Johnson 1995, Chapter 4) but this does not preclude an influence of turbidity on spatial or temporal scales where such gradients do occur. Blaber *et al.* (1985) found that the effectiveness of mangroves as nurseries in clear, deep, saline waters was reduced as piscivorous fishes could still effectively utilise these areas. There was no difference in densities of small fishes in clear *Bruguiera* and turbid *Avicennia* mangrove areas in Richards Bay Harbour, and both areas were well utilised as nursery habitats. This may have been influenced by the fact that only shallow areas in these stands were sampled during this study, and this may have precluded their utilisation by large piscivorous fishes.

Several species much less abundant than mullet in Richards Bay Harbour showed a high preference for *Bruguiera* mangrove habitat. Over 85% of the standardised catch of *Diplodus sargus capensis* and *P. olivaceum* utilised *Bruguiera* mangrove fringes. Although both these species are sometimes abundant in the lower reaches of warm-temperate South African estuaries (Smith & McKay 1991, Whitfield & Kok 1992 cited in Whitfield 1998) they are more common in the marine environment where juveniles favour inshore reefs (van der Elst 1988, Smale & Buxton 1989 cited in Whitfield 1998). Their preference for marine water in Richards Bay Harbour at a habitat that offers refuge in its structural complexity was therefore not surprising. Over 83% of *Terapon jarbua* also occurred at the *Bruguiera* fringe. In Mhlathuze Estuary 84% of this species' abundance was restricted to the sandbank near the mouth. *Terapon jarbua* is lepidophagous, feeding on the scales of other fishes (Whitfield 1998). The species may co-occur with high densities of mullet for feeding purposes, illustrating a potential influence of one taxon's habitat preference on the spatial distribution of another. Important angling species in the form of the three sparids *R. holubi*, *R. sarba* and *A. berda* all occurred more abundantly at *Bruguiera* fringes than any other harbour habitat. In the absence of *Zostera* this might be a preferred habitat for the stumpnoses (*Rhabdosargus* spp.). In the adjacent estuary *A. berda* also showed an affinity for mangrove fringes, but indications are that in the harbour, which offers different types of mangrove habitats, *Bruguiera* fringes are favoured over those at *Avicennia* stands. This may have been influenced by differences in substratum characteristics and/or water properties rather than mangrove species.

Published comparisons of fishes utilising mangrove areas in the same system, but dominated by different species, are few. In the Philippines, Rönnbäck *et al.* (1999) sampled higher densities of fishes in *Avicennia* than *Rhizophora* microhabitats, but made no comparisons of the actual species assemblages involved. Blaber and Milton (1990) studied estuaries in close proximity to one another in the Solomon Islands and reported different fish communities from *Bruguiera* mangroves on hard sand compared with *Rhizophora* mangroves on soft mud. Gobies, including burrowing species, dominated at the soft substratum systems, while fishes at hard substratum *Bruguiera* habitats were mostly taxa that needed the cover provided by an abundance of fallen mangrove branches and trunks at these sites. In Richards Bay Harbour similar tree debris was scattered over the *Bruguiera* mangrove floor. Blaber and Milton (1990) also noted that several species were apparently equally common at both estuary types and therefore indicated no strong preference for either type of habitat. Mullet or ambassids dominated all mangrove fringes in Richards Bay Harbour and in most cases co-occurred in abundance. However, they occurred in markedly different proportions, with mullet much more numerous at *Bruguiera* mangroves and *Ambassis* spp. dominating *Avicennia* fringes.

Canal *Avicennia* mangroves

The canal mangrove habitat was on average the second most densely populated site in the harbour. Although mullet contributed to this, they were only half as numerous as *Ambassis* spp. Foregoing discussion has expounded upon features of the fish fauna in Richards Bay Harbour mangroves in general, with high densities of fishes explained in terms of predation threat, and the relative abundance of mullet at the *Bruguiera* mangroves possibly related to a preference for sandy substrata. The abundance of *Ambassis* spp. at the *Avicennia* canal mangroves was in keeping with mangrove habitats from many other subtropical and tropical systems (Pinto 1983, Robertson & Duke, 1990, Rönnbäck *et al.* 1999) including Mhlathuze Estuary (Chapter 4). Fishes other than mullet and ambassids were far less abundant comprising less than 4% of the number of individuals recorded. Within this group there were taxa that were largely limited in their distribution in Richards Bay Harbour to *Avicennia* mangrove habitats. Most were gobies and included *Periophthalmus koelreuteri africanus*, *Mugilogobius inhacae*, *Pandaka silvana*, *Oligolepis acutipennis* and *Redigobius balteatops*. Of these only *P. silvana* occurred at the sandy *Bruguiera* mangrove site. These findings concurred with those of Blaber and Milton (1990) in recognising gobies as an important component of the fish assemblage characterising soft substratum mangrove habitats. The common importance of both *Avicennia* habitats to these fishes also illustrated the similarity in their fish assemblages, which was masked by the marked difference in total densities of fish sampled. Fish communities at the two *Avicennia* habitats in the harbour were more distinct in terms of the densities at which they occurred rather than their species compositions. This difference was important in assessing the value of habitats as nursery areas in the greater harbour environment. The canal mangroves were far more important than other *Avicennia* mangroves sampled in the small sub-basin on the harbour's western shores.

The importance of the Bhizolo Canal for macrocrustaceans has been reported (Forbes *et al.* 1996, Weerts *et al.* in prep.) but its significance as a nursery for fishes has not previously been investigated. Perhaps the fact that it is quite clearly a dredged canal has distracted from its true value. A perception exists that this habitat is more 'man-made' than other mangrove areas in the harbour. In most cases this is false. Other *Avicennia* mangroves in the harbour also established only after harbour construction. Those sampled in this study have colonised the banks of the causeway constructed to separate the harbour from the estuary. The Bhizolo Canal mangroves are therefore no less established than most *Avicennia* stands in the harbour. It is also unlikely that the age of these stands is a factor in their ability to act as nurseries. Studies conducted elsewhere in replanted *Rhizophora* stands have indicated that young mangrove habitats act efficiently as nurseries within five or six years (Rönnbäck *et al.* 1999).

The structure and water characteristics of the canal mangrove habitat, with its limited freshwater influence and raised turbidities, were more typical of lower reaches of permanently open KwaZulu-Natal estuaries than any other harbour habitat. These factors all contribute to a typically estuarine fish assemblage. Mangrove fringes along the Bhizolo Canal are an important nursery for high abundances of juvenile estuarine-dependent marine fishes. This area has all the attributes of mangrove habitats that are hypothesised to be reasons for their ability to serve as nursery areas (Rönnbäck *et al.* 1999). The area is a refuge from predators as reduced salinities limit the use of the canal by stenohaline marine predators, and higher turbidity and structural complexity offers protection against euryhaline predators. Food is probably abundant as retention of leaf litter creates a detrital food base. In Richards Bay Harbour cognisance of the importance of this area should be taken when planning future harbour expansions.

Embayment *Avicennia* mangroves

Low fish density was the most notable aspect of the assemblage at *Avicennia* fringes in the small embayment. Mullet were absent from most samples. Reasons for this are speculated upon above. Ambassids comprised over 80% of fishes at this mangrove habitat but they occurred far less abundantly than at the canal mangroves or the *Bruguiera* fringes. Species that did occur in relative abundance at the *Avicennia* sub-basin were those gobies noted above to also occur abundantly at the canal mangroves. On the basis of evidence presented here, there is little reason why this area should enjoy higher conservation status than mangroves which have established at areas that have been modified by canal dredging. Differences between assemblages from these habitats were reflected in abundances of fishes that occurred rather than species unique to either area. Indeed, canal mangrove habitat was far more productive in terms of small fishes than the mangrove embayment, despite the fact that the same *Avicennia* mangrove species was involved.

Mudflats

Fish communities at mudflats in Richards Bay Harbour had a more even distribution of numbers of individuals across species than most other habitats. They differed least, in terms of their community indices, with fish assemblages associated with sandbanks. However, species compositions at these two habitat types differed significantly. *Sillago sihama* and *Stolephorus* spp. were important distinguishing taxa. The former was much more common at sandbanks while *Stolephorus* spp. were more abundant at the mudflat. Other species contributing to the density of mudflat fishes were *Ambassis* spp. and *Leiognathus equula*. In the harbour ambassids occurred much more abundantly at mangrove fringes, and while mudflats were important to *Stolephorus* spp. and *L. equula*, these species also occurred in good abundances at other habitats. However, many other species showed strong preferences for the mudflat, with at least 75% of their total harbour catch concentrated at this habitat. About half of these species were caught in low numbers and their presence at the mudflat rather than any other habitat may have been chance occurrence. Those that were seined more abundantly were *Thryssa* spp., *Solea bleekeri*, *Oxyurichthys* spp., *Silhouettea sibayi*, *Hilsa kelee*, *Oligolepis keiensis* and *Pomadasys kaakan*. All of these fishes occurred as young juveniles and several were forms also noted to associate strongly with the mudflats in the adjacent estuary. Higher turbidity at the mudflats than most other harbour habitats may have been a factor in the distribution of *Thryssa* spp. This taxon was dominated by *Thryssa vitrirostris*, a species with a preference for turbid water (Cyrus & Blaber 1987a). The benthic habits of several other species and consistency in their preference for unvegetated muddy substrata highlighted the importance of mudflats as nursery habitat.

Sandbanks

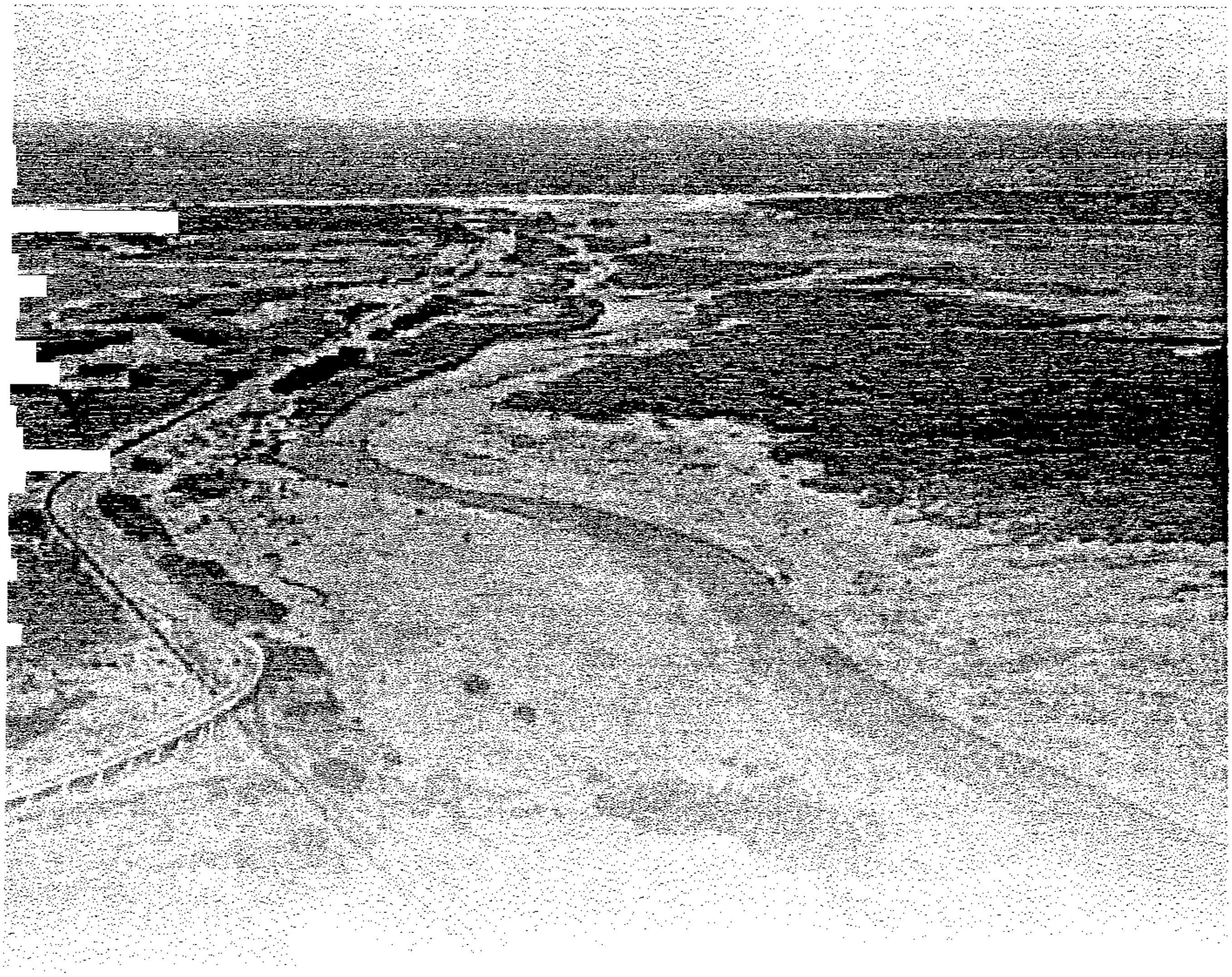
Distance from the mouth did not influence the community structure or species assemblages of sandbank habitats in the harbour. As was the case in Mhlathuze Estuary the two sandbanks were the only areas that were initially regarded as potentially different habitats, but did not support distinct juvenile and small fish assemblages. The proximity of the sandbanks to one another probably precluded any potential influence that distance from mouth may have had on fish assemblage via, for example, availability of recruits. Harbour sandbanks differed in terms of salinity and turbidity but this did not significantly influence the fish faunas. Cyrus and Blaber (1992) investigated the effects of these two variables in a tropical north Australian estuary and concluded that both were important influences on fish distribution. However, their study considered wider ranges in these variables than those recorded between the two sandbanks in Richards Bay Harbour. Comparisons were further confounded by the fact that juveniles of some species have been noted to occupy different turbidity ranges than adults (Blaber & Blaber 1980, Cyrus & Blaber 1992). Clearly the life stages of the fishes subject to investigation needs to be taken into account in comparing results from different studies. In the harbour neither turbidity nor salinity differences between the sandbanks sites had a significant effect on whole fish assemblages. The influence of reduced salinities is much more noted further up the Mzingazi Canal where estuarine gobies such as *Glossogobius callidus* and *Silhouettea sibayi* are abundant (*pers. obs*).

The preference of *S. sihama* for sandbanks in Richards Bay Harbour is attributable to the species' predisposition for clear water systems in KwaZulu-Natal, and the fact that sandy substrata are probably favoured foraging habitats (Weerts *et al.* 1997). This was probably also true of *Gerres* spp. (Cyrus & Blaber 1983, 1987a) which also showed a high preference for unvegetated sandy substrata in the harbour. Other species largely restricted in their harbour distribution to sandbanks were *Pomadasys commersonnii*, *Scomberoides* sp., *Callionymus marleyi*, *Croilia mossambica*, *Atherinomorus lacunosus* and *Favonigobius reichei*. As was the case at mudflats and other habitats investigated, there was considerable overlap in species that predominately occurred at harbour sandbanks and those associated with sandbanks in Mhlathuze Estuary. The littoral nature of this habitat type could not be discounted as a factor in its utilisation by a fish assemblage distinct from that at the mudflats. However, this appeared to be less likely than in the estuary as mullet were less important at harbour sandbanks and the presence of several species could be explained in terms of their known turbidity preferences and foraging habits.

5.3 Conclusions

Fish assemblages associated with different habitats in harbours do not appear to have been well studied despite a world trend of greater environmental concern, and moves towards integration of these concerns into the management and development of harbours (Davis 1990). Large areas of Richards Bay Harbour still exist in a natural form and have regained ecological equilibrium after the large scale changes brought about by harbour construction in the mid-1970's. Several habitat types are found in the system, and in terms of their physical structure they resemble habitats typical of tidal subtropical South African estuaries. Juveniles of a wide range of fish species make use of these habitats. They are dominated in terms of numbers of individuals by estuarine-dependent marine species. This finding supports the conclusions of Cyrus and Forbes (1996) who regarded Richards Bay Harbour as an important nursery for marine fishes that have strong estuarine affinities in KwaZulu-Natal. By considering the role of individual habitats, targeting younger fishes and using a more generalised classification of estuarine dependence, some of the findings of the previous work can be refined. Mangroves in the harbour act as important nurseries for juvenile estuarine-dependent marine species whilst open water sandbanks and mudflats support greater densities of fishes that are not dependent on South African estuaries, although they frequently occur in these environments.

Distinct assemblages of fishes associated with *Bruguiera* mangrove fringes, *Avicennia* mangrove fringes, mudflats and sandbanks in the harbour. The role of habitat structure was more important in affecting the distribution of these fishes than physico-chemical water properties, although the influence of turbidity and salinity could not be discounted in some cases. Fish assemblages were distinguished on the basis of their component species and the densities at which they occurred. The *Avicennia* mangrove fringe in the calm water embayment on the western shore of the harbour was the least important habitat in contributing to the fish assemblage of the system, but this may be due to localised factors. *Avicennia* mangroves in the Bhizolo Canal supported similar species but in much higher numbers. From the results of this study it is not possible to comment on the value of this canal habitat as a fish nursery compared to other *Avicennia* fringes around the main embayment. However, given the abundance of fishes occurring compared with *Avicennia* fringes in the small embayment, and the canal's value as a nursery area for crustaceans, the ecological functioning of this canal warrants further investigation and assessment before it is developed. The *Bruguiera* habitat supported a different fish assemblage than *Avicennia* mangroves. Differences in root structure may have played a role, but these habitats also differed in substratum characteristics and water properties. Therefore the influence of mangrove species *per se* cannot be inferred as the reason for noted differences in fish faunas. Nevertheless, there are probably very few areas in KwaZulu-Natal estuaries that share the same characteristics of mangrove roots providing structural complexity over hard sand substrata in clear marine waters. Several important angling species showed a high preference for this area in the harbour and high densities of important prey species also occurred. Accordingly these mangroves should be given high conservation status in Richards Bay Harbour. The importance of both types of mangroves is presumable due to their value as predation refugia. Deep-water channels and low turbidities render the harbour attractive to large piscivorous fishes. Indications are that the lack of mangroves may limit the role of the harbour as a nursery to estuarine-dependent marine fishes. High densities of fishes utilise these habitats but they cover relatively small surface areas compared with sand- and mudflats. However, open sand and mud substrata were both found to support species that did not occur in significant abundances in mangroves. Indeed, many species showed a high preference for only one habitat and the loss of any habitat type in the harbour would reduce biodiversity and the value of the system as a nursery area.



CHAPTER 6

Comparison of Systems

Chapter 6

Comparison of Systems

Utilisation of shallow water habitats by small and juvenile fishes in Mhlathuze Estuary and Richards Bay Harbour has been investigated in the proceeding sections (Chapters 4 and 5). A comparison of fishes associated with the full set of habitats from both these systems has been made using data from the first month of each season. This comparison is presented in Appendix 6 as a pre-galley draft of a paper accepted for publication in the Proceedings of the 6th Indo-Pacific Fish Conference, and should be read prior to this chapter. Analysis of the full data set has now allowed some refinement of the findings reported, but general conclusions regarding the role of different habitats in both systems and the greater Richards Bay area remain valid. Rather than repeat this work, the present chapter seeks to investigate differences and/or similarities, amongst fishes associated with habitats that were common to both systems. The importance of *Zostera* in the estuary as a nursery for species not found in any other habitats, and its role in structuring the fish assemblage of the estuary as a whole is illustrated in Chapter 4. The fact that it did not occur in the harbour was largely responsible for differences noted in species lists of fishes recorded in the two systems (Appendix 6). *Bruguiera* mangroves occurring in the harbour, but not the estuary, may have had a similar effect. By omitting these sites from a comparison of fishes in both systems the influence of fishes that were not similar because they associated with different habitats was avoided. Many factors important in structuring estuarine fish assemblages could then be assumed to affect Mhlathuze Estuary and Richards Bay Harbour equally. Proximity of their mouths meant that the two systems were potentially available for recruitment by the same stock of larvae and early juveniles. Marked differences between the fish faunas could therefore be investigated without the confounding influence of possible differences in larval supply. Such differences have been found to influence fish assemblages from similar habitats over wide spatial scales (Bell & Westoby 1986).

Four habitats were initially identified on the basis of their structural qualities as occurring in both systems. These were sandbanks in the vicinity of systems' mouths, sandbanks in basin areas, mudflats and *Avicennia* mangrove fringes. These habitat groups were retained even though investigation of each system separately indicated that different sandbanks did not act as different habitats (Chapters 4 and 5). Data from the two sand habitats at each system could not be pooled to give an 'average' sand habitat. This might have been acceptable for fish densities but physico-chemical comparisons also needed to be accounted for. Data from *Avicennia* habitat in the Bhizolo Canal was used in preference to that from the *Avicennia* fringes in the calm embayment. This was because the conspicuously low fish abundances at the *Avicennia* fringe in the embayment could be ascribed to conditions peculiar to that site (Chapter 5). This area was not considered to be representative of the majority of the frontage area of *Avicennia* mangroves in main harbour basin.

