THE MACROBENTHOS OF THE MLALAZI ESTUARY: KWAZULU-NATAL

By

Samkelisiwe Hlengiwe Mabaso

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Supervisor: **Mr L. Vivier**
Co-supervisor: **Dr. R. K. Owen**

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DECLARATION

I hereby declare that this study contains original work and has not been submitted in any form for any degree at another university. The work of others used in this study has been duly acknowledged in the text.

S.H.P Mabaso

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ABSTRACT

The Mlalazi estuary situated in the Mlalazi Nature Reserve is regarded as one of the least spoiled estuaries on the KwaZulu-Natal coast, with the relatively unspoilt character being enhanced by the location of the estuary in a marine protected area (MPA). Available information of the biota of the estuary is, however, limited to one ecological study undertaken by Hill (1966). A survey, conducted by the Coastal Research Unit of Zululand for the purpose of a masters study, of the benthic community was initiated during the period 1989-1991 but unfortunately was never written up or published. Additional sampling in the Mlalazi estuary was carried out from August 1999 to July 2000 at the same six sites where the 1989-1991 samples were collected, with the addition of site 7 in the mouth region.

A prawn farm was established in 1992 approximately 500 m from and adjacent to the middle reaches of the estuary. Concern has been expressed about water being discharged into the estuary from the prawn farm since this represents a source of anthropogenic disturbance.

This study focused on describing the water quality and macrobenthos of the Mlalazi estuary during the 1989-1991 and 1999-2000 sampling periods. The community structure, temporal and spatial abundance patterns of the two sampling periods were examined. A relationship between the observed community patterns and environmental variables was also determined to find the possible causes for the observed abundance patterns. Since no study has been done on the potential impact of the prawn farm on the biotic community of the estuary, comparison of the 1989-1991 (pre-prawn farm period) and 1999-2000 (post prawn farm period) sampling periods attempted to correlate the prawn farm activities to changes in community patterns of the two periods.

The physico-chemical results indicated high concentrations of nutrients (nitrates, nitrites, orthophosphates, total phosphates) during the two sampling periods. Since there were no major industries and the prawn farm was not yet established during the

1989-1991 sampling period, high nutrient concentrations were suspected to be due to agricultural runoff. The high nutrient load during the 1999-2000 sampling period indicated the prawn farm as the source. This is because high concentration of these nutrients were recorded at the prawn farm outlet and at Site 3, which is the site closest to the prawn farm outlet, compared to the rest of the sites.

The results of the biotic analysis indicated that the biotic community of both periods was disturbed to some degree with the 1989-1991 period indicating that it was more disturbed than the 1999-2000 sampling period. The 1989-1991 period was characterised by a reduced number of taxa (28) and was dominated by highly abundant opportunistic species such as *Prionospio* and capitellid polychaetes. The 1999-2000 period recorded 36 taxa but was also dominated by highly abundant *Prionospio spp*.

Multivariate analysis (classification and ordination) indicated the importance of environmental factors in determining the structure of biotic communities for both the sampling periods. The significant variable accounting for the observed distributional patterns of the biotic community in 1989-1991 was the median particle diameter (p_w = 0.505) whereas in the 1999-2000 sampling period it was salinity (p_w = 0.292). However a combination of environmental factors played an important role in determining the observed biotic patterns. A combination of salinity, oxygen, turbidity, depth and median particle size was responsible for distribution of the benthic fauna during the 1989-1991 period. During the 1999-2000 sampling period, salinity and organic content formed a combination most responsible for the biotic patterns observed.

Even though the above results indicated that organism abundance and distribution were largely determined by physical factors, the chemical results did indicate that large amounts of nutrients in the estuary are being discharged from the prawn farm. The impact on the estuary and the organisms inhabiting it may not be apparent at this stage but there is a concern that continuous discharge of such a high nutrient load may affect the estuary in future. A monitoring program should therefore be initiated with this study being used as a baseline from which references can be made.

CHAPTER 1

The Macrobenthos of the Mlalazi Estuary, KwaZulu-Natal

1.1. Introduction

1.1.1. General

Estuaries are the meeting places of rivers and the sea and their character is determined by this interplay (Branch & Branch 1981). However the widely accepted definition of an estuary in South Africa given by Day (1980) is more complex than merely the interplay of rivers and the sea, and states that an estuary is "a partially enclosed coastal body of water which is either permanently or periodically open to the sea and within which there is a measurable variation of salinity due to the mixture of sea water with freshwater derived from land drainage".

Estuaries are highly productive systems, which enhance their suitability as nursery habitats for many juvenile and adult organisms. Estuaries act as nutrient traps, with nutrients being derived from both the rivers and sea and rank among the most productive environments on earth (Branch & Branch 1981, Day 1981). This productivity is derived from the input and breakdown of vegetation and nutrients from the river and the sea. The banks of the estuary itself also provide emergent plants, such as saltmarsh and mangrove vegetation, thus increasing the detrital input. All this contributes to the accumulated detritus, which is broken down by bacteria and utilised by aquatic organisms (Day 1981).

The fluctuating physico-chemical conditions of estuaries limit the diversity of organisms, with few organisms able to adapt to life in estuaries. This has resulted in a generally low diversity. In addition, natural (episodic) and human induced activities in estuaries also play a significant role in the instability of these environments, with corresponding decreases in abundance and diversity. However, fauna has adapted to the fluctuating conditions characteristic of estuaries.

The Mlalazi estuary (28^0 57'S, 31^0 48'E) is one of 73 estuaries (Begg 1978, Whitfield 1992) situated along the KwaZulu-Natal coast, which stretches some 560 km from the Mozambique border north of Kosi Bay to the Mtamvuna River south of Port Edward. The Mlalazi estuary is situated in the Mlalazi Nature Reserve and is regarded as one of the least spoiled estuaries in the KwaZulu-Natal (Mann, *et. al* 1996). Its relatively unspoilt character is enhanced by its location in a marine protected area (MPA). There is, however, a lack of information available on biota of the Mlalazi estuary, which is limited to one ecological survey by Hill (1966) that recorded the physical conditions and fauna of the estuary.

Benthos consists of those organisms found on or in the bottom sediment of the aquatic body. The benthos can be conveniently subdivided into size groups (Day 1981). Those organisms that pass through the mesh size of 0.1 mm are termed microbenthos, e.g. protozoa, while those that are retained by a mesh of 0.1 mm are termed meiobenthos, e.g. nematodes. The macrobenthos are those organisms that are retained in a sieve with a mesh of 0.5 mm, e.g. molluscs. Benthic organisms are amongst the most sensitive faunal components of estuaries and as such are useful in indicating the status of the aquatic body (Dauer 1993, Mackay & Cyrus 1999). They are largely sedentary, have relatively long life cycles and consist of different species that exhibit different tolerances to stress (Day, *et. al* 1989).

1.1.2. <u>Stability and diversity of benthic communities</u>

Estuaries are characterised by fluctuations in physical and chemical factors, leading to the instability of the environment and reduced diversity of organisms. The stability of a community is generally defined as its ability to return to an equilibrium state after being disturbed and is interpreted on the basis of patterns in numerical abundance and biomass evident over time (Gray 1981). Two general patterns of community structure emerge with regard to stability. Some populations or communities maintain constant numbers through time, i.e. they are persistent. The other pattern as found in benthic species, is that of repeatable cycles, either annual or longer term. If the cycles are predictable, the community is considered stable (Gray 1981).

Community ecology recognises two theoretical levels of stability. The kind of stability where the community is resistant to small changes, but where shifts in species dominance produces a new state of equilibrium, is called neighbourhood or local stability. The alternative to neighbourhood stability is called global stability where the community always returns to the same equilibrium point no matter how large the disturbance (Gray 1981). This means that the community always returns to the same equilibrium point with the same species dominating. However, benthic communities may shift from one local stability state to another as a result of changes in environmental factors, such as change in salinity due to floods, competition for space or predation. In the case of global stability, a very large disturbance may cause the community to shift from one global state to another, with the community not returning to its original species composition. The difference between neighbourhood and global stability is therefore a question of the scale of disturbance.

There is also a concept of relative stability which considers the 'distance' moved by the community from its equilibrium and the time required for the community to return to that equilibrium. Relative stability includes the concepts of resistance and resilience. Resistance refers to the ability of a community to withstand disturbances, while a resilient community returns rapidly and directly to its original species composition after being disturbed.

Although estuaries are highly productive systems (Branch & Branch 1981, Day 1981), the variability of environmental conditions imposes stresses which severely limit the diversity of organisms in such a rich environment. Much ecological discussion has centred on the relationship between diversity and stability (Odum 1971, Begon, *et. al* 1986). Prior to 1974, it was commonly held that more diverse communities were generally more stable (Odum 1971). This leads to suggest that high diversity is a property of stable natural systems. However, recent studies have shown that there is not necessarily a link between high diversity and high stability (Gray 1981). May (1974) showed that increased diversity in a variety of systems led to a more fragile community structure in a stable environment while an unstable environment resulted in persistent robust communities which were not as diverse. Loss of diversity in aquatic environments has received little attention even though the physical, chemical and biological degradation of aquatic systems is widely recognised as a major problem usually in the context of loss of fisheries and water quality (Day 1981, Novotny & Olem 1994).

There are several reasons why estuaries have an inherently low diversity. The fluctuating conditions due to physico-chemical factors result in instability of the estuarine environment with few species being adapted to such changing conditions. These include the complex properties of water itself, the interactions between aquatic and terrestrial environments and the proximity of human population to aquatic systems (Fiedler & Jain 1992). The effects of terrestrial changes on estuarine systems are quite severe in areas where human populations are dense as urbanisation and agriculture developments impact on estuarine communities. For example, sewage discharged into aquatic environments frequently leads to eutrophication which, in extreme cases, leads to a lack of oxygen and the presence of hydrogen sulphide in the sediment, with a corresponding decrease in the diversity of fauna. It is an indication of their resilience that many of the estuaries still contain rich biota.

Many degraded estuaries have the potential to be at least partially restored to more natural conditions. Guidelines for selection, establishment and management of aquatic environments to conserve biodiversity have been developed (EPA 1992, DWAF 1996). However, as far as environmental impact is concerned, much remains to be done since each aquatic system combines ecological factors in a different way, and every proposed development requires research aimed at answering specific questions relevant to each system.

1.1.3. **Factors affecting estuarine benthic communities**

Spatial and temporal variations in physical and chemical conditions are characteristic features of the estuarine environment which demand special adaptations in organisms that inhabit such systems, as these conditions affect the physiological processes of an organism (Barnes 1974). The main physico-chemical factors that affect estuarine organisms are salinity, temperature, oxygen levels, the nature of the substratum and the availability of nutrients (Barnes 1974). Man's activities, through the introduction of domestic and industrial wastes and toxins, can also affect the community structure of organisms. No less important, but often less obvious, are the biological factors such as predation and competition which affect the reproduction, recruitment and survival of organisms (Knox 1986). Estuarine species occur along environmental and biological gradients with each species having an optimum niche along a gradient. Therefore many factors such as temperature, salinity, nature of substratum, currents and pH are responsible for the variations in benthic abundance and distribution.

1.1.4. **Disturbance of estuaries**

The land adjacent to many estuaries has been used extensively for residential and agricultural purposes as well as for establishment of industrial complexes in South Africa. This has led to estuaries being degraded through destruction of surrounding saltmarsh vegetation, increased siltation, pollution and increased utilisation by man (Day 1981).

There is evidence from historical records (Day 1980, Cooper, *et. al* 1993) that the rate of siltation in estuaries has increased with more intensive cultivation on the banks and the draining of swamps. However, estuaries rely on the input of silt and detritus for their natural functioning, but excessive input of silt smothers the estuarine fauna and causes shallowing, reduction of water volume and loss of habitat, which potentially reduce the estuary nursery function (Cooper, *et al* 1993). Begg (1978) noted that many of estuaries along the KwaZulu-Natal coast have been spoilt by heavy siltation. One example is the Lovu estuary, south of Durban, between 2-3 m deep, its depth has been now reduced to 0.4 m to within a kilometre from the sea.

Estuaries have also become deposits for waste dumped by industries built on their banks (McLusky 1974). This can result in nutrient enrichment (eutrophication), oxygen depletion and poisoning of organisms by heavy metals or pesticides. Apart from pollution, man has brought about many other changes in estuaries, in the form of harbours and dredging. Dredging can have potentially disastrous effects through erosion and remobilization of pollutants from sediments with accompanying increases in turbidity and burial of organisms and also removal of organisms with the sediment.

One example is the building of the Richards Bay harbour, which involved dredging of the northern part of the Bay and separation of the harbour section from a sanctuary area (Day 1981). The St Lucia Narrows also has a well-documented history of sediment disturbance arising from dredging and beam trawling (Owen & Forbes 1997). Farming in the catchment has resulted in high sedimentation rates (Begg 1978). These disturbances cause a decline in species richness due to loss of habitat. Hay (1985, in Cyrus & Blaber 1987) compared the benthic fauna in dredged and undredged areas of St. Lucia estuary. He found that the channel areas dredged in St. Lucia between 1973-76 were still devoid of benthic animals while the undredged areas were densely populated. Owen and Forbes (1997) reported that the St. Lucia Narrows channel, which was dredged, was impoverished compared with the adjacent mudflats. Therefore, any dredging project should be subjected to a proper environmental assessment before being undertaken in order to reduce such disturbances.

Estuaries are also exposed to natural episodic events to which fauna can survive if these conditions occur briefly. However if these natural conditions such as floods and droughts are severe and prolonged the nature and productivity of the estuarine biota is affected. Apart from raising the water level and increasing current velocities, floods have many secondary effects (Day & Grindley 1981). The increased discharge of the river at first reduces salinity of the surface layers and then as turbulence increases, the substrate of the estuary is affected and in extreme cases the whole estuary may become fresh. In the short term, this type of episodic event has a major direct effect on the benthic fauna with densities of some species being depressed for a number of months. Direct long-term effects are related to disappearance of some species for several months e.g. Cyrus (1988) found that estuarine bivalves were unable to successfully recolonize St Lucia system at the lowered salinity caused by Cyclone Domonia, which flushed out the entire system in 1984. The effects of drought result in increased salinity and have been investigated in the St Lucia Lake and Narrows in Natal (Millard & Broekhuysen 1970, Owen & Forbes 1997). Owen and Forbes (1997) recorded significant changes in species composition and densities after the cyclone flooding in 1984 and the hypersaline period in 1992/93. They found a decline in organism densities after the cyclones with the dominant species, *Paratylodiplax blephariskios*, density decreasing during a period of hypersalinity in St Lucia system.

1.1.5. Prawn farming and the estuarine environment

The development of marine aquacultures, such as prawn farming, generally generates profit and income, but also bears risks of negative environmental impact (Tovar, *et. al* 2000). Wastewater discharged from intensive mariculture into estuaries may lead to deterioration in water quality resulting from depletion of dissolved oxygen, discharge of organic matter and eutrophication. However, the deleterious effects of aquaculture effluents on the water quality of receiving waters depend on various factors such as the magnitude of discharge, chemical composition of effluents (nutrients, organic matter) and the characteristics of the receiving water (dilution rate, residence time) (Paez-Osuna 2001)

The Mtunzini prawn farm, registered as the Mtunzini Prawns (PTY) LTD, was established in 1992 adjacent to the middle reaches and approximately 500 m from the Mlalazi estuary. The prawn farm operates by abstracting water from the estuary, which is then spiked with nutrients to enhance the growth of the prawns. The water is then released back into the estuary as effluent about two kilometres downstream of the abstraction point. No impact study was done to establish the possible effects of the prawn farm on the estuary, although concern has been expressed about the potential impact of nutrient enriched water being discharged into the estuary (Mann, *et. al* 1996). Indications are that the water returning to estuary from the prawn farm is rich in phosphates. This is based on the fact that before post larvae are put into the growing ponds, phosphate is added for algal growth that the larvae utilise as food (pers. observ.). Much of the nutrients entering the coastal environments from aquacultures indicate that most of this material originates from added feeds (Trott & Alongi 2000).

It is important to maintain a relatively unspoilt estuary where medium to long term data from such an estuary can be useful in determining seasonal changes and natural (episodic) impacts on communities in order to distinguish these from anthropogenic and human induced impacts.

1.1.6 Motivation and objectives

From the literature and general observation it is apparent that man is generally degrading estuaries through increased utilisation. It is thus important to understand the many and complex ecological interactions that take place in estuaries if they are to be prevented from being irrevocably spoilt. This can be done by studying the different biotic components of estuaries in order to develop medium to long term data bases so as to establish natural patterns, monitor changes and be able to predict impacts in these systems. Although the estuarine environment is regarded as harsh, the plant and organism communities remain relatively stable in spite of natural fluctuations. It is mostly extreme changes of long duration, which impoverish the flora and fauna (Day 1981). Human activities in the catchment and estuary itself, such as pollution from pesticides which drain into the estuary from the farmlands, toxic metals and oily waste from industries and prawn farm activities that tend to intensify the effects of nature. Disturbances in the estuary due to above situations together with the variable nature of the physico-chemical condition increase stress leading to a decrease in diversity.

Apart from a study by Hill (1966) there have been no benthic studies undertaken in the Mlalazi estuary. Hill (1966) identified a total of 84 species of benthic invertebrates in the estuary. The distribution of fauna in the estuary was to a large extent determined by the nature of the substratum, with polychaetes such as *Glycera convoluta* dominating the muddy areas. The crabs such as *Neosersama meinerti* and *Uca urvillei* were found in consolidated mud and dominated the epifauna. The infauna was rich in polychaetes.

A survey of the benthic community of the Mlalazi estuary, conducted by the Coastal Research Unit of Zululand for the purpose of a masters study, was initiated during the period 1989 to 1991 but the data was never written up or published. The purpose of the study was to determine the composition and abundance of benthic fauna in the estuary before the construction of the prawn farm. These data are, however, available and will be used in the present study. No study to date has looked at the direct impact of the prawn farm on the biotic community of the estuary. An attempt will be made during this study to relate changes in the benthic community and water quality to the possible impact of the prawn farm effluent. It should, however, be noted that any change in the system between the two sampling periods cannot be solely attributed to prawn farm activities i.e. "cause and effect" without taking into account other environmental factors which drive changes in an estuary as reflective of the pre-prawn farm condition. This is because it frequently happens that in the estuarine environment, two or more factors vary together and therefore it is difficult to decide which of them is responsible for an observed effect. Thus a correlation may be established, but not a cause. The benthic community is influenced by various physical factors and one or all of these factors are to a lesser or greater extent responsible for any change in the community structure.

The intention of this study is to:

- Describe the benthic community structure recorded during the period February 1989 to February 1991 and August 1999 to July 2000.
- Describe and compare any spatial and temporal changes in the benthic communities as recorded during 1989-1991 and 1999-2000 sampling periods.
- Determine the physical characteristics of the Mlalazi estuary during the 1989-1991 period and the current sampling period and to relate any changes in benthic community structure to changes in the physical environment.
- Investigate the potential correlation between prawn farm activities and changes in the water quality and macrobenthos of the Mlalazi estuary.

CHAPTER 2

2.1. Study Area

The Mlalazi River (Figure 2.1) arises on the Ngoye ridge near the town of Eshowe. It has a catchment of 415 km^2, with a mean annual rainfall of approximately 1250 mm. $yr.^{-1}$, most of which falls in summer. The estuary is situated in the Umlalazi Nature Reserve and has a tidal reach of about 8 km. It is normally between 1-3 m deep and about 100 m wide for most of its length (Day 1980). The estuary can be divided into four regions, the head, channel, lake, and mouth region (Hill 1966, Figure 2.2). The lake is the widest part of the estuary, being between 200-300 m in width and about 700 m long. The channel is 1.5 km long and extends from the railway bridge to the lake. The lake and the channel are bordered by the mangroves *Avicennia marina* and *Bruguieria gymnorrhiza,* behind which lie marshes dominated by the reed *Phragmites australis* and *Juncus*. The mouth region stretches from the sea to Bishop's Trees. The Mlalazi estuary is considered a temporary open/closed estuary (Hill 1966, Day 1981, Mackay 1996) and it was open during the study period (1999-2000).

2.2. Available information on the physical and biological parameters

A synthesis of the available ecological information on the Mlalazi estuary is provided by Day (1981). Hill (1966) undertook a survey of the Mlalazi estuary during which physical and biological parameters were recorded, however the survey was not quantitative. Hill (1966) first reported salinity gradients in the Mlalazi estuary and found that vertical salinity gradients are common in the estuary. In order to obtain an overall picture of the variations he measured salinity over the whole length of the estuary during the neap and spring tides. At low neap tides, salinities of less than 10‰ extended halfway down the estuary. At high spring tides, salinities of more than 30‰ extended more than 5 km from the mouth. The smallest changes occurred at the ends of the estuary, i.e. near the sea and near the river interface. He also noted that organisms living near the bottom were subjected to small salinity variation whereas those living near the surface were exposed to considerable variation, this being due to salinity stratification.

Temperature in the Mlalazi estuary undergoes a typical seasonal variation (Hill 1966). During dry weather the water in the estuary is at a slightly higher temperature on the surface than in the sea due to the warming effects of the sun on sheltered waters. Water temperature in winter varies between 15°C and 18°C, increasing to between 23°C and 28°C in summer.

Typically, estuaries are sandy at the mouth from marine derived sediments, which enter the mouth by wave action and flood tide currents. The lower reaches are typically sandy with mud present, the middle reaches are muddy with sand present, the upper reaches are muddy and the head is often characterised by black mud (Day 1981). Hill (1966) found that the distribution of benthic fauna in Mlalazi estuary is determined to a large extent by the nature of the substratum. The substratum at the mouth was clean sand and the fauna not rich. The channel was muddy with polychaetes *Glycera sp.* and *Dendronereis arborifera* commonly found here. In the lake, mudflats were often exposed at low tide and consisted mainly of soft mud but at the Islands (Figure 2.2.) and in the south-western corner of the estuary the substrata was sandy mud. In the upper reaches, below the rail bridge, the substratum was sandy with the polychaete *Dendronereis arborifera* commonly being recorded here.

2.3. **Materials and Methods**

2.3.1. **Sampling**

Sampling was conducted during two periods. During the period February 1989 to February 1991, benthic and water quality data were recorded monthly from the estuary. These were collected from six sites located along the estuary as follows (Figure 2.2):

Site 1 - about 50-100 m above the railway bridge, in the upper reaches of the estuary (1989-1991).

Site 2 - on the bend opposite the Kwa-Zulu stream which enters the northern bank of the channel (1989-1991).

Site 3 - about 10-20 m from the slipway (1989-1991).

Site 4 - on the inside of the sweeping curve into the channel (1989-1991).

Site 5 - off the sandbank halfway across the channel (1990-1991).

Site 6 - opposite Nel's pool in the northern bank (1990-1991).

Additional sampling in the Mlalazi estuary was carried out on a monthly basis starting in August 1999 and ending in July 2000. The six sites from which the sampling was done are the same as those at which samples were collected during 1989\91, with the addition of site 7, almost 800m from the mouth and halfway across the wall of granite boulders on the western bank (Figure 2.2).

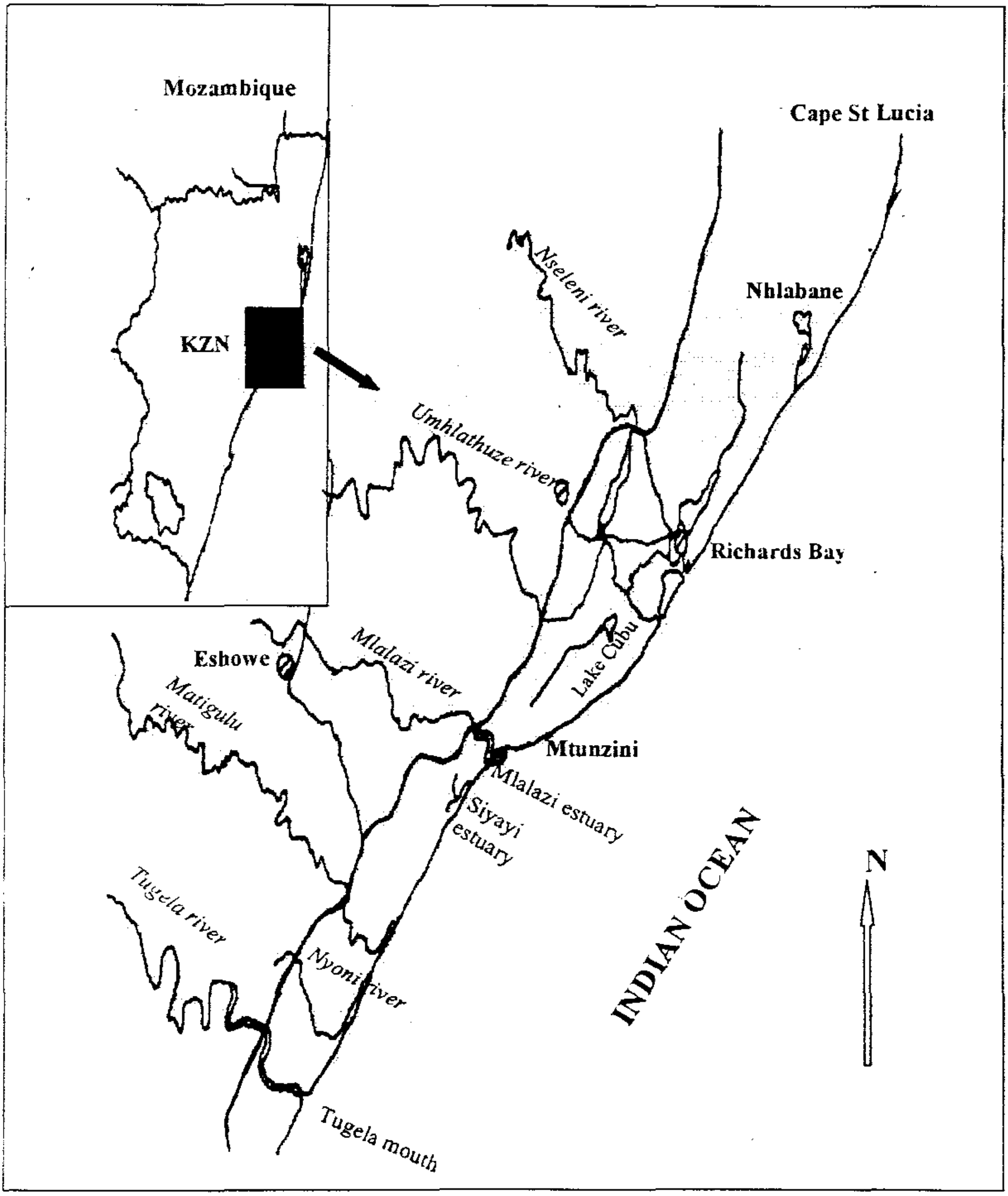

Figure. 2.1 Map showing the general locality of the Mlalazi estuary and the origin of Mlalazi river near town of Eshowe (after Cooper 1991).

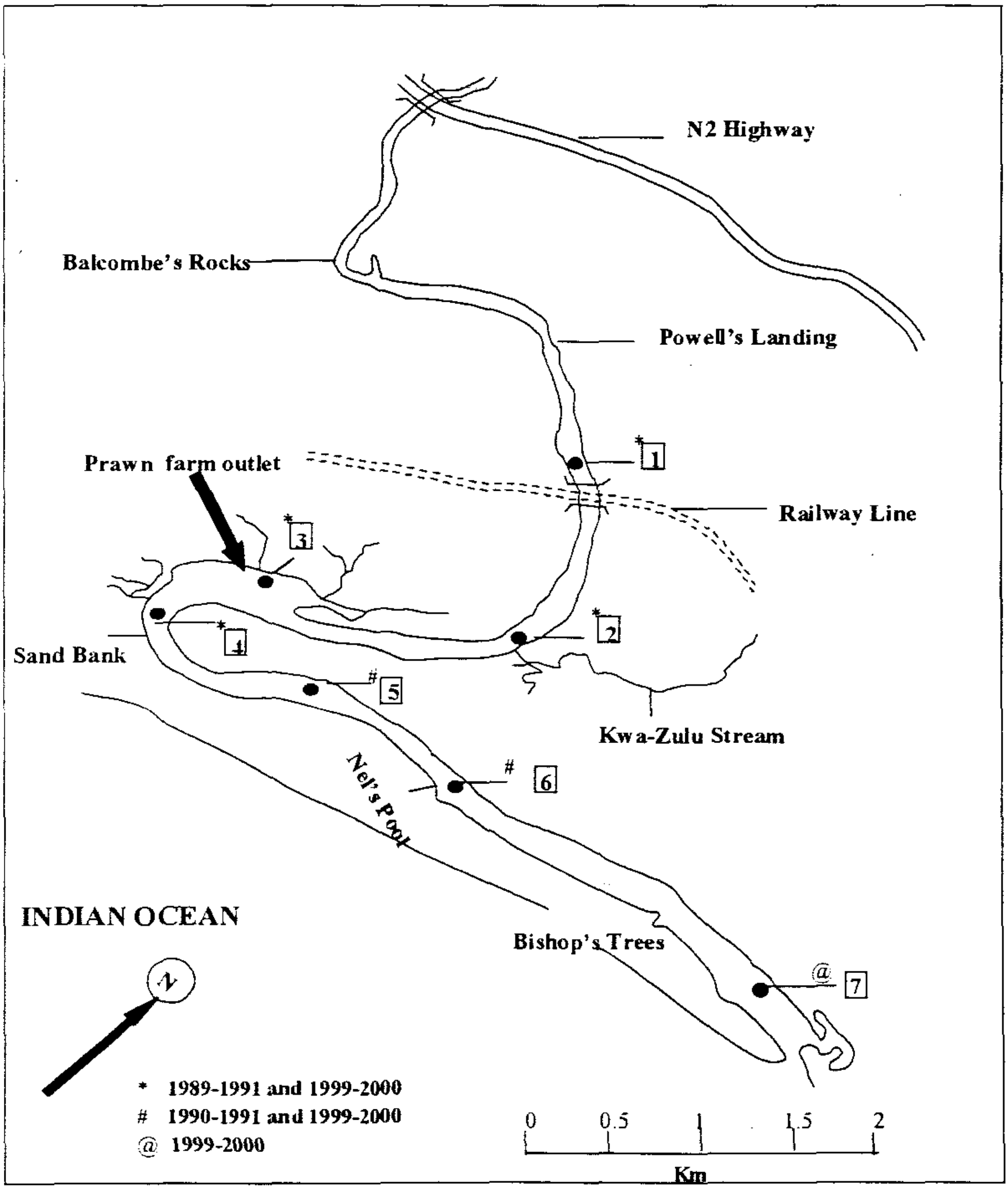

Figure. 2.2. The Mlalazi estuary, showing the location of sampling sites (after Hill 1966).

CHAPTER 3

Physico-chemical Parameters

3.1. Introduction

An estuary forms a transition between fresh water and seawater and retains characteristics of both fresh water and the marine environment. A universal and characteristic feature of estuaries is the variation in physical and chemical conditions both along the length of the system and over time. Estuaries are dynamic systems and any physical and chemical feature associated with them is subject to rapid and sometimes extreme changes. As a consequence southern African estuaries are highly variable, unstable and unpredictable habitats where species diversity is generally low (Whitfield 1994).

3.1.1. Water quality of estuaries and effects on organisms

Estuarine organisms are not only affected by the fluctuating environment but also the water quality and nature of the substratum. Water quality can be defined as the physical, chemical and biological properties of water that determines its fitness for use and for the protection of the health and integrity of aquatic ecosystems (DWAF 1995). Any dramatic change in water quality will affect the biota in various ways, depending on the variable in question, the ambient water quality as well as the organisms involved (DWAF 1995).

In South Africa, utilisation of the coastal zone, through industrial development, waste deposition and water abstraction, is increasing rapidly. This often leads to deterioration of water quality in aquatic bodies. When pollutant/toxic materials are discharged into a body of water, changes occur which may have deleterious effects on aquatic life (Cairns, *et. al* 1976). Such effects can be measured in terms of changes in the physical, chemical and biological characteristics of the receiving waters.

Effluent that is high in degradable organic matter (such as domestic sewage) will rapidly use up dissolved oxygen resources, which is essential for aquatic life (Palange & Zavala 1987). Solids suspended and dissolved, impart a turbid appearance and restrict light penetration, which lead to oxygen deficiency with deleterious effect on aquatic life. An increase in temperature may be caused by thermal pollution and this leads to lowering of dissolved oxygen concentration and acceleration of chemical and biological processes. Water bodies may also become enriched by nutrients contained in waste effluents (Palange & Zavala 1987). Nitrogen and phosphates in effluents contribute to increased growths of aquatic plants and eutrophication. Algal blooms often result in anaerobic conditions that are detrimental to aquatic life. Besides pollution induced by man's activities, natural episodic events such as floods and droughts also play a role in lowering the water quality of water bodies. Floods usually increase discharge to the estuary, which is accompanied by a reduction in salinity and the erosion of substrate, thus the loss of habitat (Day & Grindley 1981). The effects of droughts lead to a decrease of river discharge accompanied by an increase in the salinity in the estuary (Day 1981). Other undesirable water contamination due to natural causes includes deposition of fly ash from natural forest fires, natural erosion and weathering of rocks. These events may have the same potential adverse impact on water as pollution.

3.1.2. **Effect of aquaculture and agricultural practises in estuaries**

Agriculture has been identified as a potential source of water pollution as a result of the increasing intensity of animal and crop production, and the link that has been demonstrated between agricultural run off and specific water quality problems (Loehr 1979, Mackay 1996, Pearce & Schumann 1997). Although relationships between animal and crop production practises, agricultural land use and adverse impacts on water quality of aquatic bodies have intensified in recent years, there are approaches that can be used to minimise the adverse effect of agriculture on aquatic bodies. These approaches include runoff and erosion control practises, better use of pesticides and fertilisers, soil and water conservation practises (Loehr 1979). Mackay (1996) reported a decrease in nitrogen and phosphate levels entering the Siyaya estuary compared to that recorded in the estuary during previous studies (Day 1981, Birnie 1997). This decrease in nutrient concentrations was attributed to changes in land-use in the catchment as well as better use of fertilisers.

Aquaculture is the aquatic counterpart of terrestrial agriculture. The principles and many problems are the same though there are unique characteristics associated with aquaculture practises (Reay 1979). Aquaculture effluents have been related to eutrophication problems in aquatic bodies due to the fact that aquaculture operations usually discharge effluents that are enriched with suspended solids and nutrients into aquatic bodies (Beveridge 1996, Paez-Osuna 2001). Most aquaculture farms depend on aquatic bodies such as estuaries for water used in their activities and as such cause organic pollution in these aquatic bodies. The prawn farm (Mtunzini Prawns PTY LTD) situated adjacent to the Mlalazi estuary is one example. The farm abstracts water from the estuary that is pumped into the ponds and nutrients added to enhance algal production, which the prawns utilise as food. The nutrient rich water is then released back into the estuary as an effluent. Due to the impacts of these aquaculture activities on coastal waters, introduction of management measures to mitigate the adverse environmental impacts has now become necessary and urgent. The reduction of nutrient input has been shown to be an effective strategy for lowering the load of nitrogen and phosphorus released into the environment from aquaculture operations (Paez-Osuna, *et. al* 1998). At present there is no control or management strategy pertaining to the amount and quality of the effluent being released by the prawn farm.

An essential prerequisite to the management of the quality of natural aquatic bodies is a detailed inventory of water quality guidelines of all pollutants from all significant sources (Gower 1992). These guidelines should also include nutrients such as nitrogen and phosphates. This is due to an increasing awareness that the disposal of nutrient rich agricultural or aquaculture effluents into aquatic bodies can lead to accelerated growth of algae, causing eutrophication problems (Beveridge 1996).

3.1.3. **Advantages and disadvantages of water quality monitoring**

Deterioration of water quality caused by pollution and episodic events may have serious consequences for human health (drinking water), wildlife (fish and other aquatic life), sport (fishing and boating), agriculture (commercial crops) and other users of an estuary.

This emphasises the importance of water quality monitoring. The general advantages of water quality monitoring are (Gower 1992):

- Increased knowledge of existing water quality conditions and understanding of the environment.
- Provide information on post and present effects of natural and anthropogenic impacts on the aquatic environment in order to predict impacts.
- Differentiate between natural and anthropogenic impacts.
- Assess the effectiveness of pollution control measures.
- Monitor polluting systems such as industrial complexes to safeguard water supplies.
- Detect trends in water quality and provide an early warning system.
- Identification of sources of pollutants and their loads.

There are also some disadvantages to water quality monitoring. It is expensive and therefore care must be taken to ensure that the resources are employed to the best advantage. The fact that most contaminants are found at low concentrations in a body of water can introduce inadvertent errors in collection and analytical procedures due to either sample contamination or to loss of the contaminant during sample handling. The time scale factor is another disadvantage since physico-chemical parameters are measured discontinuously, thereby giving snapshots and not the whole picture of what is happening in that particular body of water (Cairns, *et. al* 1976). The greatest disadvantage of using water quality in measuring the health of a particular body is the lack of any useful correlation between the concentration of the contaminant present and their bioavailability (Phillips & Rainbow 1994). As the bioavailability of a contaminant cannot be inferred from its concentration in water samples, it can be concluded that organisms should be preferred for most monitoring tasks.

3.1.4. <u>**Water quality monitoring in the Mlalazi estuary**</u>

No historical water quality data exist for the Mlalazi estuary. Data collected during the 1989-1991 and 1999-2000 sampling periods will be used to describe the water quality of the Mlalazi estuary before and after the prawn farm was established. Due to the nature of aquaculture operations, high nutrient concentrations are discharged into estuaries leading to eutrophication problems in the estuary. Comparison of the 1989-1991 data with the 1999-2000 data will thus reveal if changes have occurred in the water quality of the estuary since the prawn farm was established and whether these changes were likely to have had any effect on the biota of the estuary.