6.1 Results

6.1.1 Abiotic characteristics

Physico-chemical conditions in the embayment areas of Mhlathuze Estuary and Richards Bay Harbour have been described in preceding chapters. The present analysis will concentrate on differences between the two systems and common habitats. Mann Whitney Rank Sum Tests indicated that water temperatures and dissolved oxygen concentrations were higher in the estuary than the harbour (Table 6.1). Actual differences in averages of these variables were 1.2°C and 0.5 mg/l respectively. The significant difference in turbidities was far more meaningful, with the estuary habitats averaging 18.0 NTU compared with 9.1 NTU in the harbour. This difference was consistent across habitats, with sandbanks, mudflats and mangroves in the estuary more turbid than the same habitats in the harbour (Table 6.2). At both harbour sandbanks average turbidities were less than 10 NTU but the corresponding estuary habitats had turbidities greater than 10 NTU. Overall, salinity did not differ between systems but there were differences between habitats. These were difficult to comment on as salinities were tested in the significance of their rank orders rather than actual levels. It was apparent that within common habitat types only mudflats and mangroves differed between systems in terms of salinities recorded (Table 6.1). These habitats in the estuary usually had lower salinities than their counterparts in the harbour (Table 6.2). Sandbanks near the mouths of both systems were subjected to freshwater influence and did not differ significantly in their salinities (Table 6.1).

Table 6.1: Nonparametric analysis of variance [Mann Whitney Rank Sum tests (T statistic) and Kruskal-Wallis tests (H statistic)] of physico-chemical variables at different habitats in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH). N = total number of replicates, df = degrees of freedom, SNK = Student-Newman-Keuls tests, * = P <0.05, ** = P <0.01, *** = P <0.001, ^{ns} = P >0.05, not significant. Differences between similar habitats in **bold**.

Source	Variable	T statistic	Comparison of significant differences
System N = 48	Temperature	2599.58*	ME > RBH
	Dissolved O_2	2664.5*	ME > RBH
	pH	2482.0^{ns}	
	Salinity	2207.0^{ns}	
	Turbidity	3007.5***	ME > RBH
Source	Variable	H statistic	SNK, comparison of median and percentiles
Habitat N = 12 df = 7	Temperature	6.816^{ns}	
	Dissolved O_2	19.109**	[RBH mangrove < ME sand (basin), RBH sand (mouth & basin), ME mud]
	pH	7.933^{ns}	
	Salinity	19.367**	SNK tests lack power for firm conclusions but [ME sand (mouth) = RBH sand (mouth)], [ME sand (basin) = RBH sand (basin)], [**ME mud < RBH mud**], [**ME mangrove < RBH mangrove**]
	Turbidity	44.379***	[**ME mud > RBH mud**, RBH sand (mouth & basin), ME sand (basin)], [**ME sand (mouth) > RBH sand (mouth** & basin), RBH mangrove, RBH mud, ME sand (basin)], [**ME sand (basin) > RBH sand (basin)**, RBH sand (mouth), RBH mangrove], [ME **mangrove > RBH mangrove**, RBH sand (mouth & basin), RBH mud, ME sand (basin)], [RBH mangrove > RBH sand (mouth & basin), RBH mud]
Season N = 24 df = 3	Temperature	52.888***	[summer > winter, spring, autumn], [autumn > winter, spring], [spring > winter]
	Dissolved O_2	19.173***	[spring > autumn, summer, winter], [winter > autumn, summer]
	pH	39.535***	[winter > summer, autumn, spring], [spring > summer, autumn]
	Salinity	39.975***	[winter > summer, autumn, spring], [autumn > summer, spring], [spring > summer]
	Turbidity	2.722^{ns}	

Seasonal changes in physico-chemical variables, replicated across the habitats considered in both systems, were similar to those found in individual systems.

Water temperatures differed predictably across seasons being lowest in winter and highest in summer. They reflected sea surface water temperatures measured off the swimming beach at Richards Bay. Dissolved oxygen was generally inversely related to temperature and was significantly higher in spring than other seasons. pH followed the same trend as that noted for individual systems, being highest in summer and lowest in autumn. Seasonal runoff influences were apparent in predictably reducing salinities most markedly in summer, then spring, autumn and winter. There was no seasonal variation in turbidity.

Table 6.2: Averages and ranges of selected physico-chemical water variables sampled at common habitats in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH).

System and habitat	Average (minimum-maximum)	
	Salinity (‰)	Turbidity (NTU)
ME sandbank (mouth)	30.80 (17.7-36.7)	16.83 (11-29)
RBH sandbank (mouth)	31.60 (22.9-36.1)	3.92 (3-5)
ME sandbank (basin)	34.20 (25.9-37.3)	12.67 (2-23)
RBH sandbank (basin)	35.21 (33.4-37.5)	8.33 (3-21)
ME mudflat	27.33 (12.4-36.4)	24.25 (7-72)
RBH mudflat	35.14 (32.8-36.8)	11.0 (7-23)
ME mangrove (*Avicennia*)	28.76 (2.1-36.8)	18.08 (6-37)
RBH mangrove (*Avicennia*)	31.17 (13.6-35.4)	13.25 (7-29)

The relative influences of system, habitat and season on measured physico-chemical variables are evident in Figure 6.1. Whilst the influence of habitat is difficult to detect (Figure 6.1B), differences between systems (Figure 6.1A) and a seasonal effect (Figure 6.1C) are quite clear in the MDS ordination. Analysis of similarities confirmed that all three factors influenced water properties measured although a stronger seasonal influence was suggested by a higher R statistic (Table 6.3).

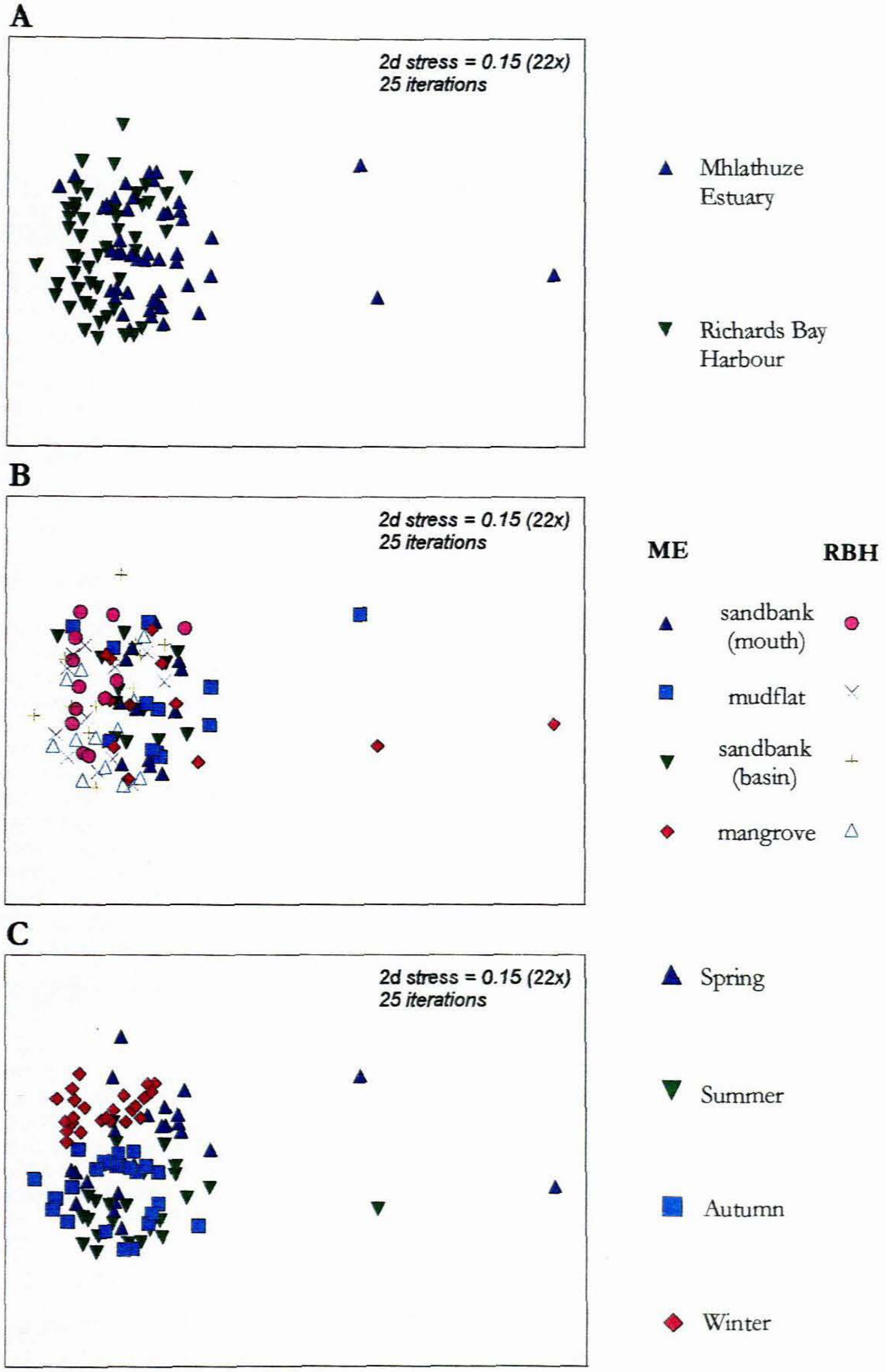


Figure 6.1: MDS ordination of physico-chemical variables (log(x+1) transformed) recorded from common habitats in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH) during twelve months (September 1997 - August 1998). Samples labelled by A) system B) habitat and C) season.

Table 6.3: Analysis of similarities of physico-chemical variables recorded from different habitats in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH). * P = <0.05, ** = P <0.01, *** = P <0.001, ns = P >0.05, not significant. Differences between similar habitats in **bold**.

Test	R statistic	P
Differences between systems		
Global Test	0.120	***
Differences amongst habitats		
Global Test	0.166	***
Pairwise Tests		
ME sandbank (mouth) vs. ME sandbank (basin)	-0.12	ns
ME sandbank (mouth) vs. ME mudflat	-0.148	ns
ME sandbank (mouth) vs. ME mangrove	-0.204	ns
ME sandbank (mouth) vs. RBH sandbank (mouth)	0.667	***
ME sandbank (mouth) vs. RBH sandbank (basin)	0.083	ns
ME sandbank (mouth) vs. RBH mudflat	0.38	*
ME sandbank (mouth) vs. RBH mangrove	0.407	*
ME sandbank (basin) vs. ME mudflat	0.065	ns
ME sandbank (basin) vs. ME mangrove	-0.185	ns
ME sandbank (basin) vs. RBH sandbank (mouth)	0.306	*
ME sandbank (basin) vs. RBH sandbank (basin)	-0.019	ns
ME sandbank (basin) vs. RBH mudflat	0.213	ns
ME sandbank (basin) vs. RBH mangrove	0.37	**
ME mudflat vs. ME mangrove	-0.065	ns
ME mudflat vs. RBH sandbank (mouth)	0.611	***
ME mudflat vs. RBH sandbank (basin)	0.241	*
ME mudflat vs. RBH mudflat	0.278	*
ME mudflat vs. RBH mangrove	0.241	ns
ME mangrove vs. RBH sandbank (mouth)	0.333	**
ME mangrove vs. RBH sandbank (basin)	-0.019	ns
ME mangrove vs. RBH mudflat	0.194	ns
ME mangrove vs. RBH mangrove	0.194	ns
RBH sandbank (mouth) vs. RBH sandbank (basin)	-0.093	ns
RBH sandbank (mouth) vs. RBH mudflat	0.407	*
RBH sandbank (mouth) vs. RBH mangrove	0.546	***
RBH sandbank (basin) vs. RBH mudflat	0.139	ns
RBH sandbank (basin) vs. RBH mangrove	0.194	ns
RBH mudflat vs. RBH mangrove	-0.102	ns
Differences amongst seasons		
Global Test	0.314	***
Pairwise Tests		
spring vs. summer	0.009	ns
spring vs. autumn	0.134	ns
spring vs. winter	0.329	***
summer vs. autumn	-0.065	ns
summer vs. winter	0.782	***
autumn vs. winter	0.648	***

When the suite of measured variables was considered, water sampled in winter was significantly different from that sampled in spring, summer and autumn. There were no other seasonal differences. Mouth sandbanks and mudflats were the only similar habitat types that differed in physico-chemical conditions between systems, although pairwise tests of several other permutations indicated significant differences in water characteristics of habitats. Most of these were between, rather than within systems.

6.1.2 Biotic characteristics

A total of 75 taxa occurred on sandbanks, mudflats and mangrove fringes in Mhlathuze Estuary and Richards Bay Harbour. Sixty-two were reported from the estuary and 61 from the harbour. There was a high degree of overlap in the species assemblages from the two systems. Those species occurring in only one system were never present in abundances over 0.1% of the total catch of that system. On average fishes were more abundant in harbour habitats (318.94 per 100 m^2) than those in the estuary (195.69 per 100 m^2). This was most notably due to higher abundances of *Ambassis* spp. and *Sillago sihama*.

Mullet, *Ambassis* spp., *S. sihama* and *Stolephorus* spp. were the most abundant fishes caught at common habitats in both systems. They were the only taxa that each contributed over 1% of the numbers of individuals in both the estuary and the harbour (Table 6.4). In the estuary eight other taxa also contributed over 1% while only four did in the harbour. Mullet and ambassids dominated fish abundances, contributing approximately two-thirds of all small fishes caught in both systems. In the estuary these two taxa contributed equally to their dominance, but in the harbour ambassids were twice as abundant as mullet. Most mullet and approximately half the ambassids in Mhlathuze Estuary utilised sandbanks, whereas over 85% of both these taxa were concentrated in the mangrove fringes in the harbour. *Sillago sihama* showed a preference for sandbanks in both systems, but contributed a greater proportion to Richards Bay catches. *Stolephorus* spp. were most abundant at mudflats, although they occurred in considerable densities at the harbour basin sandbank as well (Table 6.4).

No freshwater species were sampled in either system. Estuarine fishes comprised 14.70% of the numbers of individuals caught in Mhlathuze Estuary and 7.20% of fishes in the harbour (Figure 6.2). Estuarine-dependent marine species were also proportionally more abundant in the estuary (70.22%) than the harbour (51.31%). However, a greater percentage of marine species was sampled in the harbour (41.50%) than the estuary (15.10%). Similar types of estuarine and estuarine-dependent marine species occurred in the two systems. Both the estuary and the harbour had 20 estuarine spawning species occurring at sandbanks, mudflats and mangroves. Between the systems a total of 23 estuarine fish species was recorded. Eighteen estuarine-dependent marine species were reported from both systems, all of which occurred in the estuary and 16 in the harbour. Differences in the marine components of the fish assemblages were more conspicuous. A combined total of 34 marine species was recorded, ten of these were not sampled in the estuary and nine different species were not sampled in the harbour. None of these taxa contributed large numbers of individuals to catch totals.

Table 6.4: Fish species contributing >1% to the total standardised catch from similar habitats in Mhlathuze Estuary and Richards Bay Harbour. Percentage contribution from each habitat indicated.

Taxon	total catch		% contribution to taxon total			
	%	Σ %	sandbank (month)	sandbank (basin)	mudflat	mangrove
Mhlathuze Estuary						
Mugilidae spp.	33.7	33.7	61.5	24.5	0.4	13.6
Ambassis spp.	32.1	65.8	15.5	29.2	0.2	55.2
Oligolepis keiensis	6.0	71.8	28.2	0.4	30.5	40.9
Stolephorus spp.	5.5	77.4	28.2	9.0	41.4	21.4
Thryssa spp.	3.9	81.2	15.1	2.5	72.8	9.6
Glossogobius callidus	2.7	83.9	17.1	3.8	58.6	20.5
Rhabdosargus holubi	2.6	86.6	27.7	23.5	17.6	31.1
Sillago sihama	2.4	89.0	50.5	45.5	2.6	1.4
Favonigobius reichei	1.5	90.4	63.9	35.8	0.3	0.0
Leiognathus equula	1.1	91.6	30.1	4.7	53.1	12.1
Acanthopagrus berda	1.1	92.7	4.4	2.4	8.7	84.6
Solea bleekeri	1.0	93.7	9.3	0.5	59.6	30.6
Richards Bay Harbour						
Ambassis spp.	44.7	44.7	4.4	5.5	3.2	86.9
Mugilidae spp	22.6	67.3	13.2	0.7	0.4	85.7
Sillago sihama	11.7	79.0	31.2	68.2	0.4	0.2
Stolephorus spp.	8.4	87.4	0.0	40.6	59.0	0.4
Gerres spp.	2.2	89.6	76.4	22.2	1.4	0.0
Leiognathus equula	1.9	91.5	0.0	0.0	71.2	28.8
Herklotsichthys quadrimaculatus	1.6	93.1	79.5	6.8	13.0	0.6

In both systems estuarine species contributed the greatest proportions to the numbers of fishes at mudflats, and generally more so in winter than any other season (Figure 6.2). This appeared to be related to the breeding cycles of two gobies, *Oligolepis keiensis* and *Glossogobius callidus*, which recruited in large numbers as juveniles less than 20 mm SL onto mudflats of both systems in early winter. Mudflats were also consistent across systems in supporting high proportions of marine species. *Stolephorus* spp. were most influential in this regard in spring, summer and autumn, but showed declines in densities in both systems in winter. In the harbour *Leiognathus equula* was abundant in winter and maintained the high proportion of marine species on the mudflat (Figure 6.2).

Estuarine spawning species were more abundant (relative to total fish densities) in the mangrove habitat in the estuary than the harbour (Figure 6.2). This was exaggerated by the much higher abundances (relative and actual) of mullet and *Ambassis* spp. in the harbour mangroves. However, gobies and other estuarine species did occur in higher actual densities in the estuary mangrove than the harbour mangrove. Two exceptions were the small gobiids *Pandaka silvana* and *Mugilogobius inhacae*, which were more abundant in the harbour mangroves than the estuary. Although estuarine-dependent marine species in the form of mullet and *Ambassis* spp. dominated catches at both mangrove areas, they were particularly abundant in the harbour. *Stolephorus* spp. were more abundant in the estuary mangroves throughout the year, and increased the contribution of marine species to this habitat in winter. They occurred in the harbour mangroves only in summer, in low abundances.