3.2. **Materials and Methods**

3.2.1. **Sampling**

Water quality of the Mlalazi estuary was measured monthly at Sites 1-4 during the period February 1989 to December 1989. In January 1990, only Sites 1 and 2 were sampled. From February 1990 to February 1991, Sites 1-6 were sampled on a monthly basis. From August 1999 to July 2000, Sites 1-7 were sampled on a monthly basis. Sites 1-6 sampled during the 1999-2000 period corresponded to the sites used during the 1989/91 sampling period (Figure 2.2).

3.2.1.1. Water quality

During the 1989-1991 sampling period, top and bottom measurements of salinity, temperature and dissolved oxygen were recorded. Only bottom measurements of turbidity were taken during this period. Oxygen (0.1 mg/l precision limit) and temperature (0.1°C precision limit) were recorded using a WTW OXI Microprocessor. Turbidities (0.1 NTU precision limit) were measured in Nephelometric Turbidity Units (NTU) using a Hellige Digital Direct Reading Turbidimeter. Salinities (1 ‰ precision limit) were measured using an American Optics temperature compensated refractometer. Water samples were also collected from three sites (Sites 1, 3 and 5) in August 1990 and February 1991 from which the following physico-chemical parameters were recorded: chloride, conductivity, chemical oxygen demand, ammonium, nitrate, orthophosphates, total phosphates, total dissolved solids, pH and total suspended solids.

During the 1999-2000 sampling period, top and bottom measurements of temperature, salinity, dissolved oxygen, oxygen saturation, pH, and conductivity were measured at Sites 1-7 using a Hydrolab Datasonde data logger. In addition to the seven sites sampled, the prawn farm outlet was also sampled from February 2000 to July 2000. For turbidity, water samples were collected at each site and measured in the laboratory using a Hatch Turbidimeter. Water samples were also collected at each site in 2l plastic bottles from which ammonia, nitrate, nitrite, ortho-phosphate, total phosphate, chemical oxygen demand (COD) and sulphate concentrations were determined using a Merk SQ.118 spectro-photometer. Water samples for determination of chlorophyll-a were also collected in 2l plastic bottles and analysed by the SABS/ISO 17025 accredited water analysis laboratory at the Mhlathuze Water Board.

3.2.1.2. Sediment samples

Sediment samples from Sites 1-6 were collected once during the 1989-1991 sampling period, while monthly sediment samples were collected at Sites 1-7 during the 1999-2000 sampling period. Grain size and organic content analysis were carried out on the sediment samples collected during the 1999-2000 period, whereas only grain size analysis was performed on the 1989-1991 sediment samples. Grain size was assessed using the wet sieve method, where the sediment subsamples (±50 g) are placed on a series of screens following a decreasing geometric scale from 1000 to 63μm (Gray 1981).

The percent organic content of the 1999-2000 sediment samples was determined by oven drying the sediment samples at 60^0C for 24 hours. Samples were then transferred to crucibles and incinerated for 6 hours at 600^0C. Each sample was then weighed to a constant mass to determine the percent organic content. The percent organic content of sediment samples was then graded as follows:

Very low	=	<0.5%
Low	=	0.5 - 1.0%
Moderately low	=	1 - 2%
Medium	=	2 -4%
High	=	>4%

3.2.2. **Data analysis**

3.2.2.1. Water quality

The environmental data of the 1989-1991 and 1999-2000 sampling periods were entered into a matrix with rows as samples and columns as variables. Seasonal values were calculated by grouping monthly data, i.e. December + January + February = Summer, March + April + May = Autumn, June + July + August = Winter, September + October + November = Spring. The seasonal values calculated were then plotted to determine spatial and temporal patterns.

3.2.2.2. Sediment

Sediment samples were assessed for grain size and organic content. The cumulative percentage of particle size for each sample was plotted against the corresponding phi value (Φ), where $\Phi = -\log_2$ particle size. From the data the median phi values and sorting coefficients were determined by plotting the phi values against cumulative dry weight of the sediment fractions (Gray 1981). Sorting is a measure of the spread of grain distribution and is calculated from the Inclusive Standard Deviation index, given by the formula:

$$\frac{\text{Phi @ 84\% - phi @ 16\%}}{4} + \frac{\text{Phi @ 95\% - phi @ 5\%}}{6.6}$$

This formula covers 90 % of grain distribution and is therefore a better overall measure of sorting (Gray 1981). The sorting classes produced by this index are as follows: -

Under 0.35 Φ -- very well sorted

0.35 - 0.50 Φ -- well sorted

0.50 - 0.71 Φ -- moderately well sorted

0.71 - 1.00 Φ -- moderately sorted

1.00 - 2.00 Φ -- poorly sorted

2.00 - 4.00 Φ -- very poorly sorted

Over 4.00 Φ -- extremely poorly sorted

3.3. Results

3.3.1. Differences between sites

3.3.1.1. Physical variables

Mean and ranges of physical variables [salinity (‰), temperature (°C), oxygen (mg/l) and turbidity (NTU)] recorded at each sampling site during the 1989-1991 sampling period are illustrated in Figure 3.1. Figure 3.2 illustrates the mean of physical variables [salinity (‰), conductivity (mS/cm), dissolved oxygen (mg/l) and oxygen saturation (%)] measured during the 1999-2000 sampling period.

A salinity gradient was evident in the estuary over the study period, with mean salinity increasing from the head to the mouth of the estuary. The estuary ranged from fresh to marine dominated during both the study periods. Sites in the upper reaches showed the river influence, with low salinity values being recorded in these areas. Mean bottom salinity values were generally higher than surface values, a factor attributed to the intrusion of denser saline waters from the sea forming well marked vertical stratification in the estuary.

Conductivity values during the 1989-1991 period were very high at all sites in the August 1990 sampling period compared to the February 1991 period. In August 1990, conductivity ranged from 1 720 mS/cm at Site 1 to 2 830 mS/cm at Sites 3 and 5. In February 1991 conductivity ranged from 181 mS/cm at Site 5 to 28 mS/cm at Site 1. During the 1999-2000 sampling period, conductivity followed a similar pattern to salinity where the lowest levels were recorded in the upper and head regions increasing towards the lower and mouth regions. Conductivity ranged from 0.1 mS/cm to 53.7 mS/cm during the 1999-2000 period. Top and bottom measurements of conductivity differed with bottom conductivity consistently higher than top conductivity at all sites sampled.

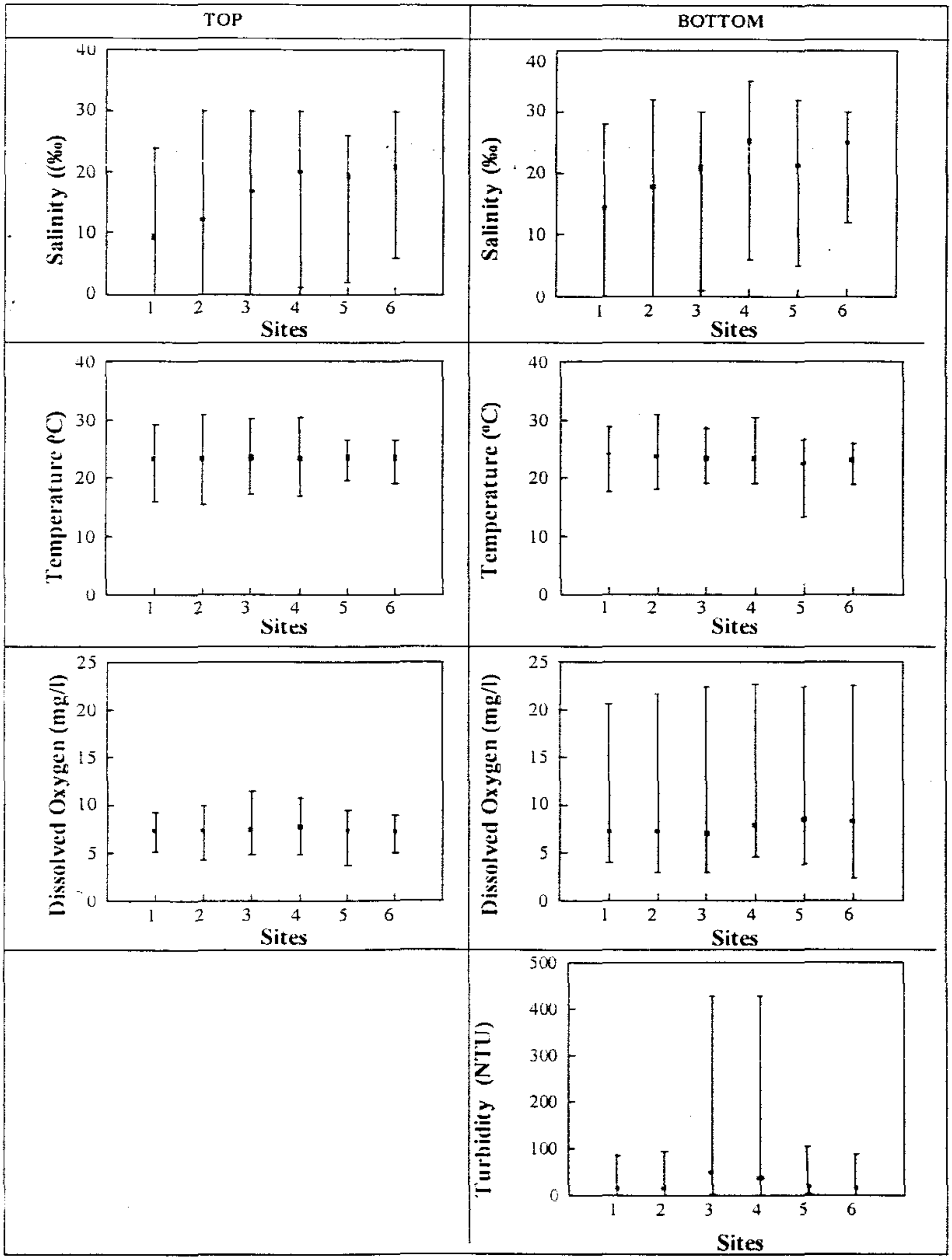

Figure 3.1. Spatial variations of salinity, temperature, dissolved oxygen and turbidity recorded from the Mlalazi estuary during the 1989-1991 sampling period. (Bars indicate mean and the range).

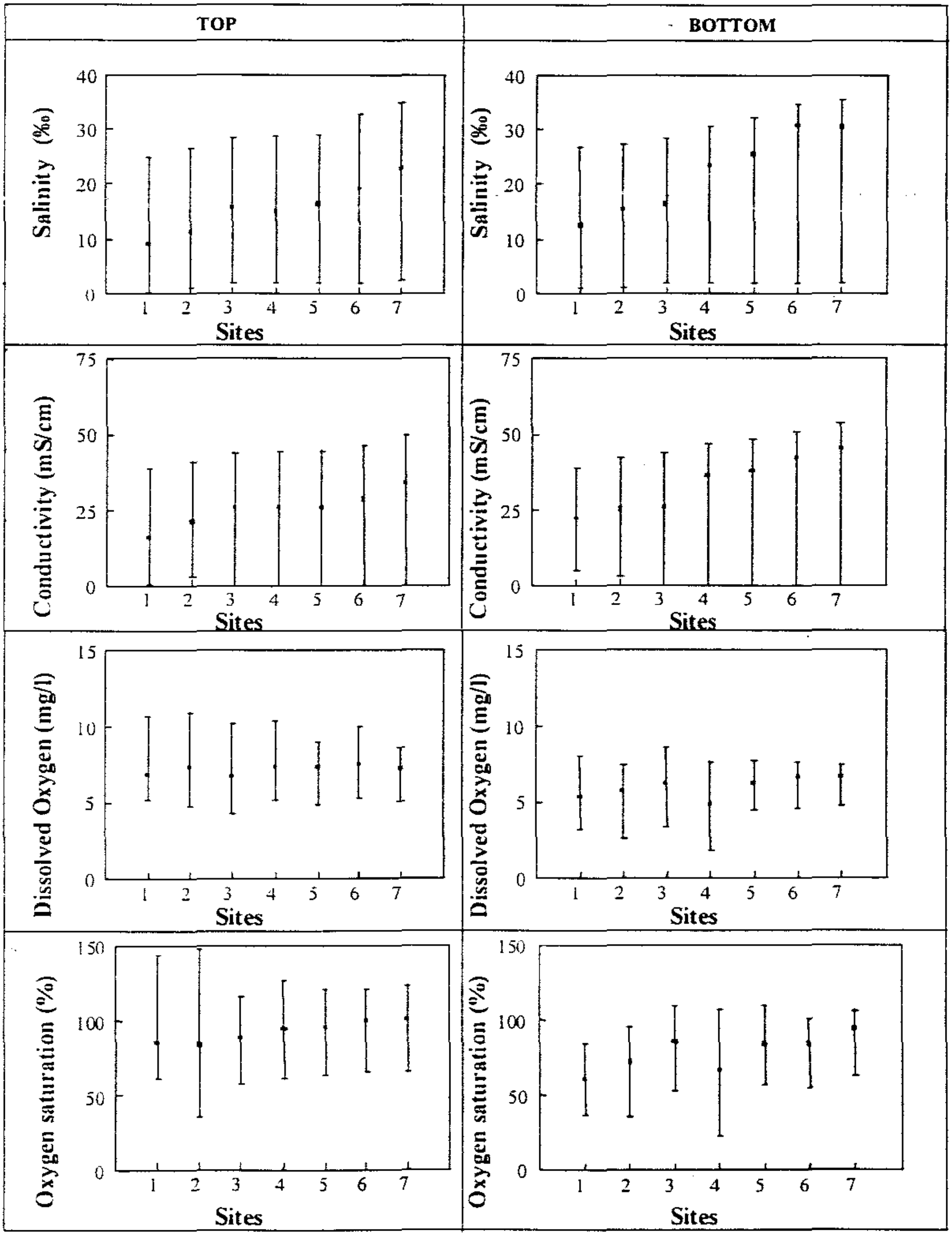

Figure 3.2. Spatial variations of salinity, conductivity, dissolved oxygen and oxygen saturation recorded from the Mlalazi estuary during the 1999-2000 sampling period. (Bars indicate mean and the range).

There was a slight increase in dissolved oxygen from the head towards the mouth of the estuary during both study periods. Minimum values were recorded in the middle reaches during both sampling periods with bottom measurements slightly lower than those from the top. Dissolved oxygen concentrations were slightly lower during the 1999-2000 period (Figure 3.2), ranging from 1.8 mg/l to 10.9 mg/l compared to the 1989-19991 period, when dissolved oxygen ranged from 2.9 mg/l to 22.6 mg/l (Figure 3.1).

Oxygen saturation generally increased from the head to the mouth of the Mlalazi estuary during the 1999-2000 period (Figure 3.2). A minimum concentration of 22.1 mg/l was recorded at Site 4 (middle reaches) while a maximum concentration of 144 mg/l was measured at Site 2. The low oxygen saturation at Site 4 corresponded to low dissolved oxygen levels also recorded at Site 4.

Figure 3.3 represents conductivity, pH, TDS, TSS and COD concentrations measured at Sites 1, 3 and 5 during August 1990 and February 1991. Temperature, turbidity, pH, and COD measured at Sites 1-7 during the 1999-2000 sampling period are presented in Figure 3.4.

The temperature data did not show any spatial pattern during the two study periods. The values ranged from 13.3 °C to 31 °C during the 1989-1991 period (Figure 3.1) while temperature ranged from 17.88 °C to 29.95 °C during the 1999-2000 period (Figure 3.4). There were no marked differences between top and bottom temperatures.

The 1989-1991 sampling period showed that the middle reaches (Sites 3 and 4) were highly turbid compared to the rest of the sites while in the 1999-2000 period Sites 2, 4 and 5 were more turbid than the other sites. The high turbidity levels in the middle reaches during the 1989-1991 period corresponded to these areas being muddy. Turbidity levels during this period ranged from 0.8 NTU at Site 1 to 427 NTU at Sites 3 and 4 (Figure 3.1). During the 1999-2000 sampling period, turbidity levels ranged from 10 NTU at Site 2 to 198.5 NTU at Site 5 (Figure 3.4). Generally, mean turbidity levels during both sampling periods were below 60 NTU.

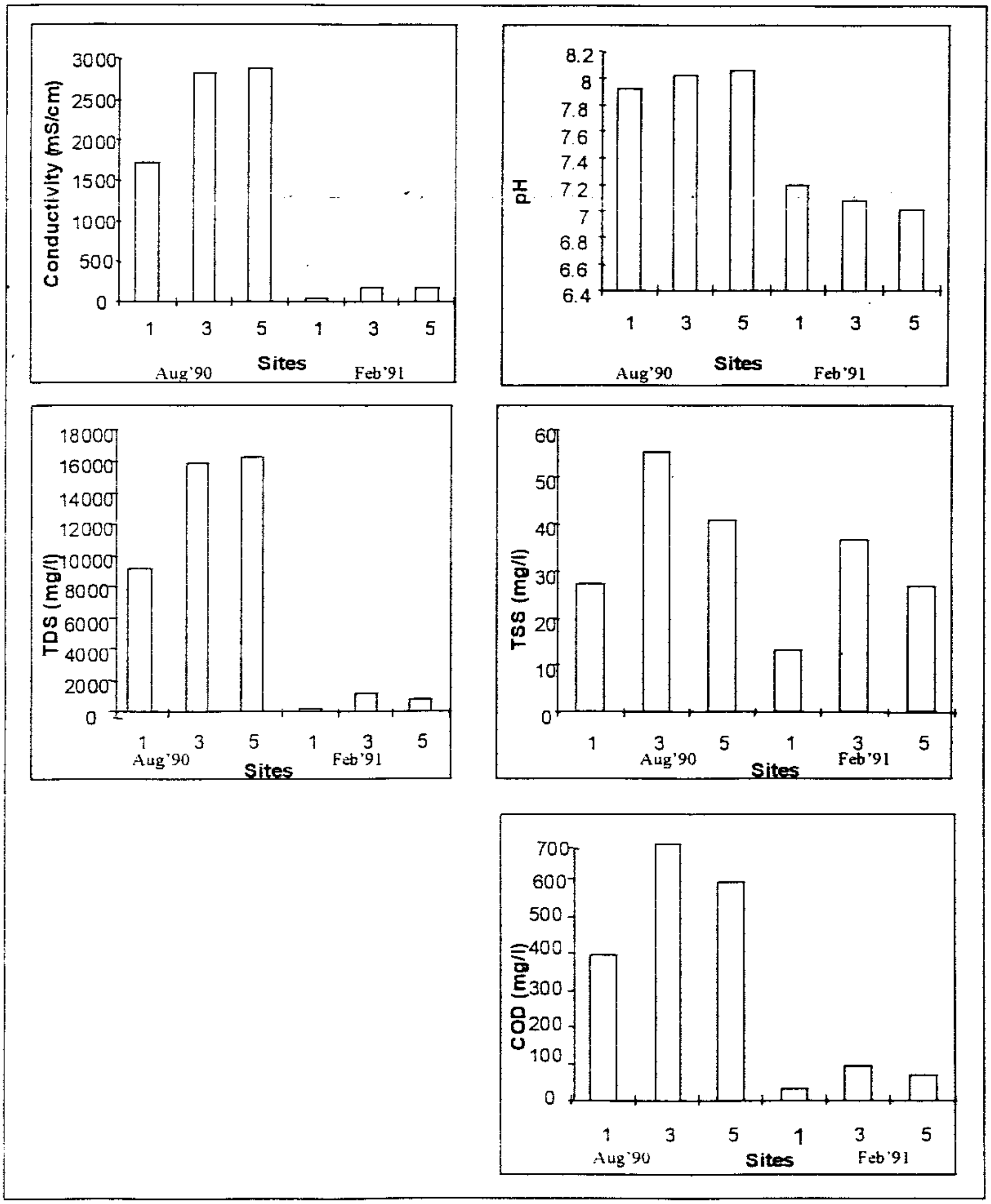

Figure 3.3. Spatial variations of conductivity, pH, TSS, TDS and COD recorded from the Mlalazi estuary during the 1990-1991 period.

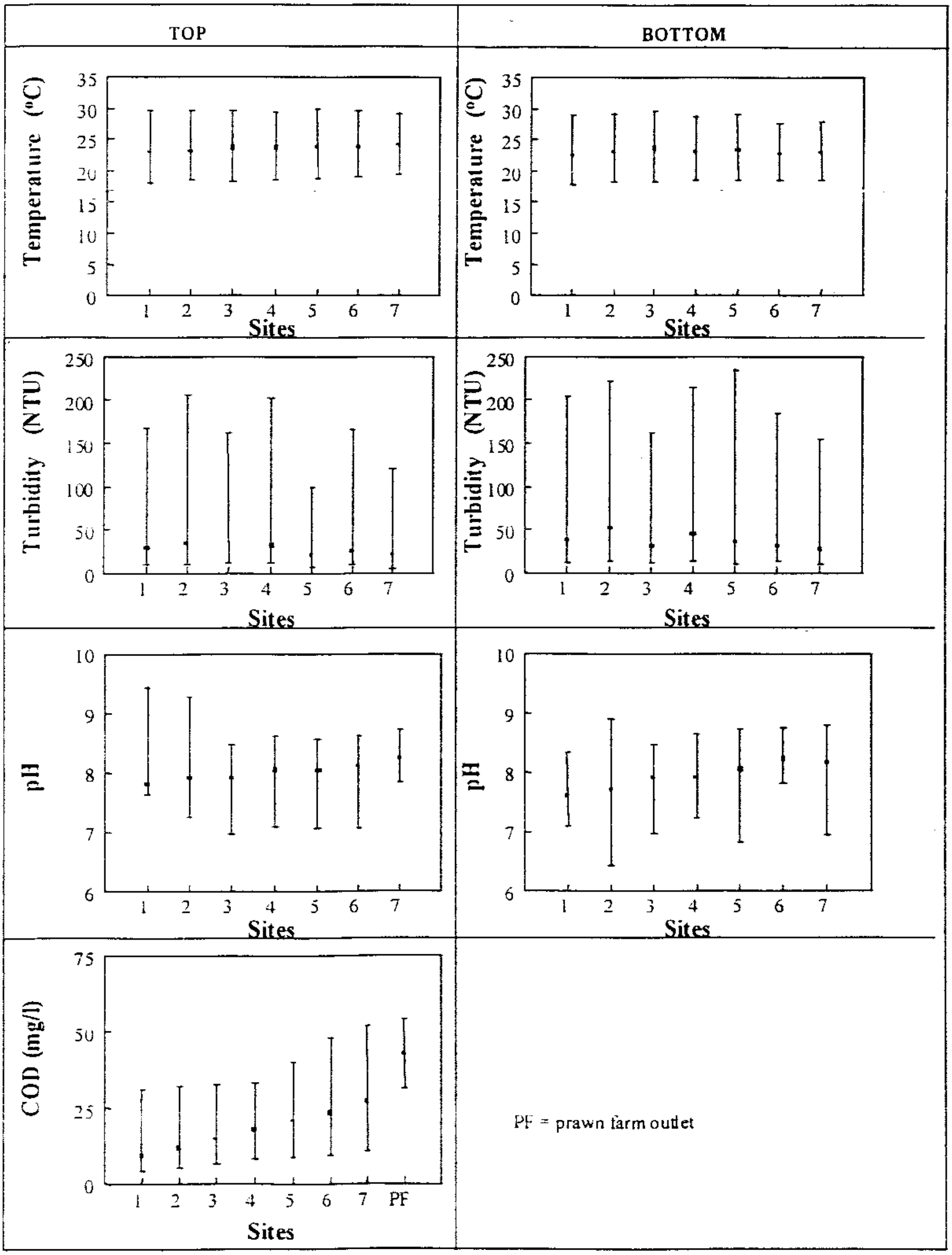

Figure 3.4. Spatial variations of temperature, turbidity, pH, and COD recorded from the Mlalazi estuary during the 1999-2000 period. (Bars indicate mean and the range)

During 1989-1991, pH ranged from 7.92 at Site 1 to 8.06 at Site 5 in August 1990 and from 7.01 at Site 5 to 7.2 at Site 1 in February 1991 (Figure 3.3). During 1999-2000, pH ranged from a minimum of 6.44 at Site 2 to a maximum of 9.45 at Site 1 (Figure 3.4). The mean pH value during this period ranged from 7.6 to 8.24 and these values are considered 'normal' for estuarine waters since are close to the neutral value of 7 preferred by aquatic organisms.

TSS and TDS values measured during the 1990 and 1991 periods showed increasing concentrations from the upper reaches (Site 1) towards the lower reaches (Site 5). The 1990 period also showed very high concentrations of these variables compared to the 1991 period. Total dissolved solids (Figure 3.3) ranged from 9 099 mg/l at Site 1 to 16 300 mg/l at Site 5 during August 1990 while in February 1991, TDS ranged from 250 mg/l at Site 1 to 1 120 mg/l at Site 5. During August 1990, total suspended solids (Figure 3.3) ranged from 27.4 mg/l at Site 1 to 55.2 mg/l at Site 3 whereas during February 1991 TSS ranged from 13.3 mg/l at Site 1 to 36.7 mg/l at Site 3.

COD was very high during August 1990 ranging from 398 mg/l at Site 1 to 697 mg/l at Site 3 (Figure 3.3). This suggests the influx of a high nutrient load into the system during that period. COD values decreased to less than 95 mg/l during February 1991 at all sites. During the 1999-2000 period, there was a general increase in COD concentration from the head to the mouth of the estuary (Figure 3.4). The prawn farm outlet exhibited highest concentrations of 54 mg/l while minimum concentrations were recorded at Site 1. High concentrations of COD in the prawn farm outlet indicated that oxygen-demanding nutrient rich effluent was being discharged into the estuary from the prawn farm.

3.3.1.2. **Chemical variables**

Figure 3.5 presents chemical variables (ammonium, nitrate, ortho-phosphates, total phosphates and chloride) measured at Sites 1, 3 and 5 during August 1990 and February 1991. With the exception of chloride and the addition of sulphate and chlorophyll-a, these variables were also measured during the 1999-2000 sampling period at Sites 1-7 and are presented in Figure 3.6. The prawn farm effluent data is presented as PF on this Figure (Figure 3.6).

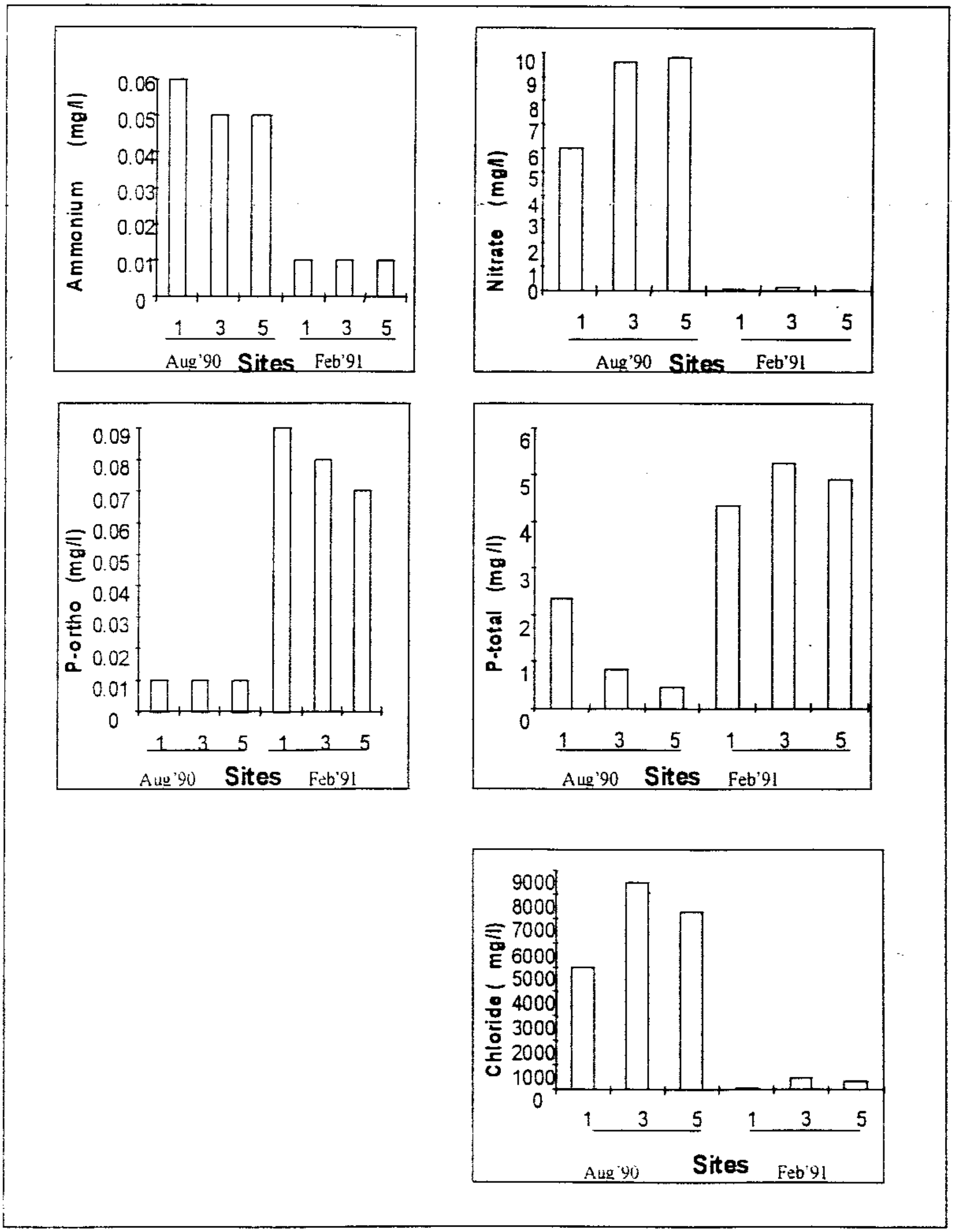

Figure 3.5. Spatial variations of ammonium, nitrate, ortho-phosphate, total phosphate and chloride recorded from the Mlalazi estuary during the 1990-1991 sampling period.

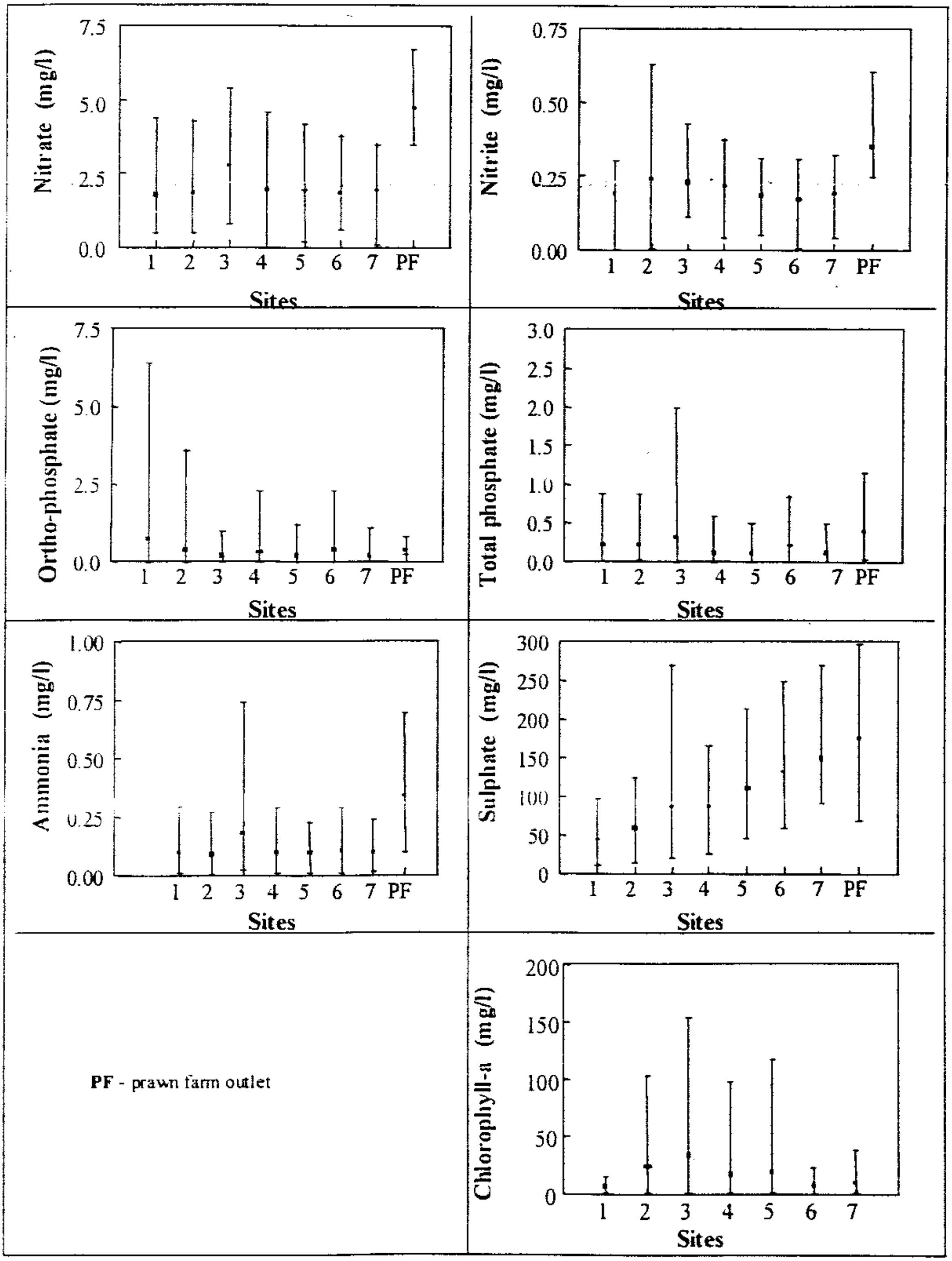

Figure 3.6. Spatial variations of nitrate, nitrite, ortho-phosphate, total phosphate, Ammonia and chlorophyll-a recorded from the Mlalazi estuary during the 1999-2000 sampling period. (Bars indicate mean and the range).

Ammonium concentration ranged from 0.05 mg/l at Sites 3 and 5 to 0.06 mg/l at Site 1 during August 1990. In February 1991, ammonium concentrations decreased to 0.01 mg/l at all sites (Figure 3.5). During the 1999-2000 period, highest ammonium concentrations were recorded in the prawn farm effluent (0.7 mg/l) and at Site 3 (0.74 mg/l), which is the site closest to the outlet. This suggested that the prawn farm effluent as the primary source of ammonia in the estuary. Ammonia concentrations at the other sites were low, ranging from 0.009 mg/l to 0.295 mg/l.

Nitrate concentrations increased from Site 1 to Site 5 during the 1990 and 1991 periods, suggesting that a high load of nitrogen entered the estuary from the river. Nitrate concentration ranged from 6.02 mg/l to 9.81 mg/l in August 1990 and from 0.07 to 0.14 mg/l in February 1991 (Figure 3.5). During the 1999-2000 period, the prawn farm effluent exhibited much higher nitrate concentrations when compared to the other sites. The mean nitrate concentration in the prawn farm effluent during this period was 4.7 mg/l compared to a mean nitrate concentration of 1.99 mg/l for the remainder of the sites. Nitrite concentrations also followed the same pattern as that observed in nitrate concentrations during 1999-2000 sampling period. The mean nitrite concentration in the prawn farm effluent was 0.35 mg/l, compared to a mean nitrite concentration of 0.2 mg/l for the remainder of the sites. This confirms the prawn farm as the primary source of nutrients in the estuary during the 1999-2000 period.

Orthophosphate concentrations remained low during the 1990 and 1991 periods at 0.01 mg/l at all the sites sampled. During the 1999-2000 sampling period highest concentrations (6.4 mg/l) were recorded in the upper reaches gradually decreasing towards the mouth region. This indicated that most orthophosphate were being discharged into the estuary from the river. Total phosphates were low in August 1990 ranging from 0.46 mg/l at Site 5 to 2.33 mg/l at Site 1, compared to February 1991 when concentrations ranged from 3.35 mg/l to 5.25 mg/l. During the 1999-2000 period, the prawn farm effluent and the site closest to it (Site 3) showed high total phosphate concentrations compared to the other sites. The mean total phosphate concentration in the prawn farm effluent was 0.37 mg/l compared to a mean of 0.18 mg/l for the remainder of the sites.

Chlorophyll-a concentration during the 1999-2000 period was very high at Sites 2 and 3. Chlorophyll-a concentration ranged from less than 0.1 mg/l in the lower reaches (Sites 5 and 6) to 33.4 mg/l at Site 3 (Adjacent to the prawn farm outlet).

3.3.2. **Differences between seasons**

3.3.2.1. **Physical variables**

Figures 3.7 represent temporal variations of salinity, temperature, dissolved oxygen and turbidity measured during the 1989-1991 period, while Figure 3.8 represent temporal variations of salinity, conductivity, dissolved oxygen and oxygen saturation measured during the 1999-2000 period. Figure 3.9 illustrates temporal variations of temperature, turbidity, pH and COD recorded during the 1999-2000 sampling period.

Salinities during both the sampling periods showed seasonal fluctuations with maximum values, between 35.0 ‰ and 35.5 ‰, being recorded in winter. It is suggested that freshwater inflow into the estuary was reduced in winter, with the system becoming marine dominated, resulting in higher salinities than during the other seasons. The Mlalazi estuary, being situated in the subtropical region, experiences summer rainfall and as such, lower salinities are expected during summer conditions. During the 1989-1991 period, lowest salinities (mean 16.9 ‰) were recorded during summer while during the 1999-2000 period, lowest salinities (mean 14. 8 ‰) were recorded in autumn. During the 1999-2000 period, heavy rainfall in May 2000 (autumn) decreased salinities to 1-2 ‰ at all sites sampled.

Conductivity values, during 1999-2000, reached a maximum of 53.7 mS/cm in winter and a minimum of 0.1 mS/cm in summer (Figure 3.8). Due to reduced freshwater inflow in winter the system became marine dominated as shown by these high conductivity values.