The most notable differences in the relative contributions of different groups of fishes to habitat catches were at the sandbanks (Figure 6.2). The main differences between similar sand habitats in different systems were consistent across sandbanks in the main basins and those nearer the respective systems' mouths. Catches from sandbanks in the harbour comprised lower proportions of estuarine-dependent marine species, but higher proportions of marine species, than those in the estuary. These differences were also reflected in actual densities, and taxa

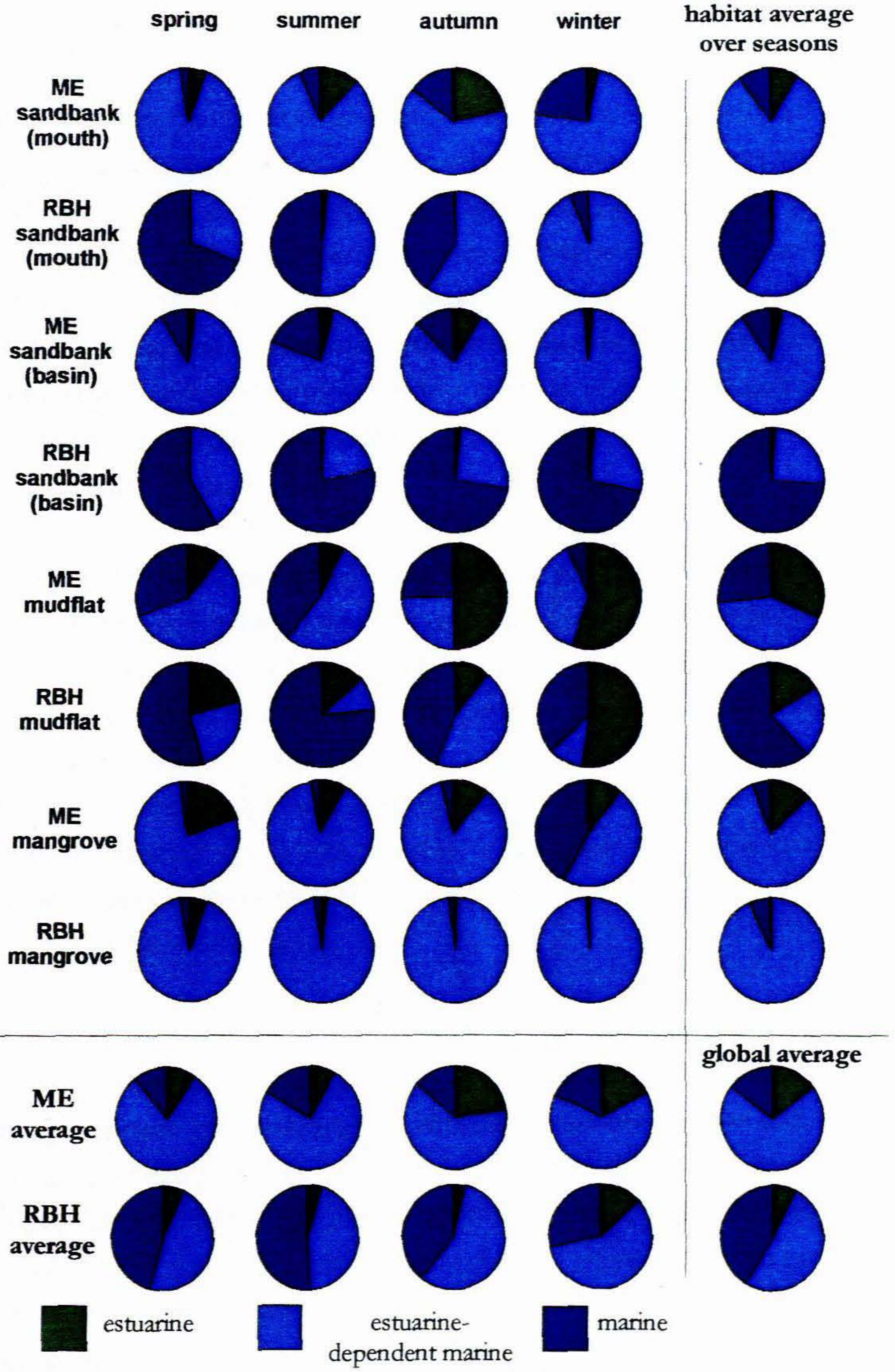


Figure 6.2: Relative contributions of different categories of fishes to common habitats, during different seasons in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH).

chiefly responsible were mullet, ambassids and *S. sihama*. Mullet and ambassids were always recorded in higher densities over estuary sandbanks than harbour sandbanks, and the opposite was true of *S. sihama*. Higher proportions (and densities) of estuarine species over estuary sandbanks were mainly due to three gobies, *O. keiensis*, *G. callidus* and *Favonigobius reichei*. Seasonal trends in the contributions of fishes with different estuarine affiliations were not consistent at sandy habitats, even within systems. In all these areas densities of mullet were highest in spring and winter, and those of ambassids in summer and autumn. *Sillago sihama* recruited to sandbanks in summer and was present in highest abundances in summer and autumn before growing out of the size classes considered for this study.

Community indices are presented in Figure 6.3 and the results of statistical testing given in Table 6.5. For the purposes of investigating differences between the study systems it was necessary to keep similar habitats independent of one another, rather than to regard them as replicated sampling areas of a single habitat type. Therefore, there was no replication of habitats within systems and the ANOVA model did not allow three-way interactions (system x habitat x season) to be investigated. Differences between systems were tested using single factor ANOVA and habitat and season effects were investigated using multiple factor ANOVA as has been performed throughout.

Significantly more species, higher species richness and diversity were reported from Mhlathuze Estuary than Richards Bay Harbour (Table 6.5). Average abundances and the evenness of abundance across species did not differ sufficiently to conclude that the variation in these index values was not due to chance. Estuarine habitats all supported more species than similar habitats in the harbour (Figure 6.3). These differences were only significant in fish assemblages associated with the mouth sandbanks and mangrove fringes (Table 6.5). Fish densities were highest at harbour mangroves but did not differ statistically from the densities reported from estuary mangrove fringes. The domination of the harbour mangrove fish assemblage by only two species was reflected in the lower species

S = Number spp. **N** = Fish Abundance **d** = Spp. Richness **H'** = Spp. Diversity **J'**= Spp. Evenness

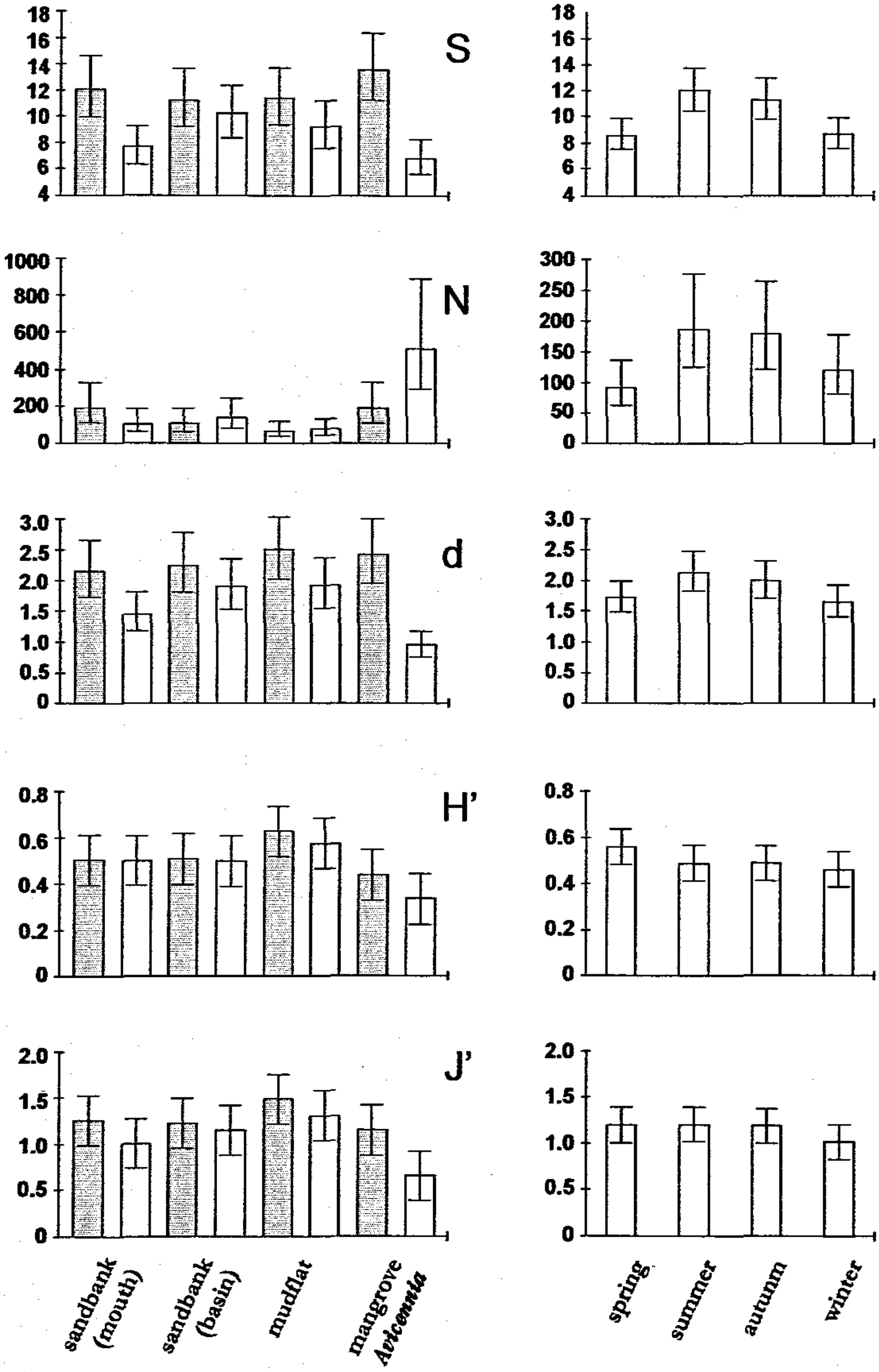


Figure 6.3: Means and 95% confidence limits of fish community indices from assemblages sampled at common habitats during different seasons in Mhlathuze Estuary (shaded) and Richards Bay Harbour (open).

richness compared to the estuary mangroves (Table 6.5). Only numbers of species were found to be influenced by season with more species reported in summer and autumn than winter and spring (Table 6.5, Figure 6.3).

Table 6.5: F-ratios and significance levels for ANOVA of selected univariate indices of fish assemblages sampled in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH). F-ratios based on the residual mean square error, multiple range analysis based on Tukey's honest significant differences, S = number of species, N = abundance (density), d = Margalef's species richness, H' = Shannon-Wiener diversity, J' = Pielou's evenness index, df = degrees of freedom, * = P <0.05, ** = P <0.01, *** = P <0.001, ^{ns} = P >0.05, not significant. Differences between similar habitats in **bold**.

Effect	Index	F statistic	Multiple range tests
System	S (log$_{10}$)	21.465***	ME > RBH
	N (log$_{10}$)	0.600^{ns}	
	d (log$_{10}$)	24.801***	ME > RBH
	H'	6.071*	ME > RBH
	J'	1.010^{ns}	
Habitat (7 df)	S (log$_{10}$)	5.827***	[**ME sand (mouth) > RBH sand (mouth)**, RBH mangrove], [ME sand (basin) > RBH mangrove], [ME mudflat > RBH mangrove], [**ME mangrove > RBH mangrove**, RBH sand (mouth)]
	N (log$_{10}$)	5.480***	[RBH mangrove > RBH sand (basin & mouth), RBH mud, ME mud, ME sand (basin)]
	d (log$_{10}$)	8.817***	[**RBH mangrove < ME mangrove**, ME sand (basin & mouth), ME sand (basin), ME mud, RBH sand (basin), RBH mud], [RBH sand (mouth) < ME mangrove, ME mud]
	H'	3.242**	[RBH mangrove < ME sand (mouth), ME mud, RBH mud]
	J'	2.512*	[RBH mangrove < ME mud]
Season (3 df)	S (log$_{10}$)	6.430***	[spring < summer, autumn], [winter < summer, autumn]
	N (log$_{10}$)	2.901^{ns}	
	d (log$_{10}$)	2.543^{ns}	
	H'	0.931^{ns}	
	J'	1.251^{ns}	
Habitat x Season (21 df)	S (log$_{10}$)	1.223^{ns}	
	N (log$_{10}$)	1.512^{ns}	
	d (log$_{10}$)	1.081^{ns}	
	H'	1.025^{ns}	
	J'	1.124^{ns}	

Multidimensional scaling produced a high stress (0.24) two-dimensional ordination plot and as in previous chapters it was necessary to use both the cluster dendrogram (Figure 6.4) and the ordination (Figure 6.5) to represent the relationships amongst samples. Two main groups of samples emerged in the cluster analysis at a low level of similarity (about 18%, Figure 6.4). On the ordination they can be seen to separate most mudflat samples from those of other habitats (Figure 6.5B). Other habitat groups were distinct although the cluster analysis tended to recognise several separate subsets of samples from each habitat, and the ordination failed to represent true separation of dissimilar samples. Whilst the influences of system (Figure 6.5A) and habitat (Figure 6.5B) were clearly evident in the ordination, that of season was more difficult to discern (Figure 6.5C).

Analysis of similarities confirmed that the influences of all three factors investigated were significant (Table 6.6). Samples were more closely related according to habitat (R = 0.602) than season (R = 0.314). Pairwise tests indicated that the only common habitats that did not support distinct fish assemblages in the estuary and the harbour were the mudflats. While sandbanks within each system did not differ in their associated fish assemblages, they did differ across systems. Analysis of similarities between seasons showed that only summer and autumn fish assemblages did not differ from each other (Table 6.6).

Similarity percentage analysis has been performed on all habitat groups in previous chapters, but for ease of reference average similarities within relevant groups and important contributing species are listed in Table 6.7. Results for analysis of system groups are also given. Ambassids, mullet and *Stolephorus* spp. were important in contributing to similarity amongst samples in the estuary. In the harbour these three species, and *S. sihama* all contributed over 10% to the average similarity amongst samples. Mullet and/or ambassids consistently played an important role in contributing to similarity of fish assemblages at sandbank habitats and mangroves in both Mhlathuze Estuary and Richards Bay Harbour. Neither of these taxa was important in mudflat fish assemblages, which were better typified by *Stolephorus* spp. and *O. keiensis* in both systems.

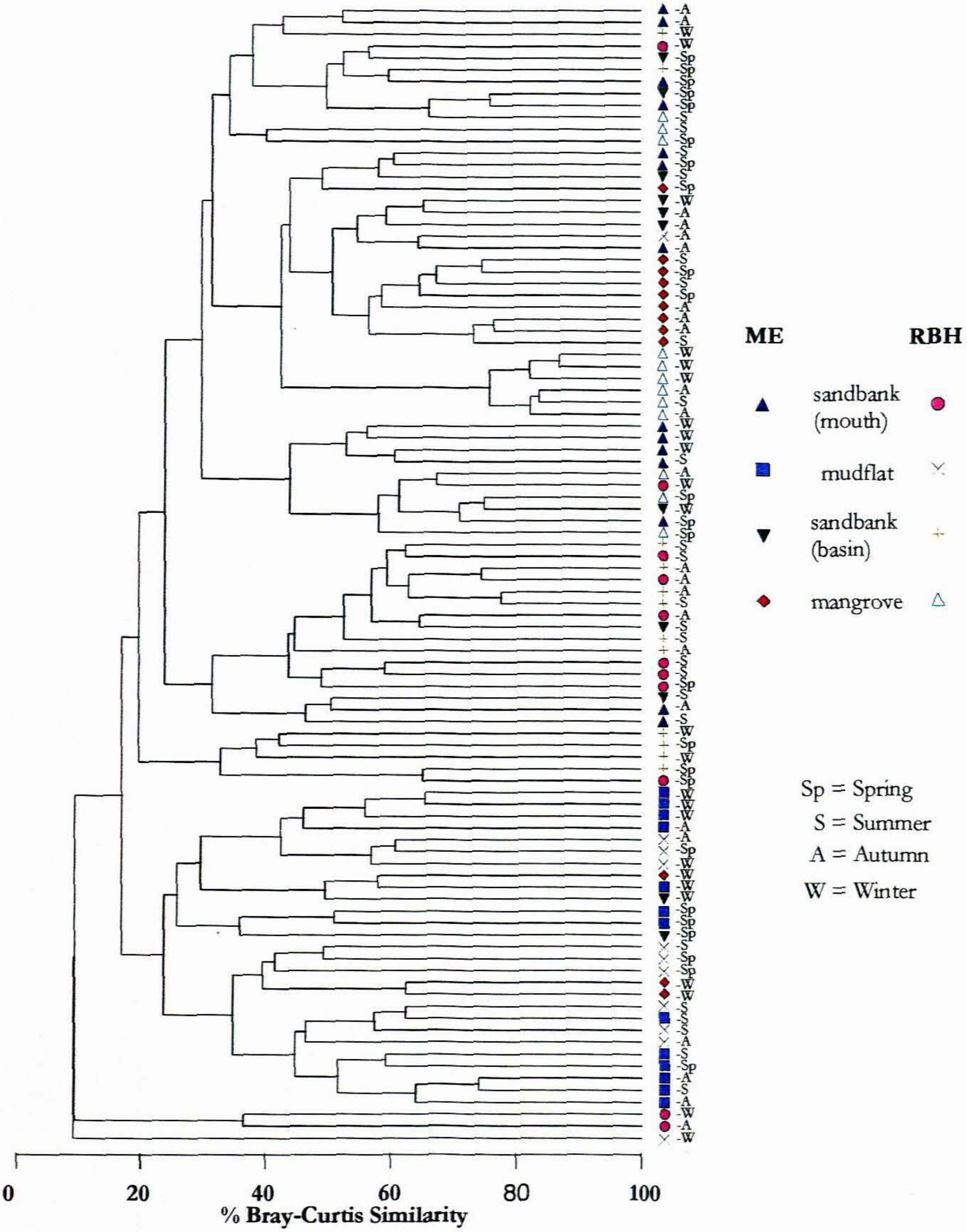


Figure 6.4: Dendrogram of Bray-Curtis similarities in densities of fishes (square root transformed) from common habitats sampled during different seasons in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH).

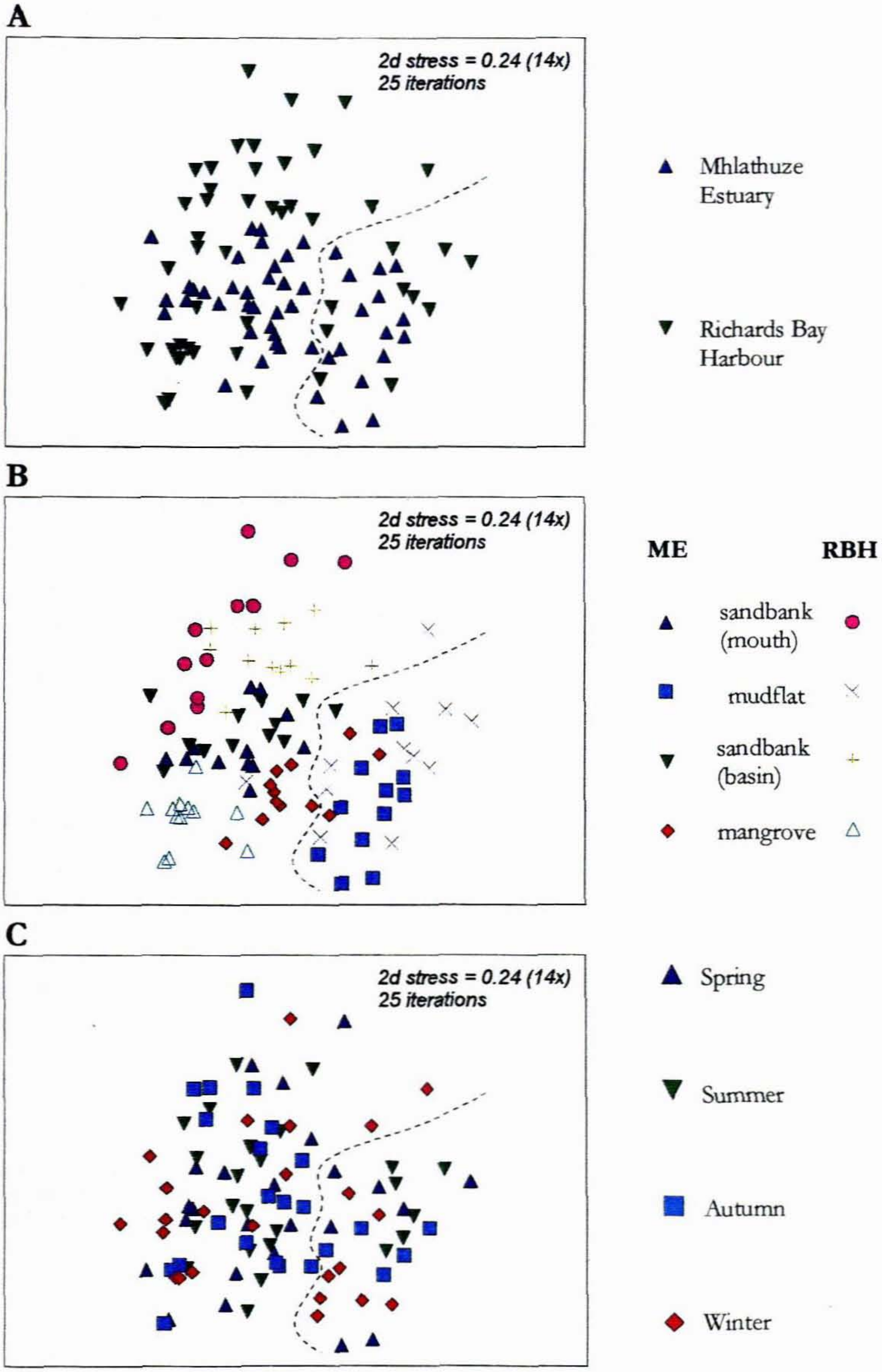


Figure 6.5: MDS ordination of densities of fishes (square root transformed) from common habitats in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH). Samples labelled by A) system B) habitat and C) season. Dashed line indicates two main groups that separate at approximately 18% Bray-Curtis similarity.

Table 6.6: Analysis of similarities of fish assemblages sampled from different habitats in Mhlathuze Estuary and Richards Bay Harbour. * = P <0.05, ** = P <0.01, *** = P <0.001, ns = P >0.05, not significant. Significant differences between similar habitats are shown in **bold**.