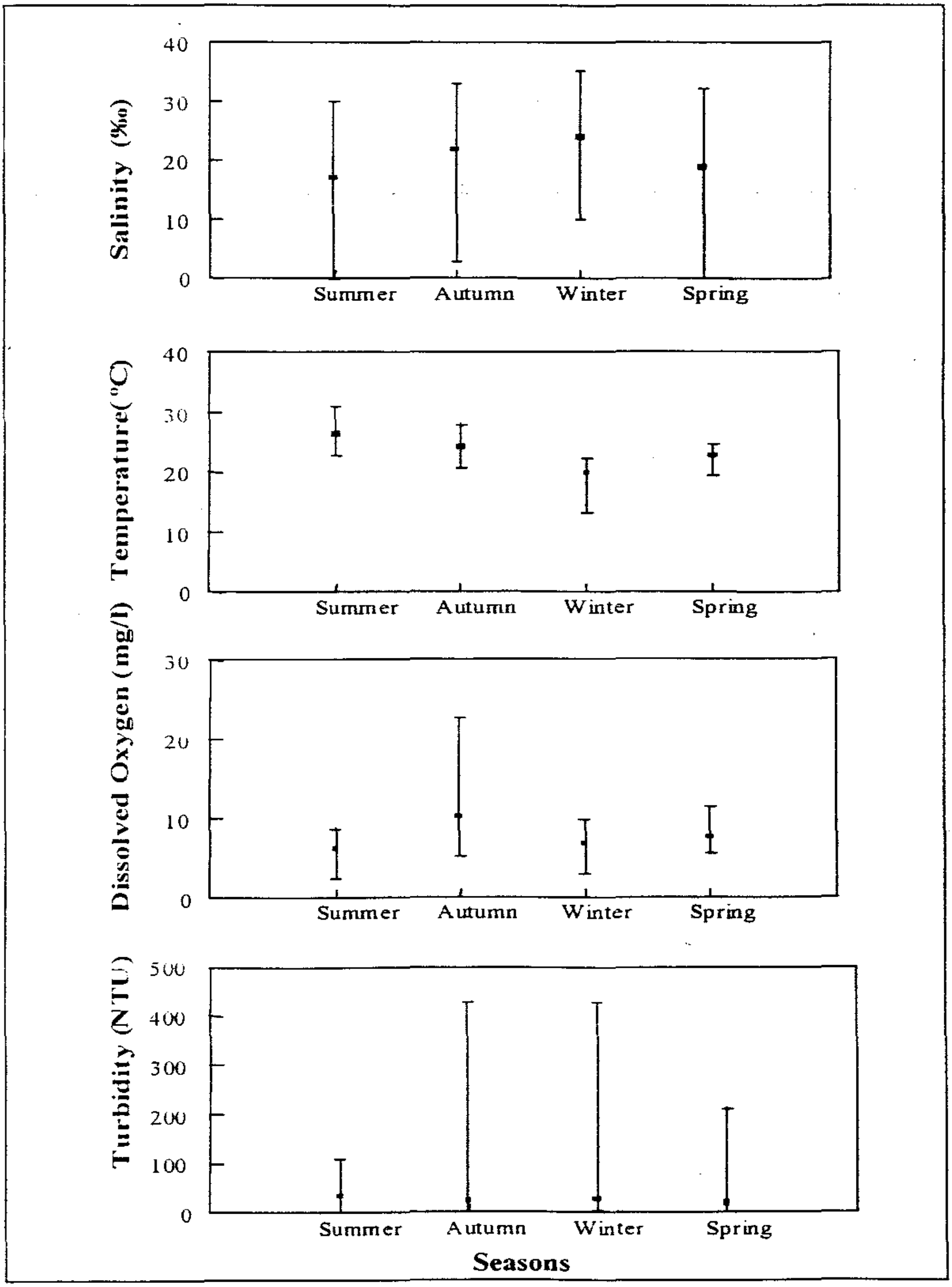

Figure 3.7. Seasonal variations in salinity, temperature, dissolved oxygen and turbidity recorded from the Mlalazi estuary during the 1989-1991 period. (Bars indicate mean and the range).

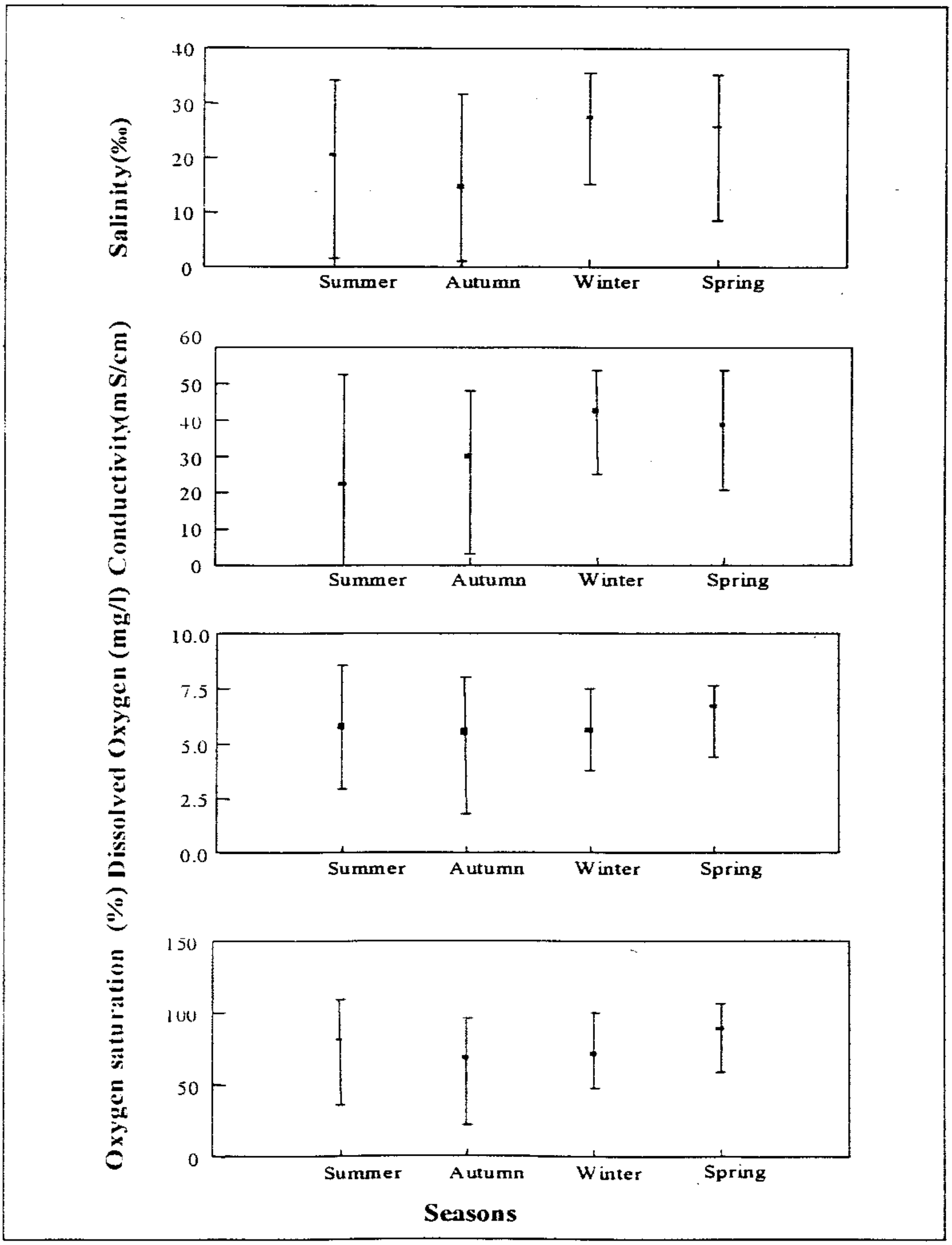

Figure 3.8 Seasonal variations in salinity, conductivity, dissolved oxygen and oxygen saturation recorded from the Mlalazi estuary during the 1999-2000 period. (Bars indicate mean and the range).

During the 1989-1991 period, a minimum dissolved oxygen concentration of 2.3 mg/l was measured in summer while a maximum of 22.6 mg/l was recorded in autumn (Figure 3.7). During the 1999-2000 period, dissolved oxygen ranged from 1.8 mg/l in autumn to 8.59 mg/l in summer. Lowest dissolved oxygen concentration measured in autumn during the 1999-2000 period corresponded to a high nutrient load measured during the same season (Figure 3.8).

Oxygen saturation ranged from 22.1 mg/l in autumn to 106.4 mg/l in spring during the 1999-2000 period (Figure 3.8). The minimum concentration recorded in autumn corresponded to a minimum dissolved oxygen concentration in autumn.

Temperature ranges showed typical seasonal variations during the 1989-1991 and 1999-2000 sampling periods. Minimum winter temperatures of 13.3 ^{0}C and maximum summer temperatures of 31 ^{0}C were recorded during the 1989-1991 period (Figure 3.7). During the 1999-2000 period, temperature ranged from 29.71 °C in summer to 18.91°C in winter (Figure 3.9).

During the 1989-1991 and 1999-2000 sampling periods, very high turbidity values were recorded in autumn and winter seasons. Turbidity values ranged from 0.8 to 427 NTU during the 1989-1991 period whereas the 1999-2000 period was less turbid with values ranging from 10.5 to 235 NTU (Figures 3.7 and 3.9, respectively).

During the 1999-2000 period, pH values ranged from 7.09 in summer to 9.45 in autumn (Figure 3.9). Since estuarine organisms prefer pH that is close to neutral, values measured during this period were considered 'normal' for estuarine waters.

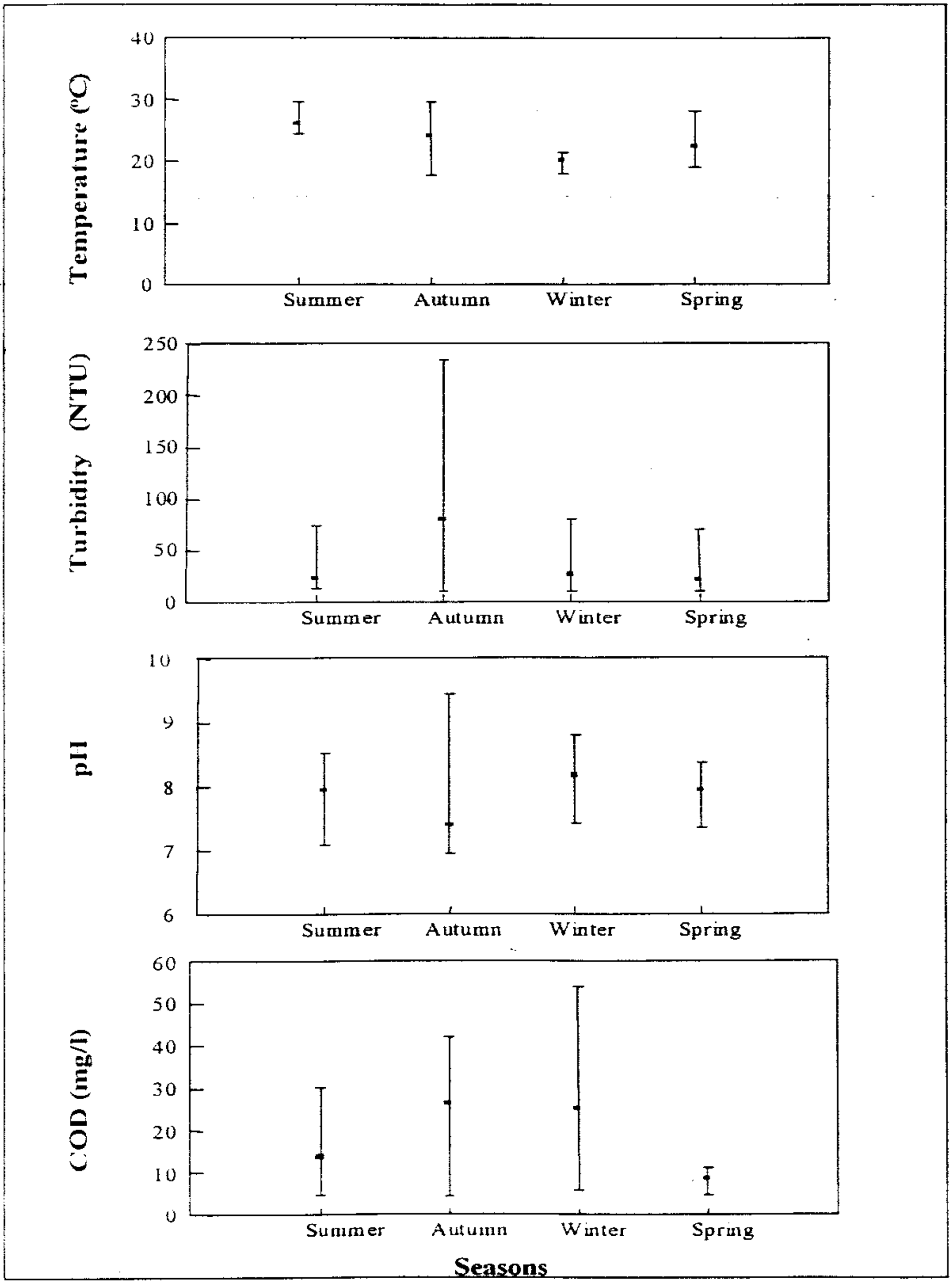

Figure 3.9. Seasonal variation of temperature, turbidity, pH, and COD recorded from the Mlalazi estuary during the 1999-2000 sampling period. (Bars indicate mean and the range).

Chemical oxygen demand reached a maximum concentration of 54 mg/l in winter during the 1999-2000 period, while a minimum of 4.5 mg/l was measured in autumn (Figure 3.9). These high COD concentrations are related to the high nutrient concentrations measured during winter, as indicated in the next section. This means that the estuary experiences greater pollution pressure during low flow conditions in winter compared to other seasons.

3.3.2.2. **Chemical variables**

Figure 3.10 represents temporal variations in nitrate, nitrite, orthophosphate, total phosphate, ammonia and chlorophyll-a concentrations measured during the 1999-2000 sampling period. Concentrations of ammonia, nitrate, nitrite and sulphate were high during autumn and winter when compared to the other seasons. Phosphates were high in spring, while chlorophyll-a reached a maximum concentration in summer.

Ammonia concentrations ranged from a minimum of 0.009 mg/l in winter to a maximum of 0.74 mg/l in autumn. Nitrate showed a concentration of less than 0.1 mg/l in spring while a maximum concentration of 6.7 mg/l was measured in autumn. Nitrite concentrations ranged from a minimum of 0.1 mg/l in spring to a maximum of 0.628 mg/l in summer. Sulphate concentrations were lowest in autumn with a minimum of 10 mg/l being measured while maximum concentrations (297 mg/l) were measured in autumn.

Concentration of less than 0.1 mg/l were measured for ortho-phosphate in summer with a maximum concentration of 6.4 mg/l being measured in summer. Total phosphate concentrations reached a maximum concentration of 2 mg/l in spring with less than of 0.1-mg/l concentration being recorded in summer.

Chlorophyll-a concentration ranged from a minimum of 0.01 mg/l measured in summer, which increased to a maximum of 154.4 mg/l also recorded in summer.

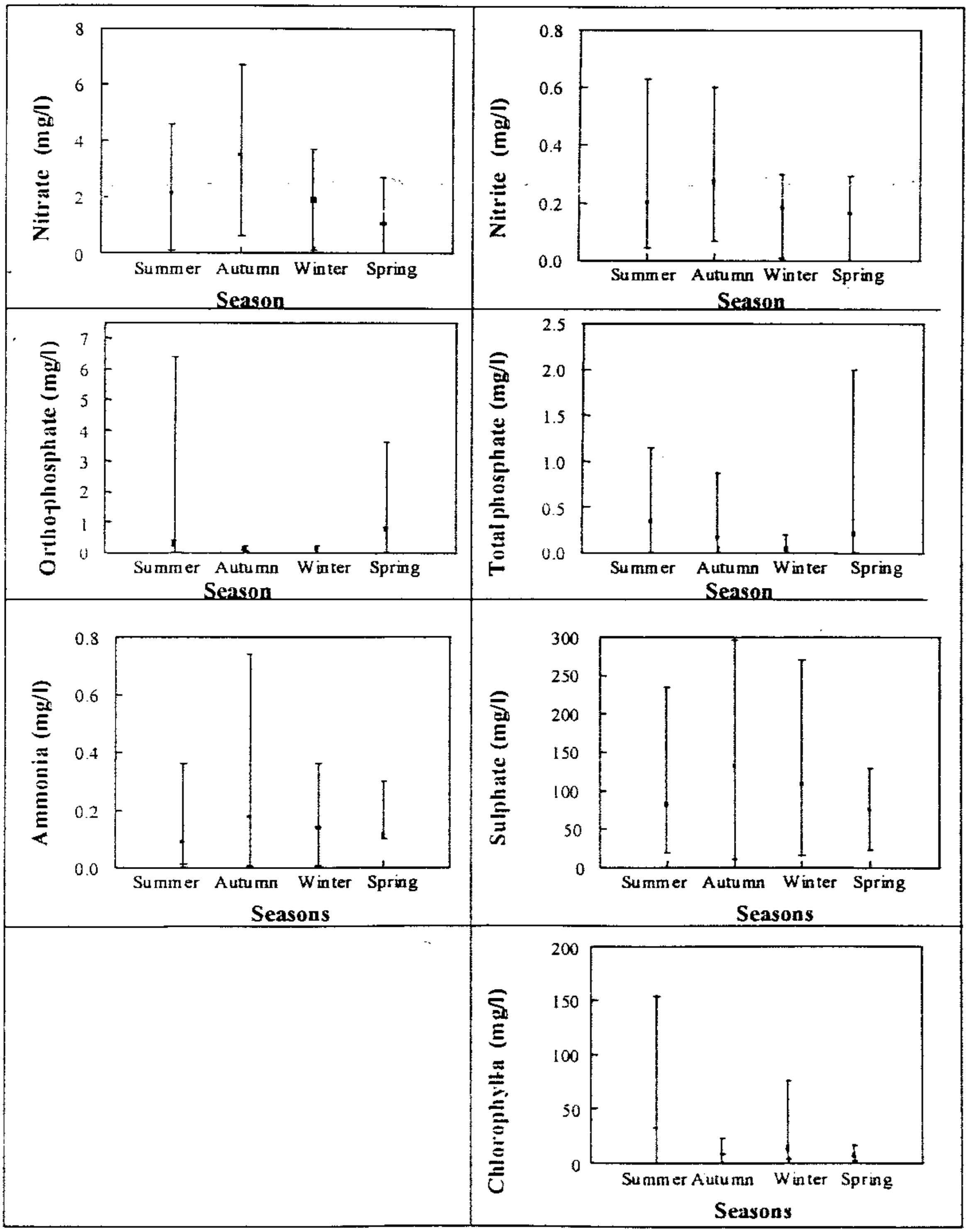

Figure 3.10. Seasonal variations of nitrate, nitrite, orthophosphate, total phosphate, ammonia, sulphate and chlorophyll-a measured from the Mlalazi estuary during the 1999-2000 period. (Bars indicate mean and the range).