Test	R statistic	P
Differences between systems		
Global Test	0.111	***
Differences amongst habitats		
Global Test	0.602	***
Pairwise Tests		
ME sandbank (mouth) vs. ME sandbank (basin)	0.074	ns
ME sandbank (mouth) vs. ME mudflat	0.824	***
ME sandbank (mouth) vs. ME mangrove	0.528	***
ME sandbank (mouth) vs. RBH sandbank (mouth)	0.444	***
ME sandbank (mouth) vs. RBH sandbank (basin)	0.713	***
ME sandbank (mouth) vs. RBH mudflat	0.676	***
ME sandbank (mouth) vs. RBH mangrove	0.583	***
ME sandbank (basin) vs. ME mudflat	0.815	***
ME sandbank (basin) vs. ME mangrove	0.528	***
ME sandbank (basin) vs. RBH sandbank (mouth)	0.352	**
ME sandbank (basin) vs. RBH sandbank (basin)	0.491	***
ME sandbank (basin) vs. RBH mudflat	0.759	***
ME sandbank (basin) vs. RBH mangrove	0.472	***
ME mudflat vs. ME mangrove	0.731	***
ME mudflat vs. RBH sandbank (mouth)	0.861	***
ME mudflat vs. RBH sandbank (basin)	0.852	***
ME mudflat vs. RBH mudflat	0.167	ns
ME mudflat vs. RBH mangrove	0.981	***
ME mangrove vs. RBH sandbank (mouth)	0.759	***
ME mangrove vs. RBH sandbank (basin)	0.741	***
ME mangrove vs. RBH mudflat	0.62	***
ME mangrove vs. RBH mangrove	0.676	***
RBH sandbank (mouth) vs. RBH sandbank (basin)	0.056	ns
RBH sandbank (mouth) vs. RBH mudflat	0.861	***
RBH sandbank (mouth) vs. RBH mangrove	0.667	***
RBH sandbank (basin) vs. RBH mudflat	0.815	***
RBH sandbank (basin) vs. RBH mangrove	0.963	***
RBH mudflat vs. RBH mangrove	0.796	***
Differences amongst seasons		
Global Test	0.314	***
Pairwise Tests		
spring vs. summer	0.324	***
spring vs. autumn	0.296	**
spring vs. winter	0.310	**
summer vs. autumn	0.093	ns
summer vs. winter	0.509	***
autumn vs. winter	0.505	***

Table 6.7: Average similarity within system and habitat groups and species contribution >10% similarity. Average abundance as fish per 100 m^2.

Taxon	average abundance	% contribution	cumulative %
ME average similarity = 18.91			
Ambassis spp.	62.87	39.50	39.50
Mugilidae spp.	56.90	21.06	60.55
Stolephorus spp.	10.84	11.48	72.04
RBH average similarity = 11.78			
Ambassis spp.	142.55	36.36	36.36
Sillago sihama	37.42	21.26	57.62
Mugilidae spp.	72.11	16.32	73.64
Stolephorus spp.	26.64	10.28	84.22
ME sandbank (mouth) average similarity = 27.87			
Mugilidae spp.	162.08	66.65	66.65
Ambassis spp.	38.89	12.77	79.42
RBH sandbank (mouth) average similarity = 16.34			
Mugilidae spp.	46.73	51.40	51.40
Ambassis spp.	25.22	17.90	69.30
Gerres spp.	21.62	12.47	81.77
ME sandbank (basin) average similarity = 26.24			
Ambassis spp.	73.40	65.87	65.87
Mugilidae spp.	64.65	16.81	82.68
RBH sandbank (basin) average similarity = 22.39			
Sillago sihama	102.01	52.63	52.63
Ambassis spp.	31.53	23.29	75.92
Stolephorus spp.	43.23	12.32	88.23
ME mudflat average similarity = 25.51			
Stolephorus spp.	17.95	22.69	22.69
Thryssa spp.	21.94	20.77	43.46
Glossogobius callidus	12.45	19.92	63.38
Oligolepis keiensis	14.40	16.60	79.98
RBH mudflat average similarity = 18.86			
Stolephorus spp.	62.91	37.17	37.17
Leiognathus equula	16.98	25.79	62.96
Oligolepis keiensis	7.92	14.74	77.70
ME mangrove average similarity = 33.27			
Ambassis spp.	138.76	71.91	71.91
Oligolepis keiensis	19.28	10.52	82.44
RBH mangrove average similarity = 31.99			
Ambassis spp.	495.36	53.86	53.86
Mugilidae spp.	247.14	42.56	96.42

Taxa that played important roles in forging dissimilar fish assemblages at similar habitats in the different systems are given in Table 6.8. Different abundances in dominant fish species, rather than differences in the types of species that dominated, influenced dissimilarities between systems and common habitats. The factors that most distinguished the two systems were higher average densities of *Ambassis* spp., mullet and *S. sihama* in the harbour (Table 6.8). *Sillago sihama* was largely restricted to sandbanks in both systems, and differences in abundances were reflected at these habitats. However, mullet and *Ambassis* spp. occurred at sandbanks and mangroves. At sandbanks their abundances were actually higher in the estuary than the harbour, so the high concentrations of these fish in harbour mangroves clearly influenced the statistic above. Differences in abundances of these taxa contributed to significant differences between fish assemblages associated with similar sandbank habitats from different systems (Table 6.8). Sandbanks within each system did not differ in their associated fish assemblages. Comparisons across system basins and mouths have been included in Table 6.8. As noted above, mullet and ambassids were always more abundant on estuary sandbanks than those in the harbour. *Sillago sihama* occurred in higher densities at harbour sandbanks. Although mullet and ambassids were plentiful at both mangrove habitats they were more abundant at harbour mangrove fringes. This was the major factor distinguishing mangrove areas in the two systems (Table 6.8).

Table 6.8: Average dissimilarity between system and habitat groups and species contributing >10% dissimilarity. Average abundance as fish per 100 m^2.

Taxon	average abundance	average abundance	% contribution	cumulative %
Groups compared →	ME	RBH	average dissimilarity 87.24	
Ambassis spp.	62.87	142.55	28.30	28.30
Mugilidae spp.	65.90	72.11	20.92	49.22
Sillago sihama	4.69	37.42	10.62	59.85
Groups compared →	ME sandbank (mouth)	RBH sandbank (mouth)	average dissimilarity 84.08	
Mugilidae spp.	162.08	38.06	37.79	37.79
Ambassis spp.	38.89	25.22	13.83	51.62
Sillago sihama	9.49	46.73	12.96	64.58
Groups compared →	ME sandbank (basin)	RBH sandbank (basin)	average dissimilarity 83.93	
Sillago sihama	8.54	102.01	26.59	26.59
Ambassis spp.	73.40	31.53	24.25	50.84
Mugilidae spp.	64.65	2.12	14.14	64.98
Stolephorus spp.	3.89	43.23	13.55	78.53
Groups compared →	ME sandbank (basin)	RBH sandbank (mouth)	average dissimilarity 84.27	
Ambassis spp.	73.40	25.22	28.05	28.05
Mugilidae spp.	64.65	38.06	22.92	50.97
Sillago sihama	8.54	46.73	16.52	67.49
Groups compared →	ME sandbank (mouth)	RBH sandbank (basin)	average dissimilarity 87.34	
Mugilidae spp.	162.08	2.12	31.79	31.79
Sillago sihama	9.49	102.01	21.81	53.60
Stolephorus spp.	12.23	43.23	12.36	65.96
Ambassis spp.	38.89	31.53	11.67	77.63
Groups compared →	ME mudflat	RBH mudflat	average dissimilarity 81.77	
Stolephorus spp.	17.95	62.91	23.59	23.59
Thryssa spp.	21.94	7.78	14.57	38.16
Oligolepis keiensis	14.40	7.92	11.75	49.91
Leiognathus equula	4.76	16.98	11.30	61.21
Groups compared →	ME mangrove	RBH mangrove	average dissimilarity 82.41	
Ambassis spp.	138.76	495.36	51.73	51.73
Mugilidae spp.	35.80	247.14	31.65	83.38

6.2 Discussion

6.2.1 Physico-chemical properties as environmental factors influencing fishes associated with structurally similar habitats in Mhlathuze Estuary and Richards Bay Harbour

Mhlathuze Estuary and Richards Bay Harbour differed in the physico-chemical characteristics of their waters. Temperature, dissolved oxygen and turbidity were all higher in the estuary than the adjacent harbour. Actual differences in the first two variables were unlikely to have significant influence in forging different fish communities in the two systems. Fishes sampled were either estuarine or marine in origin, and temperatures recorded were typical of KwaZulu-Natal estuaries (Cyrus 1988) and closely matched those in the marine environment. Although dissolved oxygen did fall below 6 mg/l in one harbour habitat, it did not appear to influence the fishes present and it is likely that this variable becomes limiting to fishes in South African estuaries only at lower concentrations (Chapter 5). However, average turbidity in the estuary was double that in the harbour and differences in this variable were probably much more biologically meaningful. In the estuary average turbidity was over 10 NTU while in the harbour it was below 10 NTU. Cyrus and Blaber (1987b) found that several species commonly occurring in subtropical South African estuaries display strong preferences for either clear (<10 NTU) or more turbid water (>10 NTU). Moreover, turbidity preferences have been shown to influence the distribution of fishes within systems (Cyrus & Blaber 1987a,b,c) and between systems (Whitfield 1980). In this study relatively few species occurred in one system and not the other, and those that did were recorded in numbers too low to draw any conclusions regarding the role of turbidity in influencing their absence from the other system. However, it is probable that the relative abundances of a number of species between the systems may have been directly affected by turbidity differences. Nine taxa with known turbidity preferences were recorded in sufficient numbers to allow comparison of their abundances across the systems (Table 6.9). The relative abundances of only three of these (two *Rhabdosargus* species and *Leiognathus equula*) in the turbid Mhlathuze Estuary and clear Richards Bay Harbour, did not match their published designations as clear or turbid water species. Strong habitat preferences for *Zostera* beds were probably the reason for higher abundances of *Rhabdosargus* species in the estuary, and in the harbour *Leiognathus equula* was restricted in distribution to mudflats and mangroves where turbidities were in fact greater than 10 NTU. Thus, only the apparent preference of *Terapon jarbua* for turbid water does not match its designation as a species indifferent to turbidity. Turbidities differed between all common habitats, but given that waters at the sandbanks in the harbour were below the 'clear' limit (10 NTU, Cyrus & Blaber 1987a,b,c) but above it in the estuary, it was probable that some differences in fish assemblages between these habitats particularly, would be turbidity-induced.

Table 6.9: Correlation of known turbidity preferences of juveniles of selected species and their occurrence in Mhlathuze Estuary (ME; turbid system) and Richards Bay Harbour (RBH; clear system). Turbidity groups adapted from Cyrus and Blaber (1987c), clear = <10 NTU, turbid = ≥10 NTU.

Taxon	Turbidity group	Average abundance (fish per 100 m^2)		Correlation
		ME	RBH	
Gerres spp.	clear	38.96	339.48	yes
Rhabdosargus holubi	clear	247.99	62.92	no
Rhabdosargus sarba	clear	23.33	12.50	no
Leiognathus equula	turbid	107.50	286.25	no
Acanthopagrus berda	indifferent	105.28	81.25	yes
Pomadasys commersonnii	indifferent	86.35	86.98	yes
Terapon jarbua	indifferent	40.23	3.33	no
Elops machnata	turbid	70.90	10.00	yes
Solea bleekeri	turbid	90.06	6.25	yes

Salinity effects were a potential factor in the case of mudflat and mangrove fish assemblages. Marine salinities were not exceeded at any habitat but measurements below seawater were most notable at the estuary mudflat and mangrove fringe. This may have favoured a greater estuarine component in the fish faunas in these areas compared to the same structural habitat in the harbour. However, conditions at these habitats in both systems were distinctively estuarine anyway, in terms of their physical structure and physico-chemical conditions. Associated fish assemblages were therefore typical in comprising estuarine spawners, estuarine-dependant marine species and marine forms with no dependence on estuaries, but nevertheless physiologically capable of utilising their lower reaches. Comparison of salinities at these habitats with the reported tolerances of the abundant taxa sampled (Whitfield *et al.* 1981) suggested that salinity was not limiting to species' distributions based on osmoregulatory capacities. Indirect effects or salinity preferences may nevertheless have played a role in influencing relative abundances of some species.

6.2.2 Comparisons of fishes associated with structurally similar habitats in Mhlathuze Estuary and Richards Bay Harbour

Whitfield (1983, 1998) and Whitfield and Marais (1999) listed several physico-chemical factors as well as habitat variability, size of estuary, mouth phase and zoogeographical considerations that influence utilisation of South African estuaries by fishes. In the present comparison many of these factors could be regarded as extraneous. Habitat variability was accounted for by only considering similar habitats and the systems studied were of similar size. The proximity of Mhlathuze Estuary and Richards Bay Harbour eliminated the need to consider zoogeographical factors and both systems are permanently open to the same stock of potential recruits. Under conditions of typical rainfall and water abstraction from Mhlathuze River, both systems function as marine dominated embayments and their water properties are broadly similar, with exceptions discussed above. This is influenced by bathymetric factors as there is little doubt that more freshwater flows into the estuary than the harbour, but most bypasses the main estuary embayment. The harbour entrance is much larger than the estuary mouth and stronger tidal exchanges could facilitate higher recruitment into harbour waters. This might be countered by greater freshwater influence in the estuary outflow as reduced salinities, or olfactory signals carried by such freshwater outflow, are believed to act as cues for recruiting fish larvae (Boehlert & Mundy 1988, Martin *et al.* 1992, Whitfield 1994c, Harris & Cyrus 1996). Under these conditions similar numbers of fish species were sampled in the two systems during the study period. Moreover, there was a large overlap in the types of species present in either system. However, the number of species per sample, species richness and diversity was greater in the estuary and the fish fauna was more typically estuarine with a greater proportion of individuals being either estuarine spawners, or estuarine-dependent marine species. Several species recorded in the harbour occurred infrequently and/or in low abundances. Analysis of the estuarine association categories and full species assemblages from both systems confirmed that although the same types of fishes use the estuary and the harbour, they do so in different abundances.

Foregoing discussion indicated that it was likely that turbidity played an important role in these differences, especially at sandbank habitats. Blaber and Blaber (1980) postulated that juvenile estuarine associated fishes in south-east Africa, and the Indo-Pacific region in general, may be attracted to estuaries specifically because they are areas of shallow and turbid water. Fishes that typically utilise estuaries are adapted to turbid conditions (Blaber 1981). Laboratory investigations and/or field observations of 20 species occurring in KwaZulu-Natal estuaries (Cyrus & Blaber 1987c) substantiated this opinion. Three species were indifferent to turbidity, four were regarded as clear water species, with preferences for waters less than 10 NTU, and 13 favoured turbidities over 10 NTU (Cyrus & Blaber 1987c). Despite this there are few published comparisons of clear and turbid systems from subtropical South Africa, possible due to the paucity of clear water systems. Whitfield (1983) cited studies that indicated that the turbid Bashee Estuary supports only ten common fish species while 62 occur in the nearby Mngazana, which is a relatively clear system. However, more than 60% of the Mngazana fish species were associated with *Zostera* beds (Branch & Grindley 1979), a habitat not available in the Bashee. Lists of fish species from northern KwaZulu-Natal estuarine systems (Whitfield 1980) indicate far more species in Kosi, which is a clear water system, than St Lucia, which is turbid. The reef at Kosi mouth, which attracts an abundance of reef taxa such as butterfly-, surgeon- and damselfishes, confounds this comparison.

The preference of *Gerres* spp. and *S. sihama* for sandbanks rather than other habitats in the harbour has been discussed in Chapter 5. Turbidity preferences and/or favoured foraging habitats were given as reasons for their abundance at sandy areas. These species also occurred in Mhlathuze Estuary where they were also largely restricted to sandbanks, but in densities less than 15% those reported from the harbour. In the lack of any other consistent differences between these sandbanks from separate systems, turbidity can be cited as an important factor influencing the distributions of these taxa between Mhlathuze Estuary and Richards Bay Harbour. *Sillago sihama*, in particular, was an important species contributing to dissimilarities of fish assemblages between whole systems, and sandbank habitats across systems. It is therefore apparent that turbidity was a factor in forging different fish communities in Mhlathuze Estuary and Richards Bay Harbour, as well as influencing the role of structurally similar sandbank habitats in these separate systems.

Mullet and ambassids were also important in influencing the dissimilarity between fish faunas of Mhlathuze Estuary and Richards Bay Harbour, and specifically those assemblages associated with sandbanks and mangroves. Mullet species studied by Cyrus and Blaber (1987c), which included the common species in Mhlathuze Estuary and Richards Bay Harbour (Weerts & Cyrus 1998, CRUZ unpublished data), were indifferent to turbidity over the ranges recorded in this study. Although densities of mullet differed markedly at sandbanks and mangroves, their average densities (Table 6.8) and total abundances (Chapters 4 and 5) in either system were quite similar. The same amounts of fish in each system were therefore distributed differently. The role of increased predation threat to mullet due to lower turbidities in the harbour, has been discussed (Chapter 5) and explains reduced densities at harbour sandbanks, and higher abundances at harbour mangroves. This also holds true for ambassids, but doesn't explain the much higher densities (Table 6.8) and total abundances (Chapters 4 and 5) of Ambassis *spp.* in the harbour. No work has been done on the turbidity preferences of ambassids in South Africa. However, all three species have been reported in waters far more turbid than those measured in the present study (160 NTU in the case of *A. natalensis* and *A. productus*, 320 NTU for *A. gymnocephalus*; Martin 1983). It is unlikely that turbidity limited their abundance in the estuary. *Ambassis gymnocephalus* was the dominant species in Mhlathuze Estuary and Richards Bay Harbour. It is the most stenohaline of the three species and frequents the mouth areas of permanently open estuaries, usually in salinities above 28‰ (Martin 1988). In the laboratory this species tolerates freshwater for extended periods (21 days; Martin 1988) and field observations in this study indicated that densities were not affected by short-term (tidal) fluctuations below 30‰. However, Sheaves (1998) noted that the abundances of several fish species common in Australian mangrove systems were negatively correlated with long-term (months and years) patterns of salinity extremes. Long-term salinity fluctuations in Mhlathuze Estuary are far more severe than in the harbour. During floods the whole estuary may be reduced to freshwater for several days (CRUZ unpublished data) and under such conditions *A. gymnocephalus* would be displaced from the system (Martin 1988). Most estuarine fishes, and many estuarine-dependent species are probably capable of tolerating flooding, and their numbers may not be as adversely affected as those of *A. gymnocephalus*. Ambassids use estuaries as adults and juveniles and forced emigration into the marine environment probably has more severe consequences for this species than for most other estuarine-dependent marine fishes. Immigration back into the system may begin soon after floodwaters start receding, but it is likely that many individuals would be lost from the system. Pulses of freshwater therefore probably prevent the abundances of ambassids in the estuary reaching the levels that they do in the harbour.

Thus, differences in the dominant fish taxa typifying assemblages associated with structurally similar habitats in different systems can be explained in terms of the influences of physico-chemical factors and biological processes. Turbidity, long-term maximum salinity deviations and predation pressure are likely factors influencing the major species components of fish communities in Mhlathuze Estuary and Richards Bay Harbour. These factors differed in the two systems, and influenced the roles of structurally similar habitats as juvenile fish nursery areas.

Mudflats were the only habitats common to both systems, which showed no differences in the fish assemblages they accommodated. They were notably the only habitat types which did not support high densities of mullet, *Ambassis* spp. or *S. sihama*.

6.3 Conclusions

Preceding chapters, which considered fishes occurring in different habitats within individual systems, indicated that physico-chemical variables played little role in structuring different assemblages of juvenile fishes, and that the structural nature of habitats was the primary determinant of the associated fish communities. However, Mhlathuze Estuary and Richards Bay Harbour were attractive as study systems to investigate the role of structure in habitat because they offered a good variety of structural habitats, but lacked steep gradients in variables that may have influenced fish distribution. Such gradients are typical of most permanently open estuaries. Comparison of fish assemblages in Mhlathuze Estuary and Richards Bay Harbour, and between similar habitats in each system, illustrated the importance of physico-chemical conditions as factors influencing habitat utilisation by different species, and how those influences can affect the fish fauna of entire systems.

Similar species of juvenile fishes occurred at similar habitats in the estuary and the harbour. Three taxa, *Ambassis* spp., mullet and *Sillago sihama* dominated both systems. However, they occurred in different abundances and differed in their utilisation of structurally similar sandbank habitats and mangrove fringes. This resulted in significantly different assemblages in the two systems. Only mudflats supported similar fish communities in both the estuary and the harbour. Turbidity, long-term salinity deviations and predation threat were likely influences on the distributions of the dominant three taxa, and these factors therefore affected the fish assemblages at habitats which were typified by one, or a set of these species.

Habitat structure is an important determinant of the types of fishes that occur in an area. However, the way in which habitat quality is influenced by physico-chemical conditions differs amongst species. Decreased turbidity can render open sandbanks more attractive as a nursery habitat to some species, and make them less suitable to others. Both abiotic and biotic processes are important in affecting the distribution of fishes in Mhlathuze Estuary and Richards Bay Harbour. This is typical of subtropical and tropical estuaries. These systems are generally characterised by higher numbers of species than temperate systems and the complexities of their biological interactions are greater (Blaber 1997). Throughout this study the underlying assumption has been that relative abundances of different species in different habitats reflect their habitat preferences. Potentially important influences on the three main taxa, which characterise the study systems and most similar habitats within them, have been put forward. These remain untested contentions and to emphasise their role in the distributions of fishes between the two study systems is undoubtedly a simplification of actual patterns and processes. A multitude of species-specific responses to physical and biological factors would have influenced the patterns noted in system and habitat utilisation. Even within broadly similar groups of fishes these factors could not be isolated in explaining differences in utilisation of habitats between systems. Gobies were much more abundant at Mhlathuze Estuary sandbanks than those in Richards Bay Harbour. It is possibly that they have a competitive advantage over *S. sihama* at sandbanks in more turbid systems, or that their tolerance for extended periods of low salinity confers advantage to goby species in allowing them to remain in the system during periodic floods. A lack of knowledge of the life histories or biologies of some species prevented even speculative comment on differences in their distributions between systems. The speartail goby *Oligolepis keiensis* was rare in Richards Bay Harbour despite an abundance of habitat that appeared to be structurally, and environmentally similar to areas where the species was abundant in Mhlathuze Estuary.

CHAPTER 7

General Conclusions & Management Implications

Chapter 7

General conclusions and management implications

7.1 General conclusions

Mhlathuze Estuary and Richards Bay Harbour are nurseries to a wide range of fishes, most of which are of tropical Indo-Pacific origin. Several species endemic to southern Africa occur as do numerous estuarine gobies recognised as world threatened by the IUCN. Both systems are marine dominated embayments and physico-chemical conditions within each are suitable for species typical of lower reaches of permanently open KwaZulu-Natal estuarine systems. Estuarine-dependent marine species dominate the fish faunas in terms of abundance of individuals. By virtue of their large sizes and habitat variability, and in the light of continued degradation of KwaZulu-Natal estuaries, these systems are important reservoirs for estuarine fishes in subtropical South Africa.

Distinct assemblages of fishes associate with different structural habitats in the estuary and the harbour. This concurs with the findings of many studies conducted elsewhere. All habitats are important nursery areas but to different species assemblages and one cannot be replaced with another. In the estuary *Zostera* supports a particularly diverse and abundant fish assemblage. The eelgrass beds in the Mhlathuze are presently probably the best representative stand of subtropical eelgrass in South Africa. The loss of this habitat would lead to a loss of biodiversity in the system and the region as a whole. *Zostera* is also an important nursery habitat for species targeted by recreational anglers. Mangrove fringes in both systems are important, but in the harbour clear waters and high predation pressure may enhance their significance as predation refugia. Loss of these habitats would substantially reduce the harbour's value in supporting high abundances of fishes, which are major trophic links to prized angling species in estuarine and marine environments. *Bruguiera* mangroves offer the only structurally complex habitat over hard sandy substrata in the harbour. This serves to attract very high abundances of several species. Different fish assemblages associate with sandbanks in either system but the mudflats appear to support similar fish communities.

Results here are broadly applicable to other permanently open KwaZulu-Natal systems in as much as they highlight the value of habitat variability for diverse fish assemblages. Habitat types are typical of those occurring in tidal subtropical estuaries. However, some of these habitat types are rare in KwaZulu-Natal. Several species of submerged aquatic macrophytes can occur in estuaries here, but they are restricted in the surface areas that they cover. Fish assemblages associated with submerged vegetation have been shown to vary depending on leaf morphology and/or other plant-specific characteristics (Jenkins & Sutherland 1997, Jenkins & Wheatley 1998). Fishes that associate with *Zostera* will probably differ from communities that may associate with other local macrophytes such as *Ruppia* or *Potamogeton*. In Siyaya Estuary, a small temporary open/closed system south of the Mhlathuze, *Potamogeton* does not appear to be an important habitat for juvenile fishes (*pers. obs*). Various factors, including salinity and turbidity, influence the distribution of submerged aquatic macrophytes (Adams *et al.* 1999) and these might also be expected to play a role in differences in fish assemblages that associate with different plants.

Physico-chemical conditions do not play a major role in directly influencing the distribution of small and juvenile fishes within the embayment areas of either Mhlathuze Estuary or Richards Bay Harbour, but this is because steep gradients in variables do not occur in either system. Comparison of similar habitats in the estuary and the harbour clearly indicated that physico-chemical variables influence fish assemblages associated with similar habitats across the systems. Therefore factors such as salinity and turbidity need to be examined when comparing habitats from different systems, or even habitats within the same system but at different localities. Moreover, variation in these parameters over different time scales should also be considered as a potential factor influencing fish utilisation of estuarine habitats. Biological influences such as predation, foraging efficiency, inter- and intra-species competition also need to be considered, and it should be expected that these factors would also be influenced by physico-chemical parameters.

This study concentrated on fishes during important developmental stages in their life histories. Survival of these young stages is obviously important in determining population sizes. Results indicated that different habitats are important and one cannot be adequately substituted for the other. Furthermore, in different systems similar habitats may support different fish assemblages.

7.2 Management implications

The need to conserve estuaries and protect estuarine habitat along the coast of KwaZulu-Natal has been recognised by scientists and managers (Begg 1978, 1984, Cyrus 1991, WBB 2000). At a broad level the South African National Water Act (Act 36 of 1998) legislates an obligation to set aside a 'reserve' water allocation which protects the ecological requirements of aquatic ecosystems. To this end Resource Directed Measures (RDM) for the protection of estuarine ecosystems have been formulated (DWAF 1999). These largely address issues pertaining to the quality and quantity of freshwater flowing into estuaries. In reality many more factors impact upon habitat variability and ecosystem functioning in these systems. In this regard it appears that no actual management objectives have been decided upon, let alone formally drafted into any official policy document. The role of harbours in KwaZulu-Natal as significant areas of estuarine habitat has been recognised (Cyrus & Forbes 1996, Forbes *et al.* 1996) but there is uncertainty as to how even the national legislation is applicable in these environments. In harbours at Durban and Richards Bay, port authorities have sponsored initiatives to increase the level of water quality and biological data, and the National Ports Authority does subscribe to Integrated Environmental Management procedures as stipulated in South African environmental legislation. However, as yet there are no formal, published objectives, or plans for water quality management (Walmsley *et al.* 1999) and no stated policies regarding the preservation of the vital role that these ports play as nurseries to an abundance of marine organisms.

The necessity for a set of management objectives prioritising the potential benefits to be gained from KwaZulu-Natal estuaries is well illustrated by consideration of the results of this study. The value of different habitats and their associated fish assemblages can be determined according to a number of criteria, but the ranking of these criteria is likely to vary from system to system depending on its location, present and historic condition and utilisation, land ownership and local interest groups. Representativeness, rarity and diversity are important considerations in managing for conservation purposes (Ratcliffe 1977) and are becoming increasingly significant in terms of economic benefits to be gained from eco-tourism. However, there are also financial benefits to be gained by managing estuarine habitat to improve stocks of targeted angling species. Favoured angling fishes are often not species that support subsistence fisheries, so managing specifically to increase yields for recreational anglers will not necessarily fulfil a social requirement of accounting for the needs of subsistence harvesters.

Representativeness

The fact that fish habitats in Mhlathuze Estuary and Richards Bay Harbour are representative of a range of estuarine habitats typical of the lower reaches of tidal subtropical South African estuaries has been emphasised throughout. The size of the systems and habitats sampled here are important considerations in the conservation potential of these areas. Most similar habitats in KwaZulu-Natal systems are small and fragmented, a factor which may have contributed to their conservation value seemingly being overlooked in the development of these systems. The large areas of several habitat types in Mhlathuze Estuary and Richards Bay Harbour highlight the conservation importance of these systems, but should not distract from the need to conserve similar estuarine habitats along the KwaZulu-Natal coast. The transitional nature of the South African estuarine fish fauna from the subtropics in the north, to the temperate systems in the south, clearly indicates the need to take biogeographical considerations into account.

Furthermore, the roles of habitat types in supporting these fishes can differ between systems even in close proximity due to physico-chemical factors.

The canal mangroves in Richards Bay Harbour are obviously not representative of canal habitat as it typically occurs in canal estates and marinas. The role of this type of habitat should be assessed before canal developments are authorised in any estuaries. Marinas have been proposed at several systems along the KwaZulu-Natal coast, most recently at the Mzimkulu Estuary, and the development of a residential canal estate is planned in Richards Bay Harbour. It is unlikely that such canals will support fish assemblages similar to those reported from the Bhizolo Canal, and they will probably reduce the nursery value of these systems unless ecological considerations are taken into account in their planning and design.

Rarity

That ongoing degradation of estuarine habitat in southern Africa is a major threat to fish conservation in the region is now accepted fact (Cyrus 1991, Whitfield 1997). The large areas of habitats referred to above should therefore be assessed not only in terms of representativeness but also rarity. *Avicennia* mangroves and sandbanks in Mhlathuze Estuary and Richards Bay Harbour clearly render these systems important. However, the rarity of *Zostera* eelgrass and *Bruguiera* mangroves from KwaZulu-Natal systems, and the distinct fish assemblages they support, highlights the exceptional conservation value of these habitats as they occur in the estuary and the harbour respectively.

Setting a conservation goal to conserve *Zostera* in Mhlathuze Estuary and KwaZulu-Natal in general is critical. As noted above it is unlikely that other submerged estuarine macrophytes will support similar fish assemblages as *Zostera* in the Mhlathuze. Furthermore, *Zostera* is fragile as a habitat type and is susceptible to natural and anthropogenic impacts. The situation as it is presently, with eelgrass limited to only two systems, is precarious. Measures should be taken to reintroduce this macrophyte to systems where it previously occurred.

Considerable scientific information is available on the environmental requirements of *Zostera capensis* (Warne 1994, Adams *et al.* 1999) and restoration projects with a congeneric, *Z. marina*, have been successful elsewhere (Davis & Short 1997).

Successes have also been achieved in replanting mangroves in estuarine habitats (Rönnbäck *et al.* 1999) but the application of techniques used elsewhere may not be applicable in the South African context. The re-introducing of mangroves to most systems where they have been lost will necessitate hydrological manipulations rather than simply planting saplings. From results here, the value of the *Bruguiera gymnorrhiza* habitat in the harbour cannot be ascribed with certainty to any particular characteristic of this mangrove species, such as the complexity of buttress root structure rather than the pneumatophores of *Avicennia marina*. The presence of structure in a habitat over hard sand substratum rather than soft mud could equally be the reason for the high abundances of fishes in this area of the harbour. *Bruguiera* mangroves are generally secondary colonisers and usually establish in the middle sections of *Avicennia* stands (Berjak *et al.* 1997). The largely monospecific stand in the harbour with an open water fringe may therefore be atypical of this species. Nevertheless, within Richards Bay Harbour, it is an important fish habitat that would benefit from protection afforded should the stand be designated as a Natural Heritage Site.

Diversity

Data collected here suggest that the absence of eelgrass from estuaries where it could occur probably limits fish species diversity. This adds considerable conservation value to this habitat, along with its rarity and fragility, and further motivates re-introduction of the species to estuaries where it previously occurred. Maintenance of biodiversity is a major objective of most conservation efforts. The diversity of small estuarine species in Mhlathuze Estuary is greater than previously thought and this can be chiefly ascribed to eelgrass in the system. Transplantation of *Zostera* into the harbour should be investigated, but the creation of a new habitat will be at the expense of another. Cyrus and Wepener (1998) motivated for an area in Richards Bay Harbour that included a large section of mudflat to be set-aside for conservation purposes. However, results from this study indicated that mudflats in the estuary and the harbour support similar juvenile fishes, and that this is the one habitat type that is duplicated in the region. Should conditions be suitable, this would be an ideal area to establish *Zostera* beds. This would enhance the value of a so-called 'Green Area' in the harbour if ever officially sanctioned and given conservation status.

Fisheries importance

As noted above management policies for recreational fisheries will differ from those for subsistence fisheries. Species favoured by recreational anglers in South Africa typically include the sea breams (Sparidae), emperors (Lethrinidae), snappers (Lutjanidae), rockcods (Serranidae) and barracudas (Sphyraenidae). Fishes from these families were sampled extensively, and often exclusively, in the estuary eelgrass. It is therefore likely that restoration of eelgrass areas in KwaZulu-Natal estuaries will increase populations of these species and improve recreational catches. However, other habitats are also important as nurseries for major prey species which are trophic links to highly prized piscivorous species, such a king- and queen fishes.

Methods of subsistence gathering usually involve netting and catches often comprise mostly of shoaling species. In KwaZulu-Natal estuaries mullet are probably harvested most abundantly. This study indicated that mullet have no particular preference for *Zostera* and occur more abundantly at mangrove fringes or even over open sandbanks. Management of KwaZulu-Natal systems to increase coverage of eelgrass will therefore not increase subsistence catches.

Gray *et al.* (1996) noted that before the true value of different nursery habitats to fisheries resources can be identified, the contribution of juvenile fishes residing in different nursery habitats needs to be quantified. This applies to important fisheries species noted above, and is also pertinent to the more general findings of this study. Estuarine fishes are naturally adaptable, and it may be expected that most will exhibit high plasticity in the types of habitat within which they can meet their biological requirements as maturing juveniles. The use of alternative nursery habitats by at least some South African fishes that usually associate with estuaries as juveniles has been noted by other researchers (tidal pools, Beckley 1985, 2000; surf zones, Lasiak 1986). Of these habitats only surf zones have been seriously contemplated as possible significant alternatives to estuarine nurseries. However, this aspect of habitat utilisation has not been well researched, probably due to the difficulties inherent in sampling surf zones on high-energy coastlines. Studies where good quantitative data have been used in addressing the issue have, of necessity relied on larval, or postlarval fish densities, rather than abundances of juveniles. Until otherwise indicated it therefore seems prudent to regard most of the estuarine-dependent species in this study (Appendix 1) as largely dependent on these habitats for their survival, or at least for healthy populations.

Results from this study have indicated that descriptions of shallow water habitats and their use by fishes are prerequisites for effective conservation and optimal management of fish resources in KwaZulu-Natal estuaries and harbours. Indications are that habitats can be managed to manipulate fish communities to meet desired management objectives. However, physical and biological interactions amongst different habitats form integrated ecosystems, and this needs to be considered in the formulation of management policies.

CHAPTER 8

References

Chapter 8

References

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APPENDICES

Appendix 1

Checklist of species captured in Mhlathuze Estuary (ME) and Richards Bay Harbour (RBH) and their classification into categories based on estuarine affiliation (EA) and biogeographical range (BR). e = estuarine, edm = estuarine-dependent marine, m = marine, s = subtropical, w = warm temperate, c = cool temperate.

Family	Species	EA	BR	ME	RBH
Elopidae	*Elops machnata*	edm	sw	X	X
Megalopidae	*Megalops cyprinoides*	edm	s	X	X
Clupeidae	*Gilchristella aestuaria*	e	swc	X	
	Herklotsichthys quadrimaculatus	m	s	X	X
	Hilsa kelee	m	s	X	X
	Dussumierinnae: Tribe Dussumierinni	m	s		X
	Dussumierinnae: Tribe Spratelloidini sp. 1	m	s	X	X
	Dussumierinnae: Tribe Spratelloidini sp. 2	m	s	X	X
Engraulidae	*Stolephorus* spp.	m	sw	X	X
	Thryssa spp.	edm	s	X	X
Chanidae	*Chanos chanos*	m	sw	X	X
Atherinidae	*Atherinomorus lacunosus*	e	s	X	X
Hemiramphidae	*Hyporhamphus capensis*	e	sw		X
Syngnathidae	*Hippichthys cyanospilos*	e	s	X	
	Hippichthys spicifer	e	s	X	X
Solenostomidae	*Solenostomus* sp. 1	m	sw	X	
Scorpaenidae	*Pterois miles*	m	sw	X	
	Sebastapistes strongia	m	s	X	
Platycephalidae	*Platycephalus indicus*	m	sw	X	X
Triglidae	Triglidae sp. 1	m	swc	X	
Ambassidae	*Ambassis* spp.	edm	sw	X	X
Serranidae	*Epinephelus malabaricus*	m	s	X	
	Epinephelus sp. 1	m	sw	X	
Teraponidae	*Pelates quadrilineatus*	m	s	X	
	Terapon jarbua	edm	sw	X	X
Apogonidae	*Foa brachygramma*	m	s	X	
Haemulidae	*Pomadasys commersonnii*	edm	sw	X	X
	Pomadasys kaakan	edm	s	X	X
	Pomadasys olivaceum	m	sw	X	X
Lutjanidae	*Lutjanus fulviflamma*	m	sw	X	
	Lutjanus sp. 1	m	s	X	X
	Lutjanus sp. 2	m	s		X
Sparidae	*Acanthopagrus berda*	edm	sw	X	X
	Crenidens crenidens	m	s	X	X
	Diplodus sargus capensis	m	swc	X	X
	Rhabdosargus globiceps	m	wc		X
	Rhabdosargus holubi	edm	swc	X	X
	Rhabdosargus sarba	edm	sw	X	X
	Rhabdosargus thorpei	edm	s	X	X
	Sparidae sp. 1	m	sw		X

Appendix 1 continued...

Appendix 1 continued

Family	Species	EA	BR	ME	RBH
Lethrinidae	*Lethrinus* spp.	m	s	X	
Monodactylidae	*Monodactylus argenteus*	edm	sw	X	
	Monodactylus falciformis	edm	sw	X	
Gerreidae	*Gerres* spp.	edm	sw	X	X
Mullidae	*Upeneus vittatus*	m	sw	X	
Sillaginidae	*Sillago sihama*	m	s	X	X
Sciaenidae	Sciaenidae sp. 1	m	sw	X	
Leiognathidae	*Leiognathus equula*	m	s	X	X
Carangidae	*Caranx* spp.	edm	sw	X	X
	Scomberoides sp.	edm	s	X	X
Pomacentridae	Pomacentridae sp. 1	m	sw	X	
	Pomacentridae sp. 2	m	sw	X	
Labridae	*Halichoeres* sp.	m	s	X	
	Labridae spp.	m	s	X	
Scaridae	Scaridae spp.	m	s	X	
Mugilidae	Mugilidae spp.	edm	sw	X	X
Sphyraenidae	*Sphyraena barracuda*	edm	s	X	
	Sphyraena jello	m	s	X	
Blenniidae	*Omobranchus* sp. 1	m	sw		X
	Blenniidae sp. 1	m	sw	X	
	Blenniidae sp. 2	m	sw		X
Clinidae	Clinidae sp. 1	m	sw	X	
Callionymidae	*Callionymus marleyi*	m	swc		X
Gobiidae	*Acentrogobius audax*	e	s	X	X
	Bathygobius laddi	e	sw	X	X
	Caffrogobius natalensis	e	sw	X	
	Croilia mossambica	e	s	X	X
	Favonigobius melanobranchus	e	s	X	
	Favonigobius reichei	e	s	X	X
	Glossogobius biocellatus	e	s	X	X
	Glossogobius callidus	e	sw	X	X
	Mugilogobius inhacae	e	s	X	X
	Oligolepis acutipennis	e	s	X	X
	Oligolepis keiensis	e	sw	X	X
	Oxyurichthys spp.	e	sw	X	X
	Pandaka silvana	e	sw	X	X
	Periophthalmus koelreuteri africanus	e	s	X	X
	Redigobius balteatops	e	s	X	X
	Silhouettea sibayi	e	s	X	X
	Taenioides jacksoni	e	s	X	X
	Gobiidae sp. 1	e	sw	X	X
Eleotridae	*Eleotris* spp.	e	s	X	X
Kraemeriidae	*Kraemeria samoensis*	m	s		X
Siganidae	*Siganus sutor*	m	sw	X	

Appendix 1 continued...

Appendix 1 continued

Family	Species	EA	BR	ME	RBH
Bothidae	*Pseudorhombus arsius*	m	sw	X	
Cynoglossidae	*Paraplagusia bilineata*	m	s		X
Soleidae	*Solea bleekeri*	edm	swc	X	X
Monacanthidae	*Stephanolepis auratus*	m	sw	X	
Tetraodontidae	*Amblyrhynchotes honckenii*	m	swc	X	X
	Arothron immaculatus	m	sw	X	X
	Arothron meleagris	m	s		X
	Chelonodon laticeps	m	sw	X	X
	Torquigener hypselogeneion	m	sw	X	X
	Tetraodontidae sp. 1	m	sw		X

Appendix 2

Physico-chemical properties of water at habitats sampled in Mhlathuze Estuary (September 1997 to August 1998).