3.3.3. Sediment analysis

3.3.3.1 Grain size

The sediment grain sizes recorded in a once off sediment survey conducted during the 1989-1991 period at six sampling sites are presented in Figure 3 11. During this period the phi values ranged from 1.0 at Site 2 to 4.3 at Sites 3 and 4. The upper and lower reaches (Sites 1, 2, 5 and 6) were dominated by medium sand, the middle reaches (Site 3 and 4) were dominated by silt, while the mouth region (Site 7) comprised fine sand.

The sediment grain sizes from the Mlalazi estuary during the 1999-2000 sampling period are presented in Figures 3.12 - 3.15. During spring 1999, median phi values ranged from 0.6 - 3.2 (Figure 3.12) and the type of sediments ranged from coarse to very fine sand. Coarse sand was present along the length of the estuary except at Site 4, in the middle reaches and at Site 7 (in the mouth region), which constituted fine to very fine sand. During summer, median phi values ranged from -0.2 to 2.0 (Figure 3.13) with the sediment ranging from very coarse sand, especially at Sites 1 and 3, to coarse and medium sand, with Site 7 constituting fine sand. The change in sediment type during the summer period might be attributed to scouring of the estuary bed due to heavy rains experienced in summer. The sediment constituted coarse to very fine sand during autumn with the phi values ranging from 0.0 to 3.0 (Figure 3.14). The pattern of sediment type is similar to that found during spring. Coarse sand was found at Site 1 and 2, which is the head region, and also at Site 5 and 6, the lower region of the estuary. Sediments at Site 3 constituted medium sand while Site 4 and 7 comprised fine to very fine sands. During winter, sediment type shifted to medium sands at Sites 1, 3 and 4 (Figure 3.15). The upper, i.e. Site 2 and lower reaches of the estuary, i.e. Sites 5 and 6, constituted coarse sands while Site 7 comprised fine sand. Despite slight seasonal changes, the sediment of the Mlalazi estuary during the 1999-2000 sampling periods showed a general pattern, with coarse sand in the head region, shifting to medium and fine sand in the middle reaches (Figure 3.16). The lower reaches were characterised by coarse sand with the mouth constituting fine sediments.

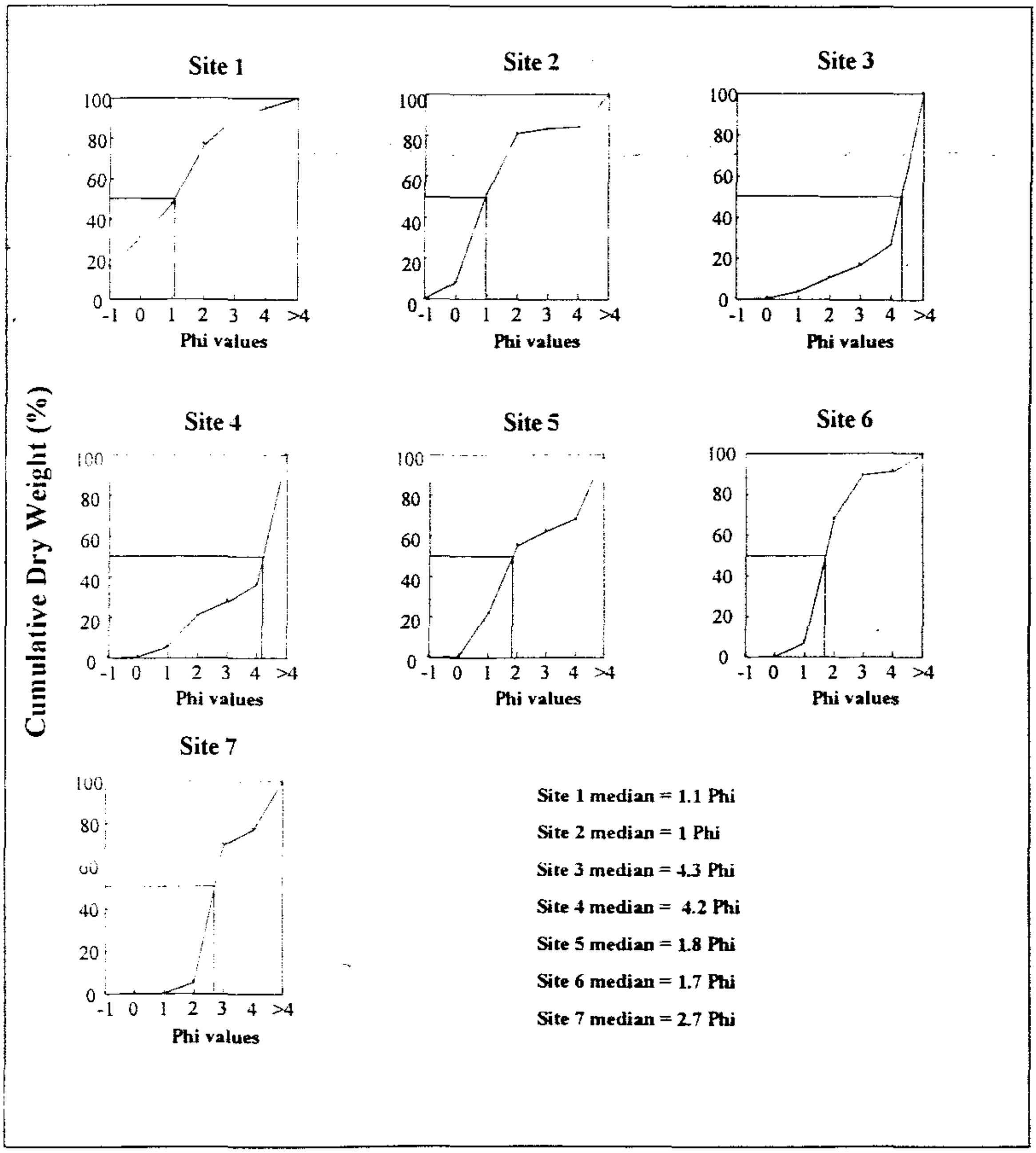

Figure 3.11. Percentage cumulative dry weight against phi values of sediment samples collected at Sites 1-7 during 1989-1991 sampling period. Lines drawn at 50% cumulative weights represent the median particle diameter for a particular site.

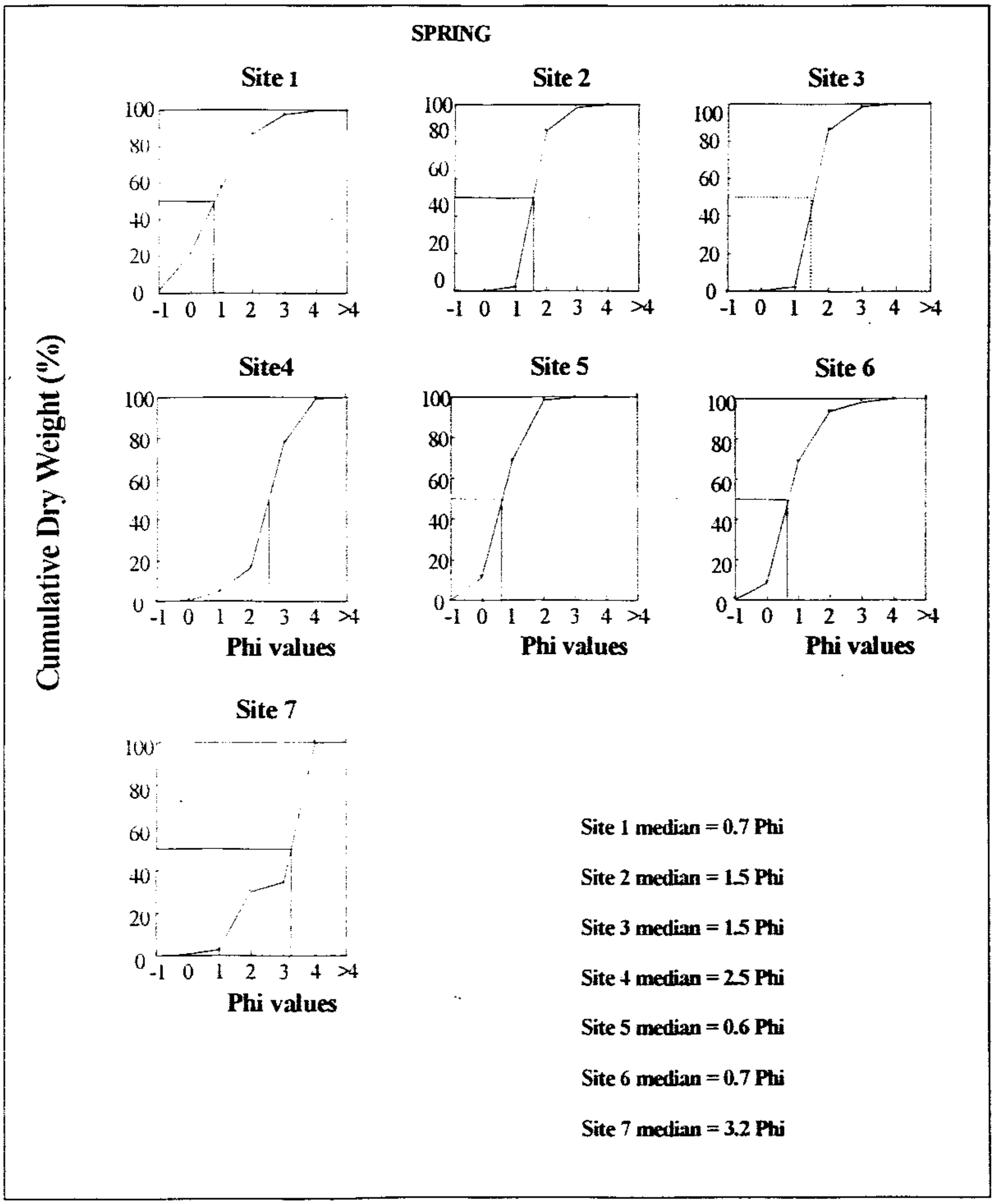

Figure 3.12. Percentage cumulative dry weight against phi values of sediment samples collected at Sites 1-7 during spring of the 1999-2000 period. Lines drawn at 50% cumulative weights represent the median particle diameter for a particular site.

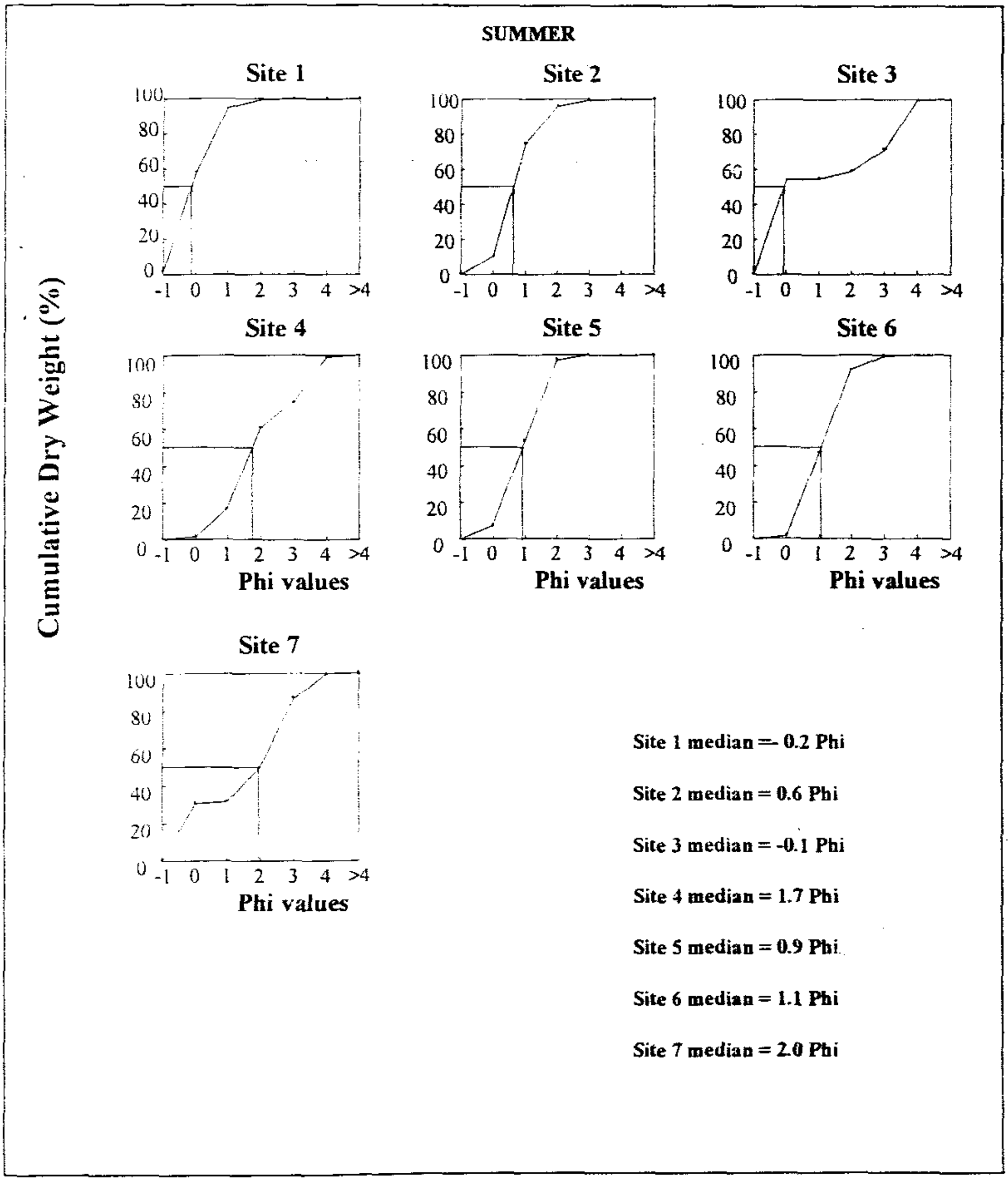

Figure 3.13. Percentage cumulative dry weight against phi values of sediment samples collected at Sites 1-7 during summer of the 1999-2000 period. Lines drawn at 50% cumulative weights represent the median particle diameter for a particular site.

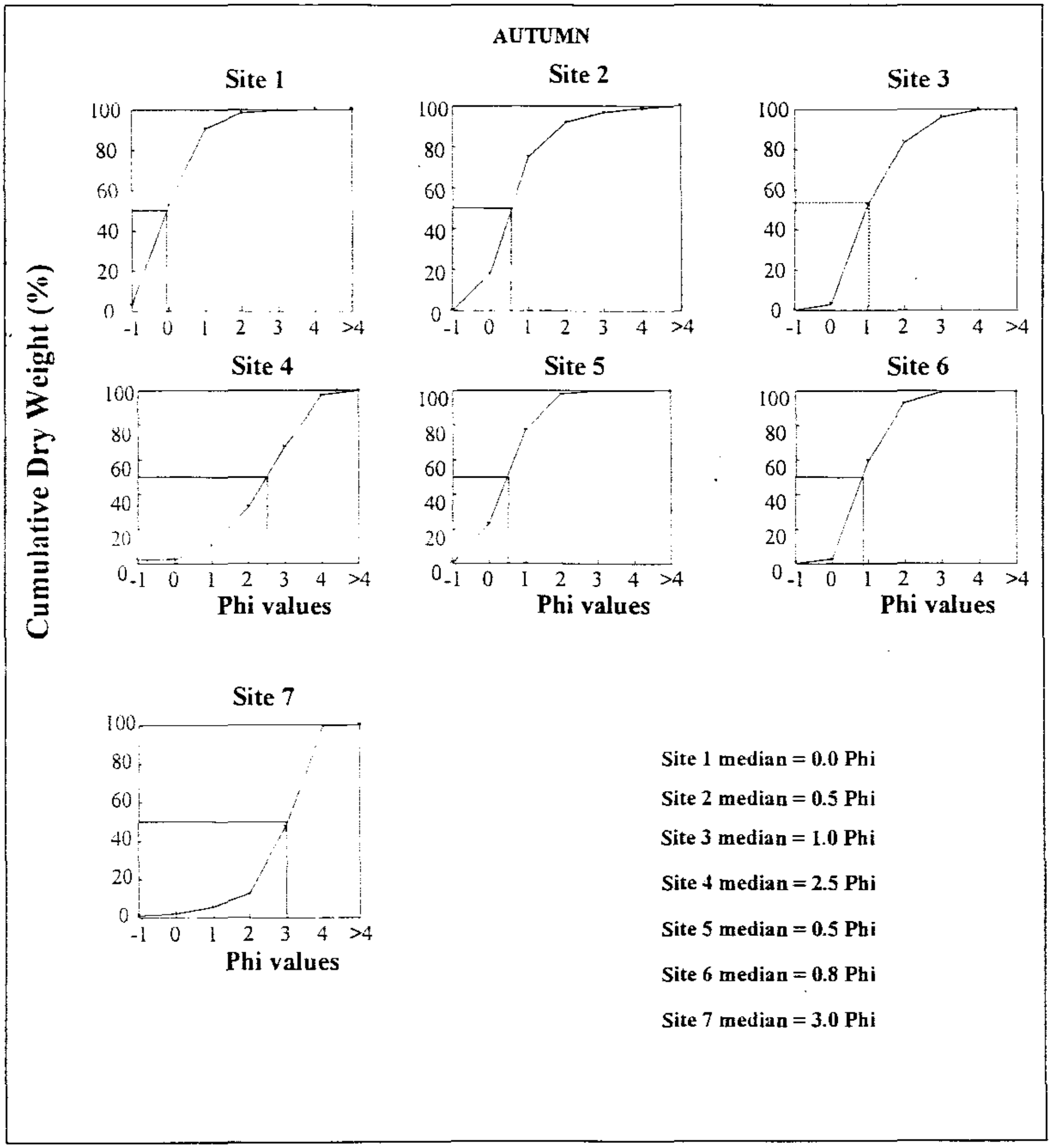

Figure 3.14. Percentage cumulative dry weight against phi values of sediment samples collected at Sites 1-7 during autumn of the 1999-2000 period. Lines drawn at 50% cumulative weights represent the median particle diameter for a particular site.

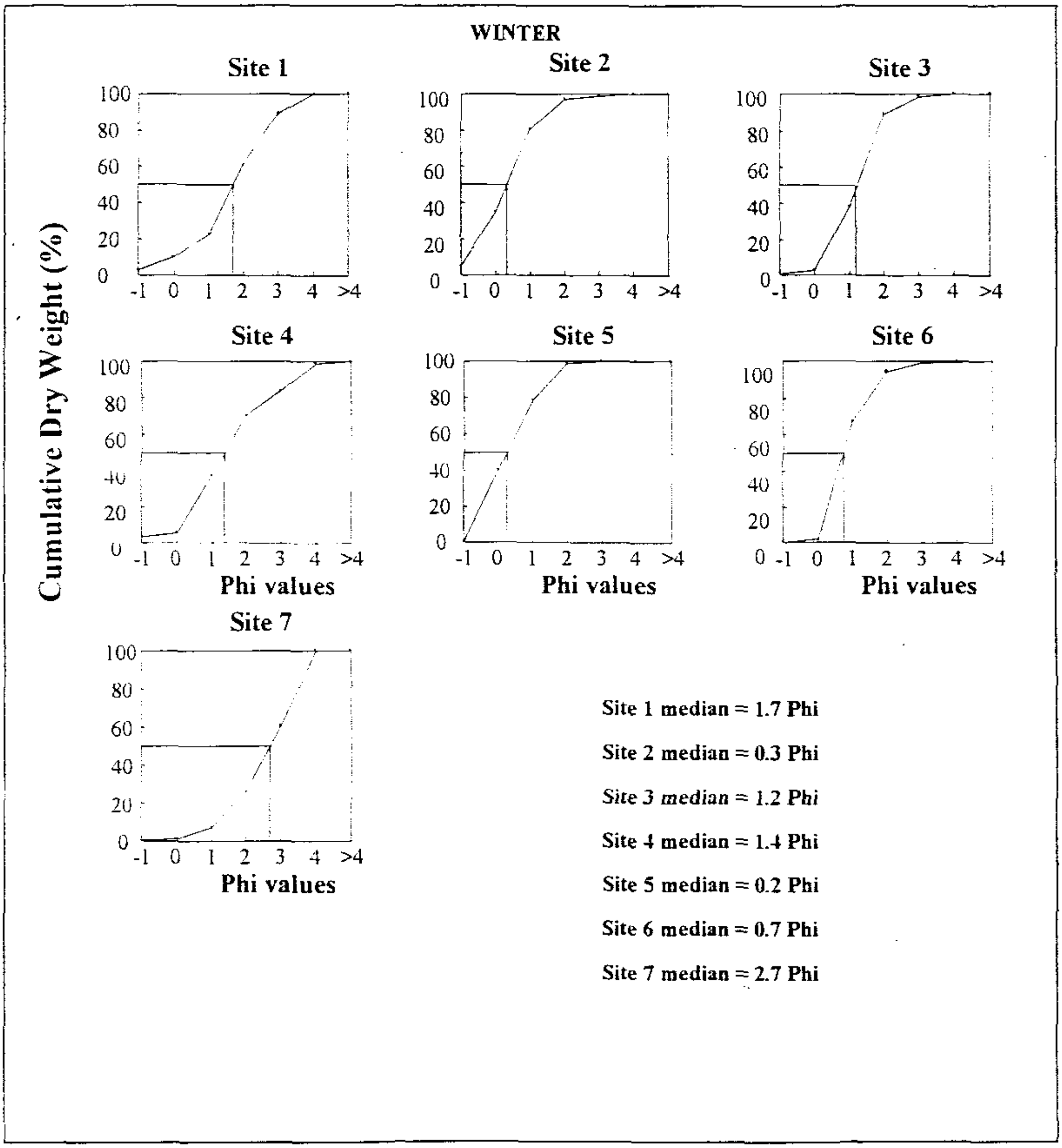

Figure 3.15. Percentage cumulative dry weight against phi values of sediment samples collected at Sites 1-7 during winter of the 1999-2000 period. Lines drawn at 50% cumulative weights represent the median particle diameter for a particular site.

3.3.3.2 Sorting co-efficient

The sorting co-efficient results for the 1989-1991 sampling period are presented in Figure 3.17 whereas those of the 1999-2000 period are presented in Figure 3.18. Sediments of the estuary during the 1989-1991 period were poorly sorted except at the lower and mouth regions, where sediments were moderately sorted (Figure 3.17). This indicated that the sediment during this period was not evenly distributed in terms of grain sizes. The mean sorting coefficient values showed that during the 1999-2000 period the sediments were moderately sorted along the length of estuary except at the mouth region which had sediments that were poorly sorted (Figure 3.18).

3.3.3.3. Organic content of the sediment

The percentage organic content of the Mlalazi estuary sediment samples recorded during the 1999-2000 period is presented in Figures 3.19. Spatial analysis revealed a high organic content of 13.49 % at Site 4 and 8.02 % at Site 7. Mud sediments have high organic content compared to other types of sediments and Site 4 was characterised by a muddy substratum. The sites that were characterised by course to medium sediments (Sites 1, 2, 3, 5 and 6) had low to moderately low organic content.

Temporal variations indicated highest organic content during autumn (13.49 %) decreasing to a minimum of 0.8 % in spring (Figure 3.19). The high organic content in autumn corresponded to heavy rains during this season, which seemed to have resulted in a large input of organic matter into the estuary. The other seasons ranged from moderately low to medium percentage of organic content.

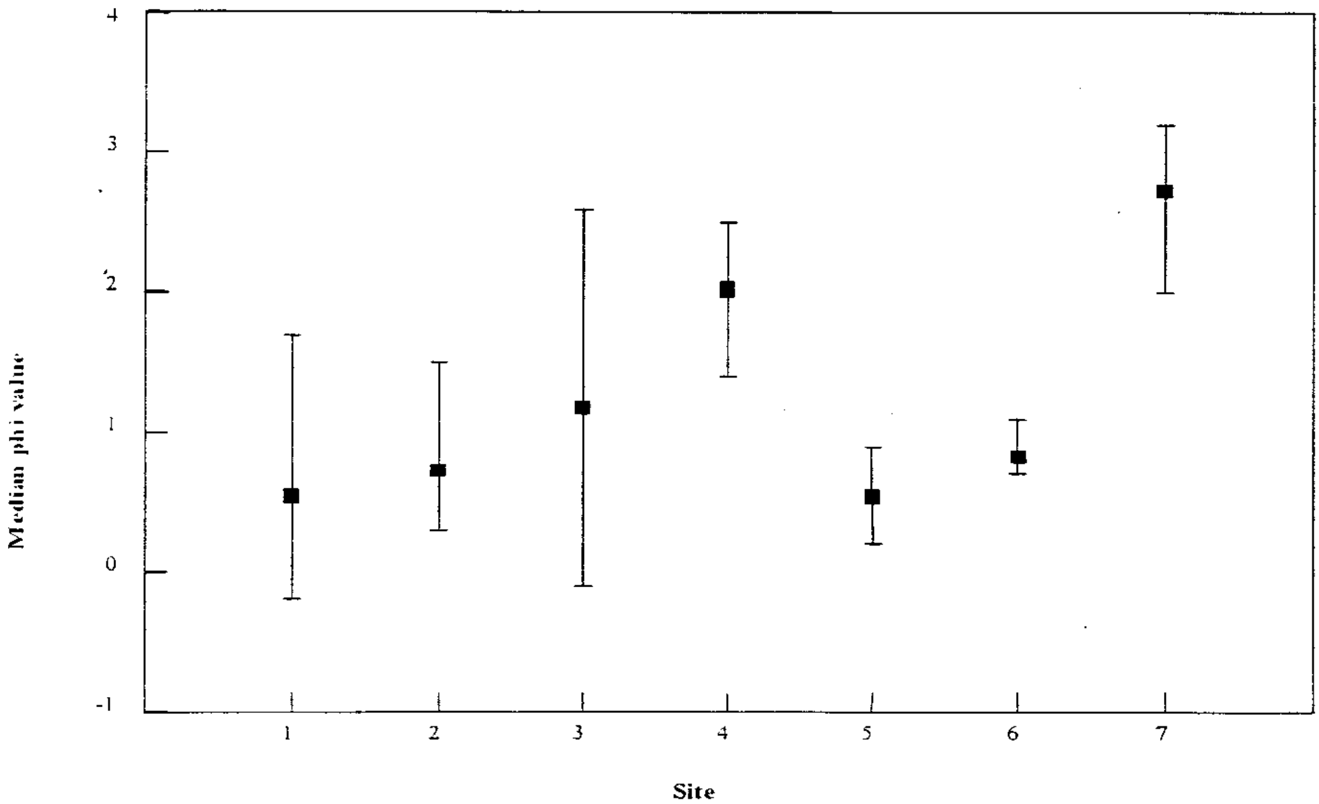

Figure 3.16. Median phi values of sediment samples collected from the Mlalazi estuary during the 1999-2000 sampling period. (Bars indicate mean and the range).

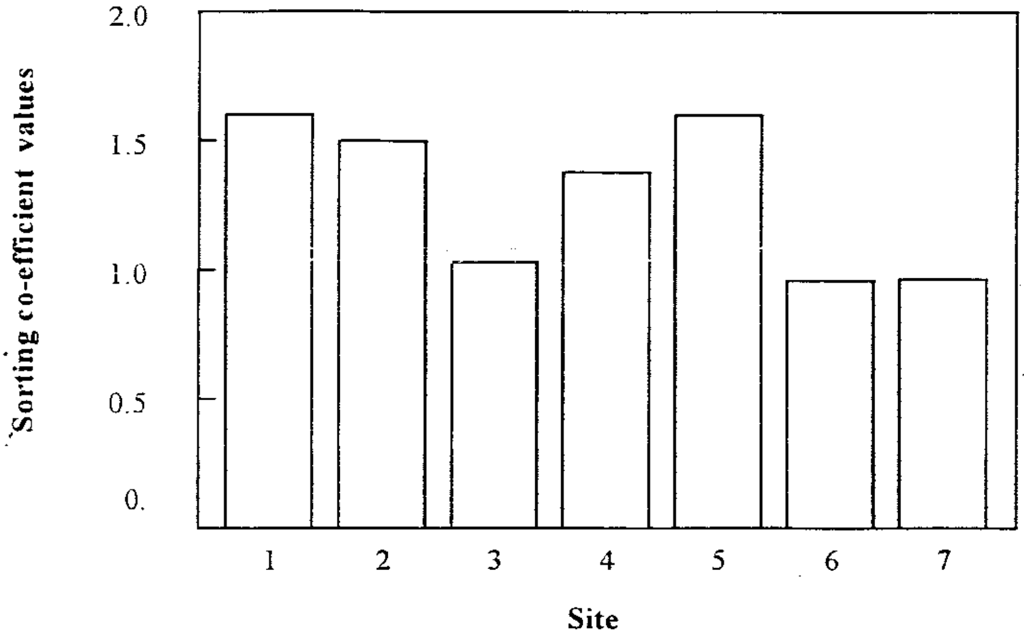

Figure 3.17. Sorting coefficient of sediment samples collected during a once off survey from the Mlalazi estuary during the 1989-1991 period.

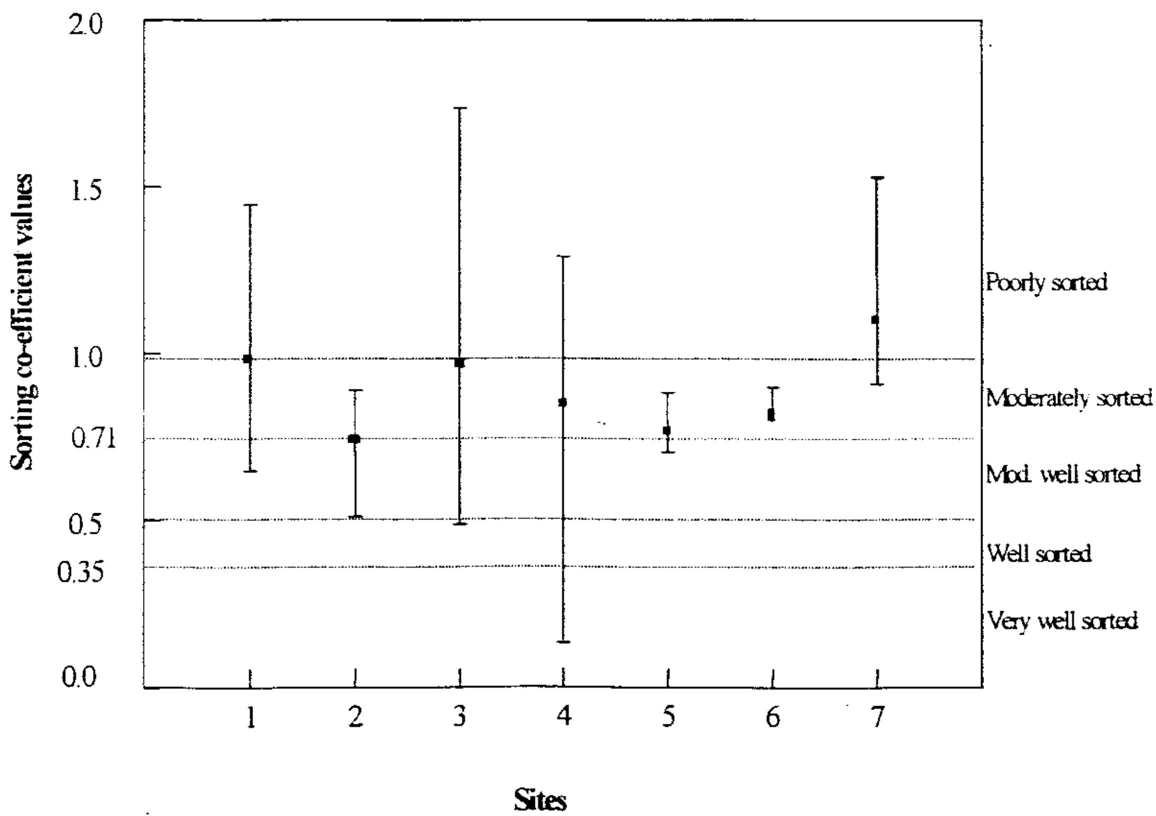

Figure 3.18. Sorting coefficients of sediment samples collected from the Mlalazi estuary during 1999-2000 period. Sorting classes on the graph are: <0.35 = very well sorted, 0.35-0.5 – well sorted, 0.50-0.71 = moderately well sorted, 0.71-1 = moderately sorted and 1-2 = poorly sorted. (Bars indicate the mean and the range).

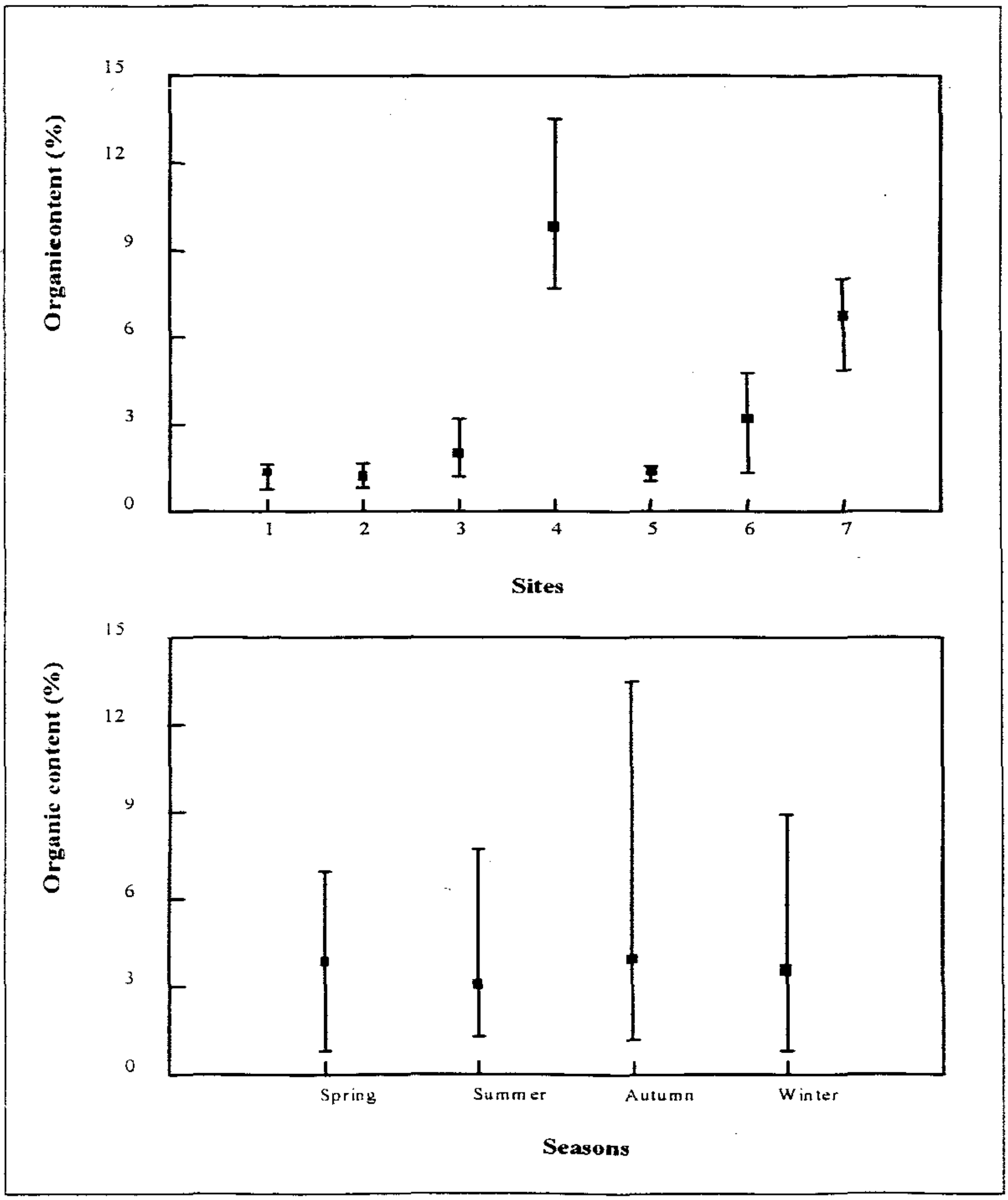

Figure 3.19. Spatial and temporal variation in organic content measured in sediment samples collected from the Mlalazi estuary during the 1999-2000 period. (Bars indicate mean and the range).

3.4. Discussion

The estuarine ecosystem is characterised by physico-chemical properties supporting a community of organisms adapted and interacting in such a way that there is transport of energy and materials through the system (Branch & Branch 1981). The estuary is thus viewed in terms of abiotic factors such as temperature, pH, inorganic nutrients and biotic factors. Because of an increase in anthropogenic pressure on the coastal zone, natural cyclic changes in the physical and chemical conditions of the estuary are often adversely affected. These changes affect the water quality of the estuary, which in turn can lead to changes in biotic communities inhabiting the estuary.

According to DWAF (1996), the quality of many water resources in the Republic of South Africa is declining. This is primarily as a result of anthropogenic activities such as industrial practices and agriculture, which often release effluents containing substances into estuaries, resulting in deleterious effects on living resources and impairing the water quality. Depending on the nature of the substance and its concentration, effects may range from negligible to disastrous. Control and management of pollution is therefore essential for the protection of the aquatic environment, and considerations in estuarine management should strive to ensure that the quality of water does not deteriorate beyond acceptable levels. This can only be done by regularly monitoring the physico-chemical parameters of the aquatic body so as to determine the current state of the water quality of the system.

3.4.1. Physical variables

Salinity is a measure of the total quantity of dissolved salts in the water body and is one of the most important characteristics of the estuarine environment (Ketchum 1983). Salinity in a typical estuary usually increases from virtually 0‰ in the river water to 35‰ in the seawater (Day 1981). This trend was been observed in the Mlalazi estuary during both the 1989-1991 and 1999-2000 sampling periods where spatial differences were found along the length of the estuary, with low salinities in the head region and more saline waters in the mouth region. Hill (1966) recorded a salinity range of 25-34‰ in the mouth region decreasing to 0.5‰ in the head region of the Mlalazi estuary.