Habitat	Month	Temp. (°C)	Dissolved Oxygen (mg/l)	pH	Salinity (‰)	Turbidity (NTU)
sandbank (mouth)	September 1997	25.32	7.57	8.77	25.5	12
	October 1997	22.41	8.07	8.33	33.8	29
	November 1997	27.09	5.87	7.85	35.1	12
	December 1997	26.40	5.71	7.97	24.8	23
	January 1998	30.37	6.17	8.03	23.9	12
	February 1998	24.73	7.10	8.24	17.7	11
	March 1998	28.45	5.91	8.11	32.7	20
	April 1998	23.61	6.60	8.19	31.2	17
	May 1998	23.46	6.26	8.31	36.4	12
	June 1998	20.14	7.21	8.53	36.7	17
	July 1998	22.25	6.89	8.66	35.9	11
	August 1998	20.42	7.31	8.78	35.9	26
sandbank (basin)	September 1997	24.67	7.90	8.76	32.7	18
	October 1997	22.12	8.12	8.35	35.6	22
	November 1997	24.51	6.22	7.88	35.9	11
	December 1997	24.29	7.50	8.02	31.6	12
	January 1998	26.63	6.13	8.01	34.6	11
	February 1998	30.03	7.14	8.28	25.9	15
	March 1998	27.57	6.45	8.12	33.9	23
	April 1998	22.63	6.35	8.21	34.0	11
	May 1998	23.60	6.75	8.35	37.3	9
	June 1998	21.43	6.98	8.55	36.9	6
	July 1998	22.49	6.78	8.68	35.9	2
	August 1998	20.68	7.10	8.79	36.1	12
mudflat	September 1997	25.75	6.78	8.74	23.8	35
	October 1997	21.93	10.52	8.24	12.4	72
	November 1997	26.52	6.63	7.86	21.4	12
	December 1997	22.7	5.84	7.89	25.8	12
	January 1998	27.87	6.22	8.01	26.8	22
	February 1998	28.23	7.74	8.20	19.2	21
	March 1998	28.00	6.25	8.05	26.6	16
	April 1998	23.90	6.77	8.22	29.7	26
	May 1998	21.16	6.15	8.20	35.0	27
	June 1998	17.44	6.93	8.45	36.4	7
	July 1998	19.53	7.13	8.48	35.2	11
	August 1998	19.04	6.87	8.56	35.6	30

Appendix 2 continued...

Appendix 2 continued

Habitat	Month	Temp. (°C)	Dissolved Oxygen (mg/l)	pH	Salinity (‰)	Turbidity (NTU)
Avicennia mangrove	September 1997	26.20	7.47	8.68	2.1	30
	October 1997	22.15	8.18	8.34	35.5	19
	November 1997	25.16	6.23	7.81	35.6	10
	December 1997	26.23	8.38	8.07	5.4	18
	January 1998	27.92	5.44	7.92	33.3	16
	February 1998	24.75	6.72	8.41	25.1	28
	March 1998	29.83	6.48	8.19	26.8	37
	April 1998	22.70	6.43	8.20	36.1	9
	May 1998	24.14	6.34	8.33	36.5	15
	June 1998	21.24	7.05	8.54	36.8	8
	July 1998	22.92	6.88	8.68	35.9	6
	August 1998	21.17	7.42	8.79	36	21
Zostera	September 1997	21.64	8.92	8.83	24.1	8
	October 1997	23.63	10.64	8.48	34.2	27
	November 1997	24.57	6.63	8.15	27.5	8
	December 1997	22.57	6.56	7.92	32.1	13
	January 1998	30.26	7.83	8.27	29.2	15
	February 1998	26.12	9.50	8.44	29.8	12
	March 1998	26.43	6.30	8.20	28.0	18
	April 1998	23.42	7.64	8.41	36.1	11
	May 1998	20.79	6.24	8.23	36.1	11
	June 1998	18.18	7.01	8.50	36.2	4
	July 1998	20.20	7.10	8.59	35.1	5
	August 1998	18.99	7.73	8.77	36.4	21

Appendix 3

Fishes sampled from different habitats in Mhlathuze Estuary. Average density (fish per 100 m^2) from September 1997 to August 1998. * = endemic to southern Africa (Whitfield 1998), ⌫ = threatened species (Hilton-Taylor 2000).

Taxon	size range (mm)	sandbank (mouth)	sandbank (basin)	mudflat	*Avicennia* mangrove	*Zostera*	% total catch
*Rhabdosargus holubi**	7-50	5.729	4.861	3.646	6.429	323.264	26.25
Mugilidae spp.	6-50	162.083	64.653	1.065	35.799	15.486	21.30
Ambassis spp.	3-50	38.895	73.403	0.432	138.756	19.942	20.72
Oligolepis keiensis ⌫	8-36	13.299	0.208	14.401	19.277	21.736	5.26
Stolephorus spp.	7-50	12.234	3.889	17.951	9.297	2.986	3.54
Arothron immaculatus	8-50	0.243	0.035	0.174	0.217	30.799	2.40
Thryssa spp.	7-50	4.537	0.764	21.944	2.896	0.069	2.31
Rhabdosargus sarba	7-50	0.069	1.181	0.382	0.313	25.336	2.08
*Glossogobius callidus**	5-47	3.634	0.799	12.449	4.352	2.118	1.78
*Diplodus sargus capensis**	7-23	0.035	0.313	0.174	1.146	20.359	1.68
Sillago sihama	7-50	9.491	8.542	0.486	0.260	0.625	1.48
Lutjanus fulviflamma	12-50	0.122	0.035		0.035	16.053	1.24
Favonigobius reichei ⌫	5-49	7.326	4.097	0.035		2.326	1.05
Acanthopagrus berda	7-50	0.382	0.208	0.764	7.419	2.604	0.87
Leiognathus equula	4-50	2.697	0.417	4.757	1.088		0.68
Pomadasys commersonnii	8-50	2.662	1.493	0.590	2.451	1.146	0.64
*Solea bleekeri**	4-37	0.700	0.035	4.476	2.295	0.451	0.61
Pelates quadrilineatus	8-50					6.273	0.48
Elops machnata	17-36	2.697	0.243	1.597	1.372	0.208	0.47
Oligolepis acutipennis	8-43	0.926		0.278	4.340	0.174	0.44
Favonigobius melanobranchus ⌫	6-44		0.104			5.243	0.41
Gerres spp.	6-50	1.372	1.528	0.069	0.278	1.979	0.40
Gobiidae sp. 1	4-11	0.301	0.799	0.521	1.858	0.833	0.33
Terapon jarbua	8-50	3.148	0.208			0.382	0.29
Monodactylus argenteus	4-50			0.035	0.480	2.951	0.26
Chelonodon laticeps	18-50	1.528	1.875			0.035	0.26
*Caffrogobius natalensis**	9-50			0.035		2.847	0.22
Siganus sutor	13-50					2.604	0.20
Glossogobius biocellatus ⌫	8-50	0.405	0.313	0.139	0.625	1.076	0.20
Halichoeres sp.	7-37					2.465	0.19
Scaridae spp.	7-50					2.396	0.18
Lethrinus spp.	10-50					2.292	0.17
*Rhabdosargus thorpei**	7-50		0.104	0.104		1.782	0.15
Eleotris spp.	7-50	0.174	0.486		0.122	0.903	0.13
Pandaka silvana	6-16	0.087		0.035	1.470		0.12
Hippichthys spicifer	12-110	0.093				1.366	0.11
Herklotsichthys quadrimaculatus	17-50	0.093	0.347	0.069	0.671		0.09
Oxyurichthys spp.	10-46	0.122	0.035	0.208	0.694	0.069	0.09

Appendix 3 continued...

Appendix 3 continued

Taxon	size range (mm)	sandbank (mouth)	sandbank (basin)	mudflat	*Avicennia* mangrove	*Zostera*	% total catch
Sebastapistes strongia	8-49					1.042	0.08
Pomadasys kaakan	11-50	0.145		0.764	0.069		0.07
*Mugilogobius inhacae**	6-25				0.243	0.694	0.07
Epinephelus malabaricus	20-50					0.833	0.06
Caranx spp.	4-50	0.069	0.347	0.069	0.139	0.035	0.05
Acentrogobius audax	6-17					0.521	0.04
Bathygobius laddi	7-23		0.069			0.451	0.04
Sciaenidae sp. 1	4-10	0.035	0.035	0.281	0.069	0.035	0.03
Pomadasys olivaceum	9-16			0.382		0.069	0.03
Crenidens crenidens	5-47					0.451	0.03
Pseudorhombus arsius	8-50	0.104	0.104	0.174	0.035	0.035	0.03
Hilsa kelee	24-50		0.382	0.035			0.03
Hippichthys cyanospilos	46-101					0.417	0.03
Scomberoides sp.	21-50	0.174	0.139			0.069	0.03
Labridae spp.	7-11				0.069	0.313	0.03
*Croilia mossambica** ⌫	9-16		0.278		0.104		0.03
Pomacentridae sp. 2	4-8					0.347	0.03
Chanos chanos	10-14	0.174	0.069		0.035	0.069	0.03
Epinephelus sp. 1	19-50					0.278	0.02
Foa brachygramma	15-24					0.278	0.02
Megalops cyprinoides	20-24			0.208			0.02
Spratelloidini sp. 2	18-20	0.035			0.174		0.02
*Gilchristella aestuaria**	47				0.174		0.01
Sphyraena barracuda	23-50				0.035	0.139	0.01
Upeneus vittatus	45-50		0.139				0.01
Platycephalus indicus	10-50			0.104			0.01
Clinidae sp. 1	8			0.069			0.01
*Silhouettea sibayi** ⌫	12-17			0.035	0.035		0.01
*Taenioides jacksoni** ⌫	27				0.069		0.01
Spratelloidini sp. 1	16	0.035					<0.005
Atherinomorus lacunosus	43		0.035				<0.005
Solenostomus sp. 1	50					0.035	<0.005
Pterois miles	22					0.035	<0.005
Triglidae sp. 1	17					0.035	<0.005
Lutjanus sp. 1	16					0.035	<0.005
Monodactylus falciformis	15					0.035	<0.005
Pomacentridae sp. 1	13		0.035				<0.005
Sphyraena jello	37-50			0.035			<0.005
Blenniidae sp. 1	15		0.035				<0.005
Periophthalmus koelreuteri africanus	12-16		0.035				<0.005
*Redigobius balteatops**	11					0.035	<0.005
Stephanolepis auratus	25					0.035	<0.005
Amblyrhynchotes honckenii	50		0.035				<0.005
Torquigener hypselogeneion	25		0.035				<0.005

Appendix 4

Physico-chemical properties of water at habitats sampled in Richards Bay Harbour (September 1997 to August 1998).

Habitat	Month	Temp. (°C)	Dissolved Oxygen (mg/l)	pH	Salinity (‰)	Turbidity (NTU)
sandbank (mouth)	September 1997	24.31	7.26	8.82	22.9	4
	October 1997	24.30	9.16	8.40	26.9	5
	November 1997	22.09	6.47	7.97	33.6	4
	December 1997	23.83	5.72	7.85	24.5	5
	January 1998	27.13	5.98	7.99	32.8	4
	February 1998	26.86	7.27	8.43	29.8	4
	March 1998	27.10	6.60	8.15	32.6	3
	April 1998	24.60	6.86	8.23	33.5	5
	May 1998	23.41	6.67	8.30	34.9	3
	June 1998	19.99	6.40	8.45	35.7	4
	July 1998	21.79	6.76	8.70	36.1	3
	August 1998	20.63	7.31	8.79	35.9	3
sandbank (basin)	September 1997	22.68	8.80	8.91	33.4	4
	October 1997	21.93	8.75	8.39	35.7	14
	November 1997	20.30	6.74	7.84	33.9	7
	December 1997	23.27	5.89	7.91	35.1	6
	January 1998	28.69	5.33	7.94	34.2	6
	February 1998	23.70	7.71	8.46	34.9	5
	March 1998	25.97	6.96	8.19	34.4	4
	April 1998	23.14	7.33	8.24	35.0	12
	May 1998	22.86	5.07	8.27	37.5	3
	June 1998	20.97	6.40	8.49	36.3	3
	July 1998	21.11	6.72	8.61	35.9	21
	August 1998	21.53	7.14	8.74	36.2	15
mudflat	September 1997	23.43	6.95	8.81	33.0	15
	October 1997	21.12	7.74	8.23	36.0	8
	November 1997	20.96	6.38	7.87	35.4	9
	December 1997	23.97	5.50	7.69	32.8	12
	January 1998	26.87	5.71	7.92	34.2	9
	February 1998	26.48	8.11	8.38	33.8	8
	March 1998	26.67	5.52	7.82	34.0	23
	April 1998	21.99	5.00	7.89	36.6	11
	May 1998	20.40	5.07	8.06	36.8	14
	June 1998	19.09	5.88	8.42	36.7	7
	July 1998	18.51	6.14	8.45	36.3	8
	August 1998	18.94	6.34	8.59	36.1	8

Appendix 4 continued...

Appendix 4 continued

Habitat	Month	Temp. (°C)	Dissolved Oxygen (mg/l)	pH	Salinity (‰)	Turbidity (NTU)
Avicennia mangrove	September 1997	26.61	5.73	8.55	33.1	17
	October 1997	22.48	8.81	8.36	36.1	8
	November 1997	21.66	6.58	7.94	35.5	5
	December 1997	23.86	4.97	7.72	34.8	13
	January 1998	25.68	5.40	7.92	34.5	5
	February 1998	23.60	6.41	8.33	34.9	3
	March 1998	27.88	5.70	8.07	34.1	8
	April 1998	23.30	7.29	8.27	36.0	8
	May 1998	22.89	6.26	8.31	37.3	6
	June 1998	20.82	6.44	8.50	36.6	4
	July 1998	20.88	6.93	8.64	35.9	4
	August 1998	20.56	7.13	8.76	36.0	6
Avicennia mangrove canal	September 1997	23.81	5.42	8.26	29.5	14
	October 1997	22.22	7.16	8.04	32.2	14
	November 1997	21.37	6.55	7.72	23.6	7
	December 1997	24.37	5.22	7.69	30.9	8
	January 1998	29.22	5.86	7.92	33.4	16
	February 1998	23.63	5.74	7.93	31.1	7
	March 1998	27.70	5.13	7.98	32.7	16
	April 1998	23.30	5.2	7.91	33.7	19
	May 1998	21.18	4.84	8.03	35.4	10
	June 1998	18.52	5.96	8.23	34.6	12
	July 1998	19.53	6.10	8.4	33.1	7
	August 1998	19.53	6.36	8.54	33.8	29
Bruguiera mangrove	September 1997	24.60	8.22	8.84	32.7	4
	October 1997	22.41	8.80	8.32	36.0.	6
	November 1997	20.95	5.68	7.72	34.3	6
	December 1997	22.95	5.67	7.66	33.6	14
	January 1998	25.63	6.25	7.97	34.5	6
	February 1998	23.33	6.62	8.27	34.4	5
	March 1998	26.36	6.94	8.15	34.2	4
	April 1998	23.84	5.50	7.68	37.2	10
	May 1998	23.04	6.61	8.35	37.3	5
	June 1998	20.26	6.80	8.49	36.4	6
	July 1998	21.80	6.91	8.69	35.9	5
	August 1998	20.63	7.28	8.77	36.1	6

Appendix 5

Fishes sampled from different habitats in Richards Bay Harbour. Average density (fish per 100 m^2) from September 1997 to August 1998. * = endemic to southern Africa (Whitfield 1998), ⌫ = threatened species (Hilton-Taylor 2000).

Taxon	size range (mm)	sandbank (mouth)	sandbank (basin)	mudflat	*Avicennia* mangrove	*Avicennia* mangrove canal	*Bruguiera* mangrove	% total catch
Mugilidae spp.	7-50	38.056	2.118	1.111	1.007	247.135	1257.917	51.27
Ambassis spp.	4-50	25.217	31.528	18.090	49.969	495.365	281.701	29.89
Sillago sihama	7-50	46.727	102.014	0.660		0.260	12.847	5.39
Stolephorus spp.	7-50		43.229	62.908	0.295	0.417	1.736	3.60
Pomadasys olivaceum	8-34	1.632	5.729	0.069			43.576	1.69
Herklotsichthys quadrimaculatus	16-45	16.172	1.389	2.639		0.130	28.889	1.63
Gerres spp.	5-50	21.623	6.285	0.382			4.514	1.09
*Diplodus sargus capensis**	7-15	0.729	1.076		0.116		28.125	1.00
Leiognathus equula	9-50			16.979	2.101	6.875		0.86
Acanthopagrus berda	6-23	0.243	5.556	0.035		0.938	6.597	0.44
Gobiidae sp. 1	4-7	0.035		7.153	0.278	0.234	5.208	0.43
Pandaka silvana	6-17	0.035		0.313	2.685	7.552	2.083	0.42
Oligolepis keiensis ⌫	8-32			7.917	2.650			0.35
*Rhabdosargus holubi**	7-13	1.493	1.979	0.104		1.667	3.646	0.29
Thryssa spp.	8-50	0.043		7.778	0.203			0.27
Pomadasys commersonnii	8-50	4.340	2.674	0.104	0.099	0.130		0.24
*Silhouettea sibayi** ⌫	4-21	1.007	0.104	3.900				0.17
*Glossogobius callidus**	6-42			2.778		1.563	0.347	0.16
Chelonodon laticeps	11-50	0.139	0.382	1.042		0.313	1.042	0.10
Mugilogobius inhacae	6-28			0.035	0.391	2.083		0.08
Rhabdosargus sarba	6-50	0.243	0.660	0.139	0.043		0.868	0.06
Favonigobius reichei ⌫	5-34		1.250	0.451				0.06
Terapon jarbua	10-50	0.035	0.208	0.035			1.389	0.06
Scomberoides sp.	20-50	0.313	1.007	0.069	0.069	0.130		0.05
Atherinomorus lacunosus	19-50	0.069	1.215					0.04
Periophthalmus koelreuteri africanus	12-26					1.146		0.04
Oligolepis acutipennis	8-42			0.278	0.582	0.208		0.04
Elops machnata	22-37		0.035	0.486	0.139	0.313		0.03
Amblyrhynchotes honckenii	36-50	0.417	0.069				0.347	0.03
Arothron immaculatus	7-24			0.174	0.035	0.573		0.03
*Rhabdosargus globiceps**	8-16	0.764						0.03
Chanos chanos	9-13	0.278	0.069			0.417		0.03
Oxyurichthys spp.	10-17		0.035	0.625	0.069			0.02
*Solea bleekeri**	5-33	0.035		0.486				0.02
*Croilia mossambica** ⌫	5-16	0.035	0.347					0.01
Eleotris spp.	10-16	0.130		0.035		0.208		0.01
Callionymus marleyi	8-50	0.313	0.035					0.01
Hilsa kelee	39-50			0.208	0.069			0.01

Appendix 5 continued...

Appendix 5 continued

Taxon	size range (mm)	sandbank (mouth)	sandbank (basin)	mudflat	*Avicennia* mangrove	*Avicennia* mangrove canal	*Bruguiera* mangrove	% total catch
*Redigobius balteatops**	6-10			0.035	0.069	0.130		0.01
Megalops cyprinoides	19-25			0.035	0.069	0.104		0.01
Caranx spp.	5-42	0.035	0.069			0.104		0.01
*Hyporhamphus capensis**	39-50	0.087	0.069	0.035				0.01
Platycephalus indicus	40-50		0.174					0.01
Crenidens crenidens	8-49	0.069	0.035	0.069				0.01
*Rhabdosargus thorpei**	8-9		0.104		0.058			0.01
Hippichthys spicifer	94					0.104		<0.005
Lutjanus sp. 1	17					0.104		<0.005
Torquigener hypselogeneion	38-50		0.104					<0.005
Pomadasys kaakan	39-50			0.069				<0.005
Omobranchus sp. 1	14				0.069			<0.005
Blenniidae sp. 2	5				0.069			<0.005
Paraplagusia bilineata	34-39	0.069						<0.005
Tetraodontidae sp. 1	4				0.069			<0.005
Dussumierinni sp. 1	34			0.035				<0.005
Spratelloidini sp. 1	15		0.035					<0.005
Spratelloidini sp. 2	14		0.035					<0.005
Lutjanus sp. 2	31			0.035				<0.005
Sparidae sp. 1	12	0.035						<0.005
Acentrogobius audax	13			0.035				<0.005
Bathygobius laddi	7	0.035						<0.005
Glossogobius biocellatus ⌫	16			0.035				<0.005
*Taenioides jacksoni** ⌫	11			0.035				<0.005
Kraemeria samoensis	28	0.035						<0.005
Arothron meleagris	24			0.035				<0.005

Appendix 6

Occurrence of young and small-sized fishes in different habitats within a subtropical South African estuary and adjacent harbour

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Mar. Freshwater Res., 2002, 53, 447–456

Occurrence of young and small-sized fishes in different habitats within a subtropical South African estuary and adjacent harbour

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Abstract. This paper investigates the association of assemblages of 0+ juveniles and small fishes with different shallow-water habitats in Mhlathuze Estuary and the adjacent Richards Bay Harbour. Prior to harbour development, these systems comprised a single large estuarine system. Sampling was conducted on a quarterly basis using a fine-mesh (1 mm) small seine net (8 m). Habitats sampled were all important as nursery areas for juvenile fishes but they supported different fish assemblages. The most distinct assemblage occurred in eelgrass, which is restricted to the estuary. Several species were strongly associated with this habitat, and its scarcity in other subtropical South African estuaries may be limiting to species richness. Mangrove fringes in both systems and sandbanks in the estuary supported similar fish compositions. Fishes associated with mudflats in the estuary and harbour were similar in community structure. Ecologically important prey species used these mangrove, estuary-sandbank and mudflat habitats as juveniles and adults. Harbour sandbanks had fish assemblages dissimilar to sandbanks in the estuary and other habitats in the area.

Extra keywords: postflexion larva, juvenile, community structure, Mhlathuze Estuary, Richards Bay Harbour

Introduction

Habitat degradation is the major factor threatening fishes worldwide (Bruton 1995) and the situation in South African estuaries is no different (Whitfield 1997). Over the past fifty years the most dramatic changes in these estuaries have resulted from human interference (Reddering and Rust 1990). The biological status of many of these systems now indicates degradation, primarily as a result of habitat destruction (Cyrus 1991). KwaZulu-Natal estuaries, in particular, have been adversely affected by sedimentation, eutrophication, bridge and causeway construction, building encroachment and abstraction of freshwater inflow (Forbes *et al.* 1996).

Although it is widely accepted that estuaries worldwide function as nurseries for fishes, Blaber (1997) noted that the concept of estuarine dependence was mainly applicable in temperate regions, where studies have shown many marine species to be dependent on estuaries as juveniles. However, most fishes occurring in subtropical and tropical estuaries are probably not strictly estuarine dependent but rather reliant on the shallow, turbid areas with variable salinity that can be found in coastal habitats outside estuaries in these regions (Blaber 1981, 1997). Most fishes found in South Africa's east-coast estuaries are of tropical or subtropical Indo-Pacific origin (Wallace 1975). Although these taxa may be common in coastal waters in other Indo-Pacific regions, in South Africa they are much more dependent on estuaries. This is because alternative shallow, turbid and brackish waters that provide a nursery role do not exist on the narrow continental shelf (Blaber 1981, 1997). Of the fishes in southern African estuaries, 66% are at least partially dependent on estuaries for their existence (Whitfield 1998).

The continued degradation of estuarine habitat therefore has severe implications for estuarine fishes. This has highlighted the potential importance of harbours, particularly along the KwaZulu-Natal coast, as alternative nursery areas (Cyrus and Forbes 1996; Forbes *et al.* 1996). Important physico-chemical factors influencing fish distribution and abundance within South African estuaries have been identified as turbidity (Cyrus and Blaber 1987*a*, 1987*b*, 1987*c*), salinity and freshwater inflow (Whitfield *et al.* 1981; Whitfield 1994; Ter Morshuizen *et al.* 1996) and salinity–temperature effects (Martin 1988). Some research has compared influences of different habitat structure in terms of reef or submerged aquatic vegetation (Blaber 1978; Branch and Grindley 1979, Beckley 1983, Hanekom and Baird 1984, Whitfield 1986). However, most work has targeted late-juvenile and adult-stage fishes and comparatively little has documented responses of late-larval, or very early-juvenile stages.

Habitat utilization by fishes in size classes <50 mm has not been well studied, even though the majority of marine species recruit into South African estuaries at lengths well below this (Wallace and van der Elst 1975; Whitfield 1998). These early life stages are clearly critical in the life cycles of

10.1071/MF01155 1323-1650/02/020447

estuary-associated fishes and it is important that their habitat requirements are recognized, especially as different juvenile stages may be adapted to very different environmental conditions compared with adults (Day *et al.* 1989). The main elements of habitat structure in subtropical South African systems are substratum type and vegetation. These estuaries are dominated by soft substrata ranging from mud to sand. In the larger systems that remain permanently open to the marine environment, vegetation usually comprises mangroves along the intertidal banks and submerged aquatic macrophytes occur less frequently.

This paper investigates the occurrence of newly recruited, early juvenile and small fishes in shallow-water estuarine habitats in subtropical South Africa and assesses the association between fish assemblages and different habitats in the Mhlathuze Estuary and an adjacent commercial harbour at Richards Bay. Both these systems are marine-dominated embayments (Wepener and Vermeulen 1998) but they exhibit differences in physico-chemical variables. The estuary is subject to a greater freshwater influence than the harbour and is characteristically a shallower and more turbid system. Although freshwater drainage enters the harbour via two canals the greater influence of marine water via the deep, wide mouth largely restricts any effect on salinities and turbidities. With some exceptions, habitats within either system are structurally similar.

Materials and methods

Study area

Mhlathuze Estuary and Richards Bay Harbour are in subtropical KwaZulu-Natal, South Africa. Before harbour development they were a single, large, relatively undisturbed estuarine system. Harbour construction commenced in 1972 with the construction of a berm, or causeway, dividing the system in two. The northern section has been developed into South Africa's largest shipping harbour (Richards Bay Harbour) whilst the southern area was designated as a nature sanctuary (Mhlathuze Estuary) following the dredging of a new river channel and estuary mouth (Fig. 1). The two adjacent bays now function as independent systems with separate mouths ~4 km apart. This study focussed on habitats in these embayment areas. Mangroves dominate the vegetation fringing the estuary and undeveloped shorelines of the harbour. White mangrove, *Avicennia marina*, occurs as a pioneer species on muddy banks of the estuary and the harbour. Black mangrove, *Bruguiera gymnorrhiza*, has also established localized patches on sandy sediments in the harbour.

Only shallow-water habitats (average depth 0.7 m, range 0.4–1.2 m) were sampled during this study because deep-water areas are found only in the major shipping basins and channels. Five sites, representing different habitat in the estuary, and six in the harbour were sampled (Fig. 1). In terms of substratum and physical structure, five habitat types were recognized (Table 1). Three of these, intertidal sandbanks, subtidal mudflats and muddy fringes around *A. marina* mangrove stands, were sampled in both systems. In the harbour a *B. gymnorrhiza* mangrove stand on sand substratum was sampled but this habitat was not matched in the estuary. The eelgrass, *Zostera capensis*, was sampled in the estuary but did not occur in the harbour.

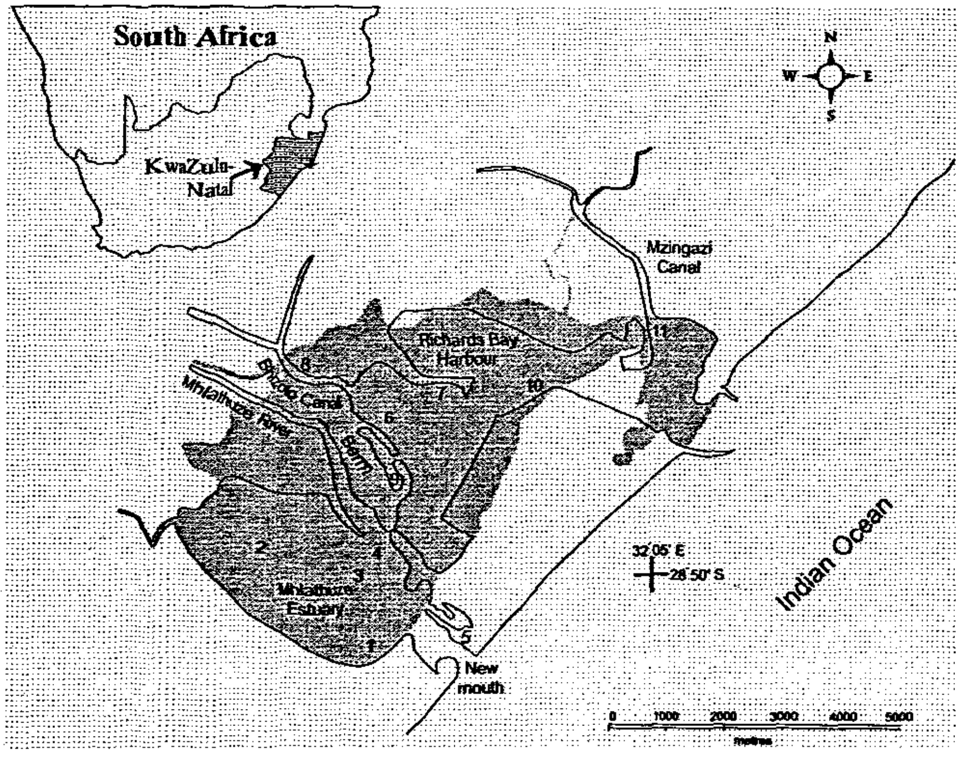

Fig. 1. Location of Mhlathuze Estuary and Richards Bay Harbour and sampling sites. Shaded area indicates Mhlathuze Estuary before harbour development. Site numbers correspond with descriptions in Table 1.

Table 1. Descriptions of sampling sites in Mhlathuze Estuary and Richards Bay Harbour
Physical water variables are averages (range in parenthesis). Site numbers correspond with locations on Fig. 1

Site	System	Habitat structure and substratum		Water	
			Temperature (°C)	Salinity	Turbidity (NTU)
1	Estuary	Eelgrass meadow (*Z. capensis*), sandy mud	23 (18–26)	30 (24–36)	11 (4–18)
2		Subtidal mudflat, mud	23 (17–28)	28 (24–36)	18 (7–35)
3		Intertidal sandbank, sand (main basin, estuarine)	24 (21–28)	34 (32–37)	15 (6–23)
4		Mangrove fringe (*A. marina*), mud	26 (21–30)	18 (2–37)	23 (7–37)
5		Intertidal sandbank, sand (sub-basin, mouth)	25 (20–28)	30 (25–37)	18 (12–23)
6	Harbour	Subtidal mudflat, mud	23 (19–27)	34 (33–37)	14 (7–23)
7		Intertidal sandbank, sand (central sand spit)	23 (21–26)	35 (33–36)	4 (3–6)
8		Mangrove fringe (*A. marina*), mud (Bhizolo canal)	24 (19–28)	32 (30–35)	13 (8–16)
9		Mangrove fringe (*A. marina*), mud (embayment)	25 (21–28)	35 (33–37)	11 (4–17)
10		Mangrove fringe (*B. gymnorrhiza*), sand	24 (20–26)	34 (33–36)	7 (4–14)
11		Intertidal sandbank, sand (Mzingazi canal)	24 (20–27)	29 (23–36)	4 (3–5)

Sampling and analyses

Juvenile and small fishes in the two systems were sampled with a small seine net (1 mm bar mesh, 2.5 m depth, 8 m weighted toe line and 10 m float line). Similar equipment has been found effective in sampling similar habitats in other studies (e.g. Beckley 1983; Laegdsgaard and Johnson 1995). Although it was not possible to seine in intertidal mangrove forests, the net was effectively deployed in open water immediately adjacent to mangroves. Hauls were kept as consistent as possible and measured to allow fish abundances to be determined and expressed as fish per 100 m^2. In this way, meaningful comparisons of fish densities recorded from different sites and habitats could be made. Three replicate hauls were taken a distance apart from one another at each site with the exception of the *Bruguiera* stand and *Avicennia* mangroves in the Bhizolo canal. Steep bank profiles restricted suitable areas for landing nets, and sampling was reduced to a single haul at these sites.

Sampling was conducted in September and December 1997 and in March and June 1998. Fieldwork in the two systems was performed on consecutive days on neap tides, three or four days before the full-moon spring tide. Fishes that could be positively identified and accurately measured to the nearest millimetre (standard length, SL) were recorded and returned to the water. Others were preserved in a 4% buffered formaldehyde solution and returned to the laboratory for identification and measurement. Physico–chemical variables (salinity, temperature and depth) were recorded at each site with a Hydrolab Datasonde 3 Water Quality Multiprobe. Turbidity samples were returned to the laboratory and analysed with a Hach Model 2100A Turbidimeter.

For the purposes of this study, only fish up to 50 mm SL were considered with the exception of two pipefish species (Syngnathidae) and an eel (Anguillidae). Owing to their elongate body shapes these species were considered up to 100 mm SL. The small-mesh seine net was considered to be effective in sampling these size classes of fish, which included juveniles of all species as well as adults of some smaller-bodied species such as gobiids and ambassids. In some cases, species were pooled at the genus level because of the morphological similarity of their early juvenile stages. Mullet were pooled at the family level because of the difficulty in distinguishing small specimens, even to the level of genus. Each taxon was categorized according to its association with estuaries in southern Africa by use of the classification lists of Whitfield (1998):

I Estuarine species that breed in estuaries;

II Euryhaline marine species that breed at sea but with juveniles that show varying degrees of dependence on estuaries;

III Marine species that occur in estuaries in small numbers but are not dependent on these systems (marine stragglers);

IV Euryhaline freshwater species, including some that may breed in both freshwater and estuarine environments (freshwater migrants);

V Catadromous species that use estuaries as transit routes between the marine and freshwater environments (catadromous migrants).

Analysis of variance (ANOVA) was performed on water variables and univariate community indices of number of species (S), abundance (N), Margalef's species richness ($d = (S - 1)/\log N$) and Shannon–Wiener diversity ($H' = -\Sigma_i\ p_i(\log_e\ p_i)$). Data were transformed to meet the assumptions of normality and equal variance where necessary. Multiple range tests (Tukey's honest significant differences) were used to make comparisons between habitats and determine which means were statistically different ($P < 0.05$). Multivariate analyses were conducted using the computer software package PRIMER v5 and included hierarchical clustering (Bray–Curtis) and ordination (non-metric multi-dimensional scaling, nMDS). Tests for differences in structure and composition of assemblages used analysis of similarities (ANOSIM). This procedure uses ranked Bray–Curtis similarities to compute a statistical test of the validity of assemblage differences among groups of samples specified *a priori* (Clarke and Warwick 1994). Multivariate statistical analyses used 4th-root transformed data and a significance level (P) of <0.05.

Results

Physical water characteristics varied between systems and among habitats and months sampled (Table 2). Water temperatures reflected seasonal changes in sea surface water temperature and ambient air temperatures. Salinities at the mangrove fringes in the estuary were significantly lower than at other sites in both systems with the exception of the estuary mudflat and Mzingazi sandbank. On average, salinities were lower in the estuary than the harbour. They also varied significantly with month, being lowest in September 1997 (estuary average 21.6, harbour average 30.3) and highest during the period of reduced rainfall in June 1998 (36.6 and 35.9 respectively). Salinities as low as 2.1 and 5.4 were recorded along estuary mangrove fringes in September and December 1997 respectively. These were unusual because recordings of <20 were not made at any other time. In the harbour, salinities below that of marine water were recorded only at the Bhizolo mangroves (September 29.5, December 30.9) and Mzingazi sandbank (September 22.9, December

Table 2. F-ratios and significance levels for analysis of variance of physical water variables

F-ratios based on the residual mean square error. df, degrees of freedom; *P <0.05; **P <0.01; ***P <0.001; ns, not significant

	Effects		
	System (1 df)	Site (9 df)	Month (3 df)
Temperature	65.258*	3.746**	124.271***
Salinity	12.684**	3.346**	8.801***
Turbidity	18.761***	2.069$^{\text{ns}}$	4.196*

24.5). Although turbidities did not differ significantly among sites they were significantly higher in the estuary (average 17 NTU) than in the harbour (average 9 NTU), reflecting a greater area of deep water and marine influence in the harbour.

Totals of 9185 and 9030 fishes ≤50 mm SL were caught in Mhlathuze Estuary and Richards Bay Harbour respectively. Seventy-three different fish taxa were sampled, 65 from the estuary and 43 from the harbour. Marine spawners were present as recently settled postflexion larvae and 0+ juveniles. Resident species that breed in southern African estuaries were represented by postflexion larval to adult stages. Ladyfish *Elops machnata* (Forsskål 1775), oxeye tarpon *Megalops cyprinoides* (Broussonet 1782) and freshwater eel *Anguilla* sp. were recorded as leptocephalus larvae but no preflexion larval stages were sampled.

Grey mullet (Mugilidae) and glassies (Ambassidae) were the most abundant fishes caught and together constituted 44% of the fishes recorded in the estuary and 84% of those sampled in the harbour. They were most abundant along mangrove fringes and over unvegetated sand substrata in both systems. Marked abundance peaks were recorded in the harbour in September 1997 and June 1998 when mullet between 10 and 20 mm and ambassids of a range of sizes were caught. In the estuary, peaks in density were much less pronounced and generally occurred in December 1997, when mullet between 20 and 30 mm and an abundance of newly recruited ambassids <10 mm were present. The sparid *Rhabdosargus holubi* (Steindachner 1881) was the third most abundant species in the estuary and as newly recruited postflexion larvae occurred almost exclusively in the *Zostera* beds in September and December 1997. Fewer, but larger specimens occurred over estuarine sandbanks. In the harbour, *Stolephorus* spp. (Engraulidae) were the third most abundant fish and mostly used mudflats.

Over 95% of fishes (by abundance) in both systems were species known to have strong estuarine associations (category I and II species, Whitfield 1998). In the estuary 31% of species and 36% of individuals were estuarine residents (primarily over unvegetated sand substrata) compared with 28% of species and 20% of individuals in the harbour (primarily along mangrove fringes) (Fig. 2). Conversely, proportionally more euryhaline marine species (47%), and individuals (76%), were found in the harbour than in the estuary (42% and 60% respectively). In Mhlathuze Estuary, marine stragglers (category III) were encountered to a notable degree (>10%) only in the *Zostera*, mostly in March and June 1998. They were represented mainly by an abundance of small (<25 mm) blackedged puffer *Arothron immaculatus* (Bloch & Schneider 1801), a species commonly encountered in weedy areas of subtropical South African estuaries (Whitfield 1998). In the harbour, marine stragglers were abundant only in September and December 1997. Recently recruited piggy *Pomadasys olivaceum* (Day 1875) (<20 mm) were sampled over sandy

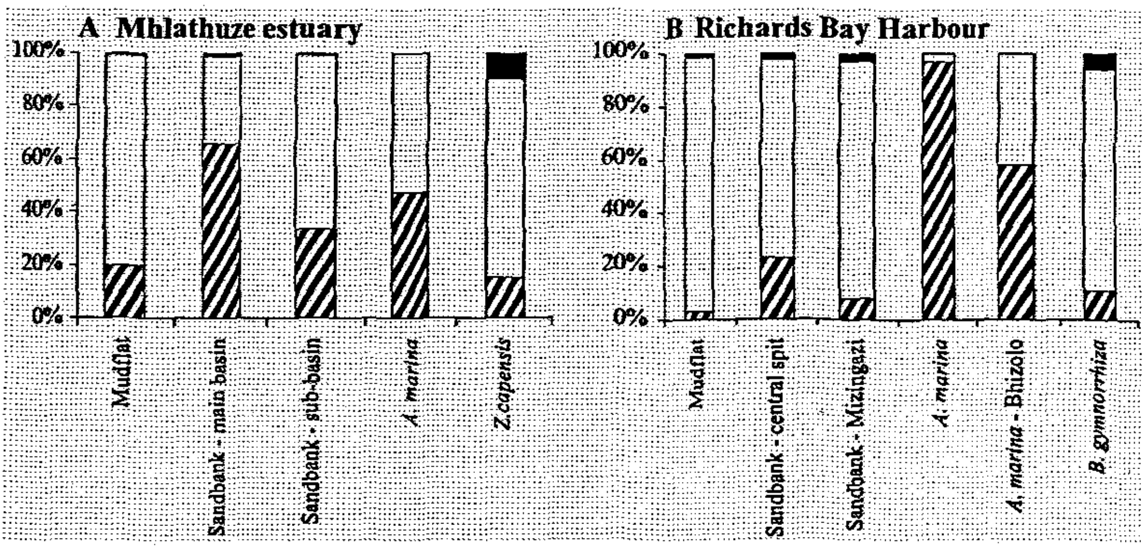

Fig. 2. Average relative contributions of different estuarine association categories to total fish abundance in *(A)* Mhlathuze Estuary and *(B)* Richards Bay Harbour. Categories: I Estuarine breeders and residents (hatched columns); II Euryhaline marine species (white); III Marine stragglers (black).

substrata at both the central spit and Mzingazi canal in September and the clupeid *Herklotsichthys quadrimaculatus* (Rüppell 1837) was abundant at the *Bruguiera* mangrove fringe in December 1997.

Numbers of species associated with different habitats (averaged across quarterly samples) were higher in the estuary than in the harbour (Table 3) and most notably in the *Zostera* bed (Fig. 3). Average abundance did not differ

Table 3. F-ratios and significance levels for analysis of variance of selected univariate indices of fish assemblages
F-ratios based on the residual mean square error. df, degrees of freedom; $^{*}P < 0.05$; $^{**}P < 0.01$; $^{***}P < 0.001$; ns, not significant

	Effects		
	System (1 df)	Site (9 df)	Month (3 df)
Species number ($\log_{10}$)	63.514^{***}	4.771^{***}	3.397^{*}
Abundance ($\log_{10}$)	0.066^{ns}	1.904^{ns}	1.020^{ns}
Species richness ($\log_{10}$)	43.512^{***}	5.252^{***}	1.274^{ns}
Species diversity	12.019^{**}	0.494^{ns}	0.494^{ns}

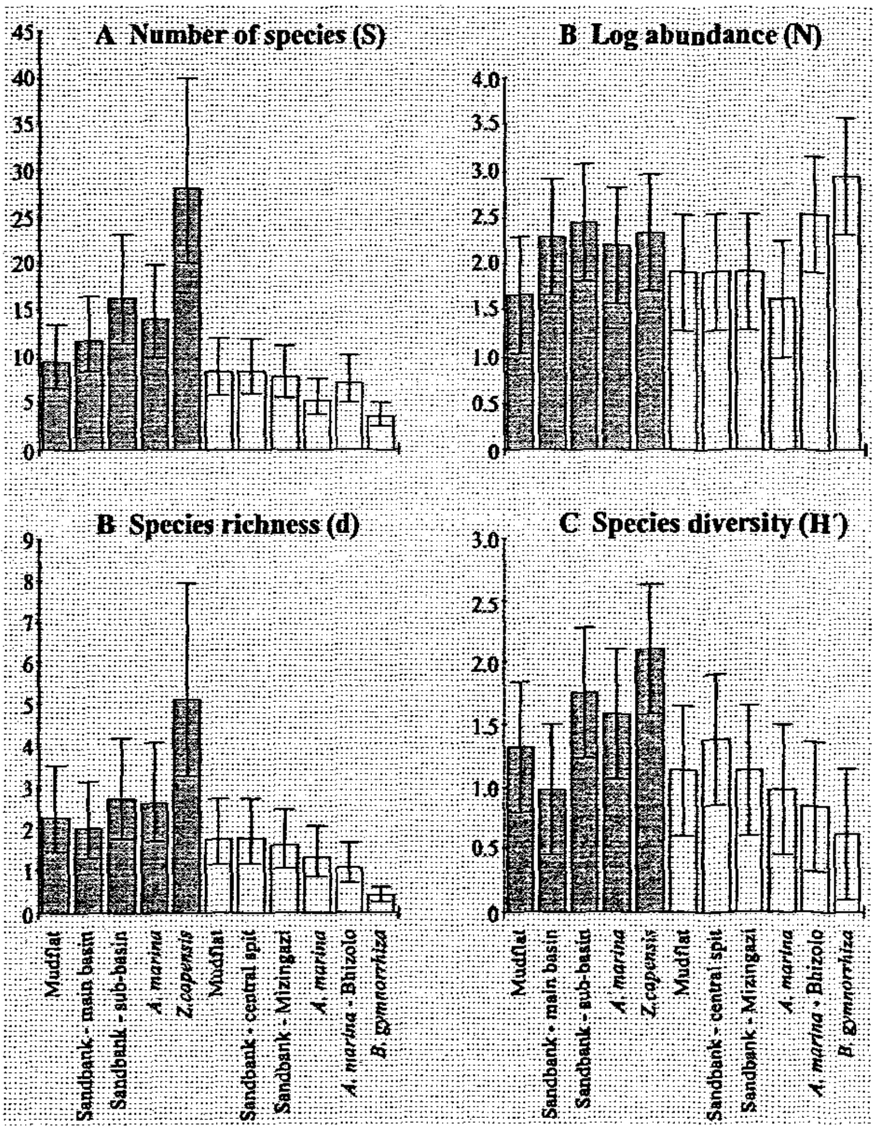

Fig. 3. Comparisons of means (with 95% confidence limits) of selected community indices from different habitats in Mhlathuze Estuary (shaded) and Richards Bay Harbour (not shaded): *(A)* number of species, *(B)* log total abundance, *(C)* Margalef's species richness and *(D)* Shannon–Wiener diversity.

significantly, either between systems or among habitats sampled. Large variance in densities of fishes sampled at the *Bruguiera* mangrove stand in the harbour (Fig. 3) was the result of very high abundances of mullet in September and June 1998. Species richness and diversity were both higher in Mhlathuze Estuary than in Richards Bay Harbour, but only richness differed significantly across habitats (Table 3, Fig. 3).

Multivariate analyses of transformed fish densities from each month indicated distinct fish communities in different habitats. Analysis of all data from each system permitted these differences to be investigated over the four months sampled. In the estuary, the assemblages of fishes varied significantly among habitats (averaged across months) and also over the months sampled (averaged across habitat groups). Similarities within habitat groups were best illustrated in the cluster dendrogram (Fig. 4*a*). In most cases the effect of month accounted for the relatively low similarity in the major groupings. For instance, *Zostera* supported a distinct fish community over the year, but within this broadly similar fish assemblage distinct temporal differences were apparent. In September and December 1997, similar communities were present which were distinct from those recorded in March and June 1998. Analysis of similarity showed that the greatest overlap in catches from any two habitats was recorded between the sandbank in the sub-basin and mangrove-fringe habitat.

The fish fauna from the harbour mudflat was distinct from those at other harbour habitats (Fig. 4*b*). Assemblages associated with different mangrove habitats did not differ significantly. Monthly differences across averaged habitat data were significant and were prominent in sand-associated fish assemblages. Replicates from the two sand sites fell within similar groups influenced by month rather than by location.

Similarities among averaged replicates from both systems are indicated in Fig. 4*c*. A distinct fish fauna was associated with *Zostera* habitat whilst broadly similar assemblages were associated with estuarine sandbanks and mangrove fringes. This assemblage of fishes associated with estuarine sandbanks and mangrove fringes was most similar to fishes associated with different mangrove habitats in the harbour. Mudflats in the estuary and the harbour supported similar fish communities, and fishes associated with harbour sandbanks were most often distinct from those associated with other habitats from both systems.

Discussion

Although freshwater influence was detectable in both Mhlathuze Estuary and Richards Bay Harbour, they can both be regarded as marine-dominated embayments. South African estuarine fishes are generally more tolerant of low than of high salinities, and ranges reported in the systems studied were within the tolerance limits of most estuarine and marine-spawning species present in KwaZulu-Natal systems (Whitfield *et al.* 1981). Similarly, water temperatures were typical of the ranges that occur seasonally in KwaZulu-Natal estuaries (Cyrus 1988) and within the tolerance limits of the juvenile fishes present (Cyrus and Blaber 1987*b*). Turbidities at which changes in juvenile fish assemblages from KwaZulu-Natal estuaries are most obvious are 10, 50 and 80 NTU (Cyrus and Blaber 1987*a*, 1987*b*). In a regional context, the Mhlathuze is a semi-turbid estuary (mean turbidity within the range of 10–50 NTU) whereas the harbour is a clear system (mean turbidity <10 NTU).

The small-mesh seine net was effective in sampling the habitats investigated for fishes ranging from 5 to 50 mm. Species sampled included many taxa commonly encountered in the subtropical estuaries of southern Africa. The most abundantly occurring fishes in both systems were mullet (0+ juveniles) and ambassids (postflexion larvae to adults). These species are the most abundant fishes in many systems of KwaZulu-Natal as juveniles (and adults in the case of ambassids) (Wallace 1975) but are rare (contributing <2%) as recruiting larvae at the mouths of major systems along the coast (Kosi Bay, Harris *et al.* 1995; St Lucia Estuary, Harris and Cyrus 1995; Richards Bay, Harris and Cyrus 1997; Mhlathuze Estuary, Viljoen and Cyrus 2002; Durban Harbour, Harris and Cyrus 1999). Common species of mullet recruit most abundantly into KwaZulu-Natal estuaries as young juveniles rather than as early postlarvae (Wallace and van der Elst 1975), and standard ichthyoplankton sampling techniques probably underestimate recruitment by these fishes. Ambassids in southern Africa can spawn in estuaries (Whitfield 1998) and may develop without an obligatory marine phase. If eggs and larvae are retained in an estuary, recruitment via the mouth from marine waters need not occur. This may account for the apparent abundance of juveniles and adults in KwaZulu-Natal estuaries despite the low recruitment recorded in the studies cited above.

Salinity preferences of local freshwater species prohibited their use of either Mhlathuze Estuary or Richards Bay Harbour. Most fishes recorded in this study were species typically associated with estuaries, and the relative lack of marine stragglers is a reflection of the estuarine character of the systems studied, if not in the physical and chemical properties of the water then in the nature of the habitat. In the case of Richards Bay Harbour this indicated the importance of shipping ports in KwaZulu-Natal as nurseries for estuarine fishes, a role previously identified by Cyrus and Forbes (1996) and Forbes *et al.* (1996). Moreover, with physical properties of harbour waters being marine in nature, it indicated the importance of calm, shallow water with varied substrata and structure in habitat. The larger proportion of estuarine residents in the estuary (individuals and species) suggested that mixing of greater volumes of

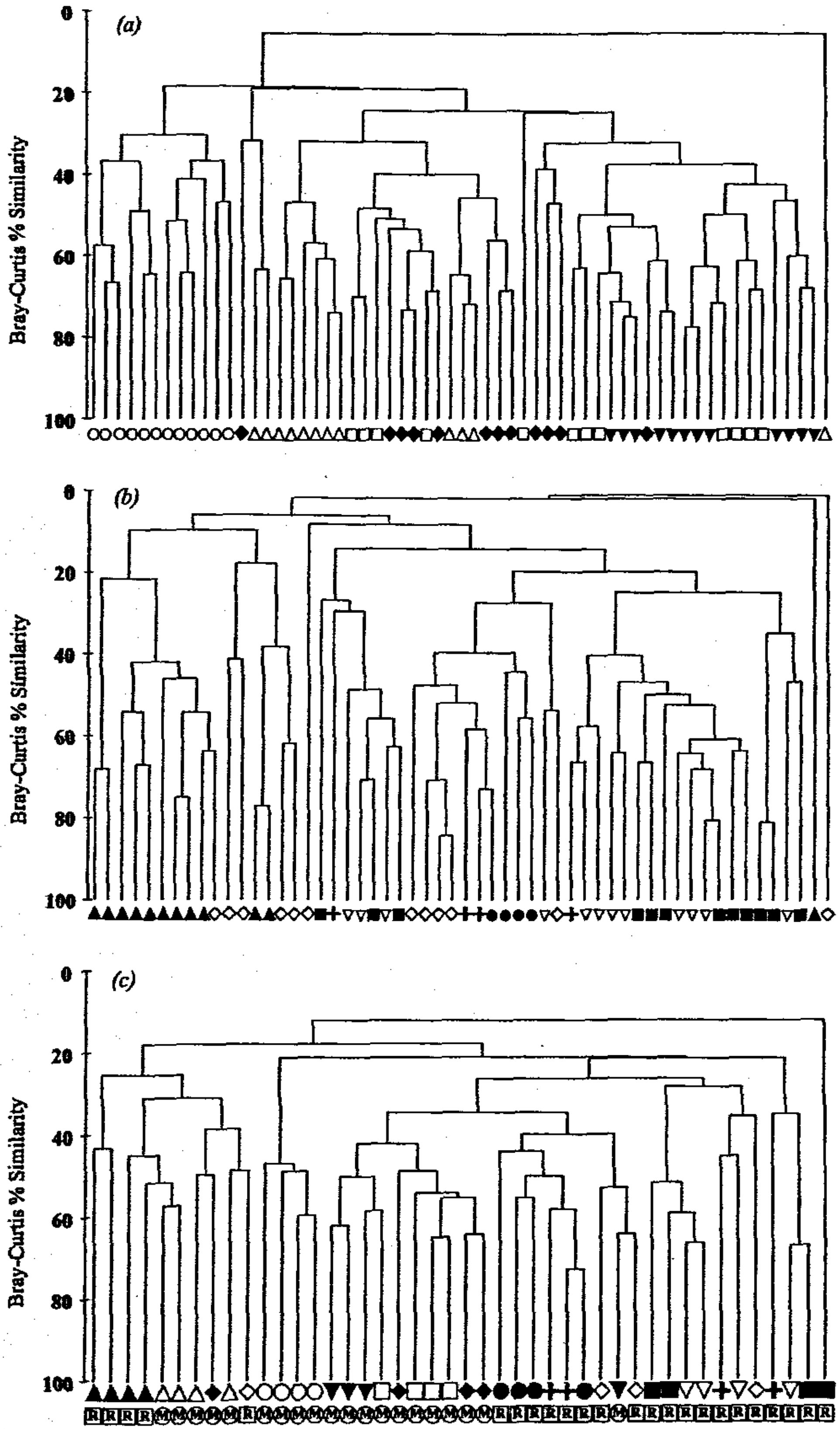

Fig. 4. Bray–Curtis similarity dendrograms: *(a)* replicated hauls taken at different habitats during all months in Mhlathuze Estuary. *(b)* replicated hauls taken at different habitats during all months in Richards Bay Harbour. *(c)* hauls (replicates averaged) taken at different habitats during all months in Mhlathuze Estuary and Richards Bay Harbour. **R, Richards Bay Harbour**: ▲, mudflat; ▽, sandbank, central spit; ■, sandbank, Mzingazi; ◇, *A. marina*; +, *B. gymnorrhiza*; ●, *A. marina*, Bhizolo. **M, Mhlathuze Estuary**: △, mudflat; ▼, sandbank, main basin; □, sandbank, sub-basin; ◆, *A. marina*; ○, *Z. capensis*.

freshwater with lower volumes of marine water was influential in determining species assemblages.

The higher number of species recorded in Mhlathuze Estuary was largely due to the presence of eelgrass in this system. Of the 65 species recorded in the estuary, 49 were associated with *Zostera* habitat. Fifteen of these, including *Hippichthys cyanospilos*, *Sebastapistes strongia*, *Epinephelus malabaricus*, *Pelates quadrilineatus*, *Siganus sutor*, *Lethrinus* sp. and Labridae and Scaridae species, were not recorded from any other habitat in both systems. Others, although they occurred in non-*Zostera* habitats, were found in highest densities in the eelgrass (e.g. 84% of all *R. holubi* were recorded in the *Zostera*). The species composition associated with *Zostera* was therefore unique within the estuary and in a more regional context when the fishes of the harbour were also considered. Pollard (1984) noted from studies worldwide that there was a common suite of fish families that use seagrass habitat; many of these were sampled in the *Zostera* bed during this study. Seven of the ten dominant fish families found in Australian seagrass habitats (Pollard 1984) were recorded here: Syngnathidae, Gobiidae, Sparidae, Teraponidae, Apogonidae, Tetraodontidae and Ambassidae. In addition to estuarine fish species, some taxa more typically associated with reef habitat were also recorded from *Zostera* in this study. This has been noted in seagrasses elsewhere (Nagelkerken *et al.* 2000).

The importance of seagrass habitats as fish nurseries, and higher numbers of species and individuals in their associated fish faunas compared with those of unvegetated habitats, has been noted from many studies (Branch and Grindley 1979; Beckley 1983; Heck *et al.* 1989; Whitfield *et al.* 1989; Ferrell and Bell 1991). There are exceptions but they are generally few (Pollard 1984). In the Mhlathuze Estuary, eelgrass is an important nursery habitat to many fish species, but total fish abundance was not significantly higher than at other habitats. Eelgrass habitat is rare in KwaZulu-Natal and its absence from other systems probably limits fish species richness. Indications are that in Mhlathuze Estuary *Zostera* is sensitive to environmental perturbations, particularly deposition of fine sediments.

Mangroves are also widely considered to fulfil an important role as nursery to diverse fish communities. In tropical Australia, Robertson and Duke (1987) found mangroves to sustain much higher fish densities than nearby seagrass habitat. Similarly, Laegdsgaard and Johnson (1995) concluded that although seagrass habitats in subtropical Australia have a nursery function for fishes, the importance of this role is reduced when compared with the gross abundances of juveniles and number of economically important species found in mangrove habitats. On the basis of a limited data set, this was found not to be the case in the subtropical South African systems studied here. In the estuary, fish species richness was lower among mangroves than in eelgrass habitat, and abundance, richness and diversity were similar in mangroves to those in other habitats. In the harbour, species numbers and richness indices were often lowest for fish communities associated with mangroves.

Similar fish faunas associate with muddy *Avicennia* fringes and sand substrata adjacent to *Bruguiera* mangroves in Richards Bay Harbour. Blaber and Milton (1990) found fish faunas to differ between soft-substratum *Rhizophora* mangrove estuaries and hard-substratum *Bruguiera* systems in the Solomon Islands. These results were based on comparisons of whole systems characterized by different mangrove species and substrata rather than different sites within the same system. *Avicennia marina* dominates the species composition of mangroves in Richards Bay Harbour and occurs as large monospecific stands. This may influence the mangrove fish fauna of the system as a whole.

Similar reasoning may explain the similarity in fishes associated with estuarine sandbanks and mangrove fringes. The area covered by mangroves at Mhlathuze Estuary is even greater than that at Richards Bay Harbour. There are few natural shoreline reaches that are not lined with *A. marina*, and where they do occur they are short. Sandbanks in the harbour supported more distinct small-fish assemblages, possibly because they were a greater distance away from mangroves than those in the estuary. Ferrell and Bell (1991) recorded differences among assemblages of fish over bare sand depending on the proximity of seagrass beds, and a similar situation may exist here with respect to mangroves. In comparison, fish assemblages from open-water habitats over muddy substrata from both systems were similar to one another irrespective of differences in their distance from mangrove fringes (although the possibility of the influence of distance cannot be discounted from results here).

At the broadest level, the juvenile and small-bodied fishes of the embayment areas of Mhlathuze Estuary and Richards Bay Harbour fall into four categories determined largely by habitat. *Zostera capensis* beds in the estuary support a locally unique assemblage and there is no match for this habitat in the harbour. Sparids, especially of the genus *Rhabdosargus*, rely heavily upon this habitat as a nursery soon after recruiting into the system, but apparently move to open sand habitat at later juvenile stages. Several other species use this habitat exclusively as 0+ juveniles before emigrating, possible back into the marine environment. Fewer species associate with mangroves from both systems, together with sandbanks in the estuary. These habitats support the majority of juvenile mullet, and juvenile and adult ambassids occurring in the two systems. These are the most frequently and abundantly occurring fishes in KwaZulu-Natal estuaries and are important as prey species for other fishes as well as avifauna. Mudflat assemblages, similar in both systems, also consisted of important prey fishes in the form of *Stolephorus* spp. and *Thryssa* spp., as well as more benthic groups including gobies and blackhand sole *Solea bleekeri* (Boulenger 1898). A final category of fishes is associated with sandbanks away from mangrove stands in the harbour and comprises largely silver sillago *Sillago sihama* (Forsskål 1775) and pursemouth *Gerres* spp. as postflexion larvae and young juveniles, ambassids and to a lesser degree spotted grunter *Pomadasys commersonnii* (Lacepède 1801). The sand dragonet *Callionymus marleyi* (Regan 1919) was recorded only at these two sand sites, although generally in low abundances and not consistently over the months sampled.

Habitats in these two adjacent systems are used by juveniles of many marine-spawning estuarine fishes, including important linefish species, as well as by juveniles and adults of small but ecologically important species that breed in both the marine and estuarine environments. The degree of estuarine dependence of these fishes varies from one species to another. There is a need to address conservation issues pertaining to estuaries and their fish faunas in South Africa (Whitfield 1997). Habitat protection is clearly a relevant issue and this study indicated the need for system protection, which accounts for habitat preferences of different fishes. Important in this regard in the Mhlathuze Estuary are areas of *Zostera*. The estuary is managed as a Marine Protected Area but, as with all estuaries, it is an open system. The most serious threats arise beyond its immediate boundaries. Effects of poor catchment management are introduced via inflowing river waters, and dredged harbour-spoil material, pumped onto a nearby beach and into the surf zone, has been noted to intrude via the mouth. These are serious concerns because *Zostera* is likely to be the habitat most negatively influenced by deposition of fine sediments. Sandbanks in the harbour are locally important because they support fish assemblages not associated with other habitats in either system. Being isolated from any major river system, habitats in Richards Bay Harbour are more protected from the catchment degradation effects common in most KwaZulu-Natal estuaries. Here, more immediate threats arise from ongoing harbour development and activities related to shipping and handling of bulk cargoes.

Acknowledgments

We thank personnel of the Coastal Research Unit of Zululand for help in the field and Fiona MacKay for assistance with statistical procedures and data presentation. Portnet, the University of Zululand Research Committee and the Coastal Research Unit of Zululand assisted in financing this research.

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Manuscript received 21 May 2001; revised and accepted 9 December 2001