The salinity may also vary vertically due to stratification, with the water being fresher at the surface and more saline at the bottom. In the Mlalazi estuary, during both sampling periods, bottom salinities were higher than those at the surface due to the intrusion of denser saline waters from the ocean, thus forming vertical stratification. Hill (1966) also recorded these marked vertical salinity gradients in the estuary.

Conditions in southern African estuaries are variable, with many drainage basins showing a markedly seasonal rainfall pattern with heavy winter rains in the western Cape and heavy summer rains in KwaZulu-Natal and Mozambique (Branch & Branch 1981). Due to different rainfall patterns, river discharge may vary greatly, with the estuary being river dominated during one season and marine dominated during another. Differences between seasons may also occur due to evaporation because of an increase in temperature. Subtropical surface waters are usually characterised by a relatively high salinity level (30-35 ‰) caused by greater evaporation rates, however, the input of freshwater due to rains reduce salinity in summer (DWAF 1995). In a system such as the Mlalazi estuary, dilution due to rain seemed to account for most of the changes in salinity patterns. The low salinities during summer and spring of the 1989-1991 period were attributed to summer rains, which characterise the KwaZulu-Natal region. This was also the case during autumn of the 1999-2000 sampling period, when salinities ranged between 1-2 ‰ at all sites. This decline was offset by the much higher salinities recorded during winter.

The conductivity of water refers to its ability to conduct an electrical current and is measured as the total amount of dissolved anions and cations in water (Emmel 1973). The most common dissolved substances are usually cations Na^+, K^+, Ca^{2+} and anions $NHCO_3^-$, CO_3^{2-}, Cl^-. Conductivity is directly proportional to salinity, i.e. as salinity increases, conductivity also increases. Conductivity of the estuary increased towards the mouth during 1989-1991 and 1999-2000 with bottom concentrations consistently higher than those at the surface. The influence of marine waters is usually reduced in the upper reaches with freshwater influence decreasing towards the mouth region. A relationship between conductivity and turbidity has also been found where an increase in cation concentration indicated by increased electrical conductivity enhance flocculation of clay particles, thus increasing water transparency (Scharler, *et. al* 1997). In the Mlalazi estuary, a trend that supports the above statement was found during the 1999-2000 sampling period. In the mouth region, where conductivity was high, turbidity was lower than in the head region, where conductivity was low.

During the 1989-1991 sampling period, conductivity, TSS and TDS, were only measured during August 1990 and February 1991. Highest values of these variables were measured during winter (August 1990), after which the levels decreased markedly in summer (February 1991). During winter, less freshwater enters the estuary due to reduced rainfall, and therefore the system becomes marine dominated. The high levels of conductivity, TSS and TDS indicated that the estuary was marine dominated during August 1991 compared to February 1991. High conductivities were recorded during winter of the 1999-2000 period, which then decreased during summer. In aquatic bodies, conductivity under natural conditions should not exceed 450 mS/cm for optimal functioning of the system (Willoughby 1976). In the Mlalazi estuary during the 1989-1991 period, conductivity reached a maximum of 2 820 mS/cm in winter. It is suggested that these extremely high levels of conductivity during this period were caused by elevated concentrations of some elements combined with reduced freshwater inflow into the system. During the 1999-2000 period, conductivity ranged between 6.93 to 52.7 mS/cm and thus fell within the range for optimal functioning of the system. Generally, conductivity levels in the Mlalazi estuary indicated a brackish/marine situation due to influence of both the river and the sea.

Most aquatic organisms have a particular temperature range at which optimal growth, reproduction and general fitness for the environment occur. If the temperature goes any lower or higher than the optimal range, many organisms become unproductive and may migrate to a more suitable environment, or die (Whitfield 1992). The temperature of an estuary usually follows a fairly predictable seasonal pattern with some variations within each season depending on river flow and weather conditions. Temperature fluctuations in the estuary tend to be greater than those of either the river or the open ocean since inflowing river water tends to be colder than seawater in winter and warmer in summer. The Mlalazi estuary is situated in the coastal zone that ranges from the Mbashe estuary in the Transkei to Maputo Bay and is influenced by a subtropical climate where estuarine water temperatures are high in summer (28°C) and lower in winter (14°C) (Whitfield 1992). The results of the study indicated seasonal variations during the 1989-1991 and

1999-2000 sampling periods, which followed the typical seasonal trend. Cyrus (1988) reported on temperature ranges of a number of estuaries in KwaZulu-Natal and found that they show typical seasonal variations with ranges close to those recorded in the Mlalazi estuary. Since the Mlalazi estuary is a shallow estuary (Day 1981), temperature values recorded during both sampling periods indicated no thermal stratification. On the whole, the effects of temperature changes are much smaller than those of salinity in estuaries.

pH expresses the molar concentration of H^+ ions in the water and the pH scale is based on a negative logarithm [$pH = -\log (H^+)$]. It is an important indicator used in evaluation of water quality (Novotny & Olem 1994). The higher the concentration of hydrogen ions, the more acidic the water, and more hydroxide ions within the water, the more basic it will be. Human impacts causing reduced pH levels are basically ascribed to acidic point-source effluents from industries entering the aquatic body. Events leading to an increased water pH are less common and are generally due to alkaline effluents from industries and anthropogenic eutrophication. Alabaster and Lloyd (1980) stated that higher pH levels are not as great a threat to aquatic organisms as are low pH levels. Increased pH is however important as it creates more favourable conditions for algal blooms, increased aquatic weed growth and is thus a concern in areas with nutrient enrichment. Lusher (1984) states that for maintenance of aquatic life, pH should lie within the range of 7.3 - 8.6 since most aquatic organisms prefer a pH near neutral, however they can withstand a pH in a range of about 6 - 8.6. DWAF (1995) reported a pH range of 7.9-8.2 for marine coastal waters. The pH in the Mlalazi estuary ranged between 7 - 9.45 throughout the study periods. The 9.45 pH value was higher than normal for estuarine waters and it has been stated above that higher pH values in estuarine waters cause conditions that are favourable for algal blooms. It seems this high pH value indicates conditions that favour algal blooms in the estuary, i.e. eutrophication.

Oxygen concentration is one of the most important abiotic determinants of survival of most aquatic organisms (Head 1970). It is a measure of the amount of oxygen present in water and available for respiration. Various factors determine the amount of oxygen that dissolves in water. These include the rate of aeration from the atmosphere, temperature, salinity, and respiration by all organisms and photosynthesis by plants (Dallas, Day & Reynolds 1994). Ideally, dissolved oxygen concentration in water should approach saturation, however it can be significantly reduced by oxygen consuming chemicals, effluents, high levels of organic waste and bacterial decomposition which consumes oxygen. According to Novotny and Olem (1994), for the maintenance and protection of biological resources of aquatic ecosystems, particularly estuaries, oxygen should not fall below 4 mg/l. For the South African coastal zone, it has been targeted that for the East Coast the dissolved oxygen should not fall below 5 mg/l (DWAF 1995). Many estuarine habitats experience periods of oxygen deprivation and the oxygen concentration on the bottom is often lower than in water above (Ketchum 1983). This may result from heavier, more saline water wedging underneath the less saline surface water from fresh water run-offs (Ketchum 1983). The amount of dissolved oxygen in water can also be reduced by increases in temperature and salinity (Dallas, *et. al* 1994, and DWAF 1995). In the Mlalazi estuary concentrations of below 4 mg/l were recorded at Sites 2 and 3 during winter of the 1989-1991 period, which corresponded to high salinity values recorded at these two sites. Site 3 was also characterised by silty sediments and high turbidity levels, which exacerbates hypoxic conditions within the estuary (Heinis, *et. al* 1994, in Mackay 1996, Pearce & Schumann 1997). This is caused by a drop in light penetration resulting in a reduction in the rate of photosynthesis of aquatic plants with a corresponding reduction in oxygen replenishment. During the 1999-2000 sampling period, low oxygen concentrations were also observed in the muddy substrate at Site 4. Mackay (1996) found oxygen levels in the Siyaya estuary to be more related to the nature of the substrate than changes in salinity. Those sites that were characterised by detritus and fine-grained sediments denoted anaerobic conditions. The hypoxic conditions experienced in the Gamtoos estuary mostly occurred in bottom waters of the upper and middle reaches (Pearce & Schumann 1997). This was attributed to the restricted movement of tidal water upstream of the shallow system.

The two sampling periods showed that the middle reaches was less oxygenated than the other areas. This was attributed to the nature of the substrate, high turbidity values and elevated levels of nutrients in this area. The high nutrient load seemed responsible for the slight reduction in oxygen levels during the 1999-2000 period and this is evident from the high COD values measured in the estuary. Reyes and Merion (1991) reported on the low oxygen conditions in the shallow Borjorquez Lagoon, Mexico. They found that periods of low oxygen were experienced following introduction and resuspension of nutrients.

Turbidity refers to the optical property that causes light to be scattered and absorbed rather than transmitted in straight lines through a water sample (Dallas, *et. al* 1994) and is measured as NTU's (Nephelometric Turbidity Units). Any suspended matter, including silt, clay and finely dissolved organic and inorganic matter causes the turbidity of water to increase. Floods, discharges from industries and other anthropogenic activities also cause increased turbidity of water. The values of turbidity are often highest in the upper muddy reaches of estuaries and increase with depth and discharge of the river. In an estuary, the amount of material in suspension generally decreases from the upper reaches to the mouth. This is due to a slackening in velocity and carrying capacity of inflowing river water and the electrolytic effect of seawater (Day 1981). The latter process involves the coagulation of negatively charged particles of colloidal silt by positive ions of certain metals present in seawater. Water with turbidity above 80 NTU is usually regarded as turbid and such high turbidities have been recognised as adversely affecting aquatic life (Cyrus 1988). There were fluctuations in turbidity levels during both sampling periods. Turbidity values in this study were greater in the middle reaches with clearest waters occurring towards the mouth. The volume of silt transported into estuaries by rivers fluctuates seasonally with the maximum discharge taking place during the wet season (Day 1981). Very high turbidities, i.e. above 80 NTU, were recorded at Sites 3 and 4 during autumn and winter of the 1989-1991 and also during autumn 1999-2000 at all sites. The increase in turbidity during the 1999-2000 period was due to an increased freshwater discharge due to heavy rains that occurred during the month of May 2000. This is supported by the fact that during the same month the estuary was fresh at all sites with salinities ranging from 1-2‰ along the length of the estuary. The increase in turbidity at Sites 3 and 4 during the 1989-1991 period was attributed to the muddy substrate in that area. In general mean turbidities during both sampling periods were slightly higher than the mean turbidity of 25,5 NTU recorded in the Mlalazi estuary by Cyrus (1988). Turbidity did not differ markedly between top and bottom measurements because of the shallowness of the estuary.

Turbidity has been viewed as a diagnostic property of water that may reveal important information about the water quality, changes, origins and ecological impacts (Wilber 1983). Eroded soils produce the most important type of suspended solids. Land-use practises such as overgrazing, non-contour ploughing and removal of riparian vegetation accelerate the rate of erosion (Dallas, *et. al* 1994). Sand, silt, and clay are discharged by rainfall and riverflow and carried into estuaries from rural and agricultural areas. Sediment resuspended in the course of the river (bed load) is also an important type of suspended solids in estuaries. There is a growing understanding of the production of turbidity by human activities in estuaries. Various anthropogenic activities such as the release of domestic sewage or industrial discharge, physical perturbations (e.g. road and bridge construction) and bad catchment management have been implicated in increasing turbidity in estuaries (Dallas, *et. al* 1994, Mackay 1996). The agricultural practices, discharge of the prawn farm effluents and bad catchment management may have resulted in high turbidity values recorded in the middle reaches of the Mlalazi estuary during the study periods. Wepener and Vermeulen (2000) reported an increase in turbidity values in the Mhlathuze estuary as compared to the historical data. This was attributed to the deposition of sediments from the catchment in the estuary. Continuously high levels of turbidity in estuaries may have serious consequences on estuarine biota whereby some organisms are smothered and communities may be dominated by organisms that are best able to cope with this alteration in habitat, resulting in low diversity.

Most benthic organisms are relatively non-mobile due to their burrowing nature and as such cannot move away from highly turbid areas, a factor that could be highly detrimental to them (Day 1981). Filter feeding organisms such as bivalves are mostly affected by high turbidities since their gills become clogged when excessive fine suspended matter is present. However, in some instances increased turbidity can be favourable to aquatic ecosystem by reducing algal blooms and increasing protection from predators. High turbidities affect light penetration, thereby inhibiting photosynthesis and production of plants.

3.4.2. **Chemical variables**

Estuaries are the most productive ecosystems in the world because they are shallow, well mixed and receive nutrients from rivers, coastal waters and adjacent marshes and swamps (Branch & Branch 1981). Elements such as phosphorus and nitrogen are essential plant nutrients required for growth of algae, and limitations in amounts of these elements are usually the factors that control the rate of algae growth (Head 1970). Nutrients are derived mainly from the river, with small supplies coming from the sea as well as traces of nitrogen from rain. Concentrations of nutrients in estuaries vary on a temporal and spatial scale due to inputs from rivers and the ocean as well as biological uptake and regeneration.

Nutrients may be utilised and regenerated to various extents along the length of an estuary and the estuary may act as a source or a sink of nutrients to adjacent ecosystems (Head 1970, Scharler, *et. al* 1997). Alterations to the nutrient status of an estuary are often brought about by anthropogenic influences such as point source effluents from industries, intensive aquaculture or diffuse sources such as agricultural run-offs. The continued development of shrimp aquaculture in Mexico has created symptoms of negative environmental impacts, mainly due to the discharge of nutrients and organic matter into adjacent coastal waters (Paez-Osuna, *et. al* 1998). It has been suggested prior to the start of this study that the Mtunzini prawn farm, which is situated adjacent to the estuary, discharges effluent containing high levels of nutrients into the estuary, thus posing a eutrophication problem. One of the objectives of this study was to look at the water quality of the Mlalazi estuary in terms of the probable eutrophication effect of the prawn farm. Eutrophication refers to the enrichment of the aquatic system by nutrient elements notably nitrogen and phosphorus (Emmel 1973). Eutrophication of coastal waters as a result of human activity is widely recognised as a major pollution threat (Willoughby 1976). The increased anthropogenic source of nutrients interferes with the natural nutrient cycles and can artificially enhance primary production. At first sight, eutrophication may seem a positive occurrence, as plants need nutrients and the entire food chain benefits. Instead, eutrophication causes algal blooms leading to depletion of oxygen, which is detrimental to biological resources. The following are some of the several detrimental consequences of eutrophication (Scharler, *et. al* 1997):

- Algal mats, decaying algal clumps, odours and discoloration of the water, which can interfere with recreational and aesthetic water uses.
- Extensive growth of rooted aquatic macrophytes interferes with aeration.
- Dead macrophytes and phytoplankton settle on the bottom of a water body, stimulating microbial breakdown processes that require oxygen. Eventually oxygen will be depleted and such anoxic conditions results in the death of organisms.
- Algal blooms shade submerged aquatic vegetation, reducing or eliminating photosynthesis and productivity.

Phosphorus is an essential nutrient for aquatic life and it enters the estuary either from weathering of soil and rocks, point-source discharges such as sewage plants and agricultural run-offs. In water, phosphorus is present in a dissolved phase as orthophosphate that is immediately available by natural processes to aquatic biota (Ketchum 1983). In most waters, phosphorus functions as a growth limiting factor because it is usually present in very low concentrations. Algae and other plants rapidly take up any free phosphorus in the form of inorganic phosphates. Because algae only require small amounts of phosphorus to survive, excess phosphorus may cause extensive algal blooms, leading to water quality deterioration. Ketchum (1983) has given a phosphate standard of 1 mg/l for maintenance of aquatic bodies. EPA (1992) has recommended that the level of total phosphate in estuaries and coastal systems should be between 0.01 to 0.1 mg/l to avoid algal blooms and a decrease in organism diversity. DWAF (1995) estimated a mean of 0.062 mg/l for coastal marine waters. Orthophosphates and total phosphate concentrations in the Mlalazi estuary were higher in summer when compared to winter during the 1989-1991 period. Although the increased concentration of phosphate during summer 1989-1991 suggests that the anthropogenic source of phosphate was in the form of agricultural runoff, such high concentrations are often difficult to explain. The subtropical summer season is characterised by heavy rainfall, which increases return flows from agricultural areas and rural settlements in the area. It has been reported that agriculture affects the concentration of nutrients in estuaries (Dallas & Day 1993, Mackay 1996). An improvement in land-use practices in the catchment of the Siyaya estuary, following consultations with farmers and the local community, resulted in a decrease in phosphorus levels entering the estuary compared to concentrations reported in previous studies (Mackay 1996).

The high orthophosphate and total phosphate concentrations recorded during the 1999-2000 period in the prawn farm effluent and at Site 3, suggested that the prawn farm had become the major source of phosphates in the Mlalazi estuary. The high phosphate concentrations measured in the estuary were above the standards given by EPA (1992) and Ketchum (1983) recommended for the maintenance of estuaries, and those given by DWAF (1995) for marine waters. This emphasises the extent to which the prawn farm has become the anthropogenic source of increased phosphate levels in the estuary. This concern about the prawn farm effluent must be addressed in order to prevent a future eutrophication problem, which would eventually lead to a severe degradation of the estuarine environment.

Nitrogen occurs abundantly in natural systems where it occurs as nitrate, nitrite and ammonia (Dallas, *et. al* 1994). Many natural water bodies are nitrogen limited, and as such even small changes in biologically available nitrogen levels can affect plants and eventually animal life. Nitrite (NO_2), an inorganic ion occurring naturally as part of the nitrogen cycle, is an intermediate in the conversion of ammonia and nitrate. It is a naturally occurring anion in saline waters and is generally present in trace quantities because of the rapidity of nitrification (Head 1970). Nitrate (NO_3), which is an ionised form of nitrogen, may enter aquatic bodies via agricultural run-off and is seldom abundant in water because it is incorporated into organic nitrogen in plant cells (Dallas, *et. al* 1994). Nitrate (NO_3) is related to ammonia in that nitrifying bacteria convert ammonia to nitrate, which is less toxic to animal life than nitrite. EPA (1992) has recommended that no more than 10 mg/l of nitrate be permitted in receiving waters since high concentrations can stimulate overgrowth of macrophytes and phytoplankton. Nitrite seldom appears in concentrations of greater than 1 mg/l even in waste effluents (Sawyer & McCarty 1989).

During the 1999-2000 sampling period, the prawn farm effluent showed very high concentrations of nitrate and nitrite. This showed that the Mlalazi River is not the primary source of nitrogen for the estuary and that the prawn farm acts as a point source in the estuary suggesting that the farm is responsible for the bulk of the nitrogen load in the estuary. Site 3, adjacent to the outlet, also showed high concentrations of these nutrients compared to other sites sampled. The maximum concentrations of nitrate and nitrite in the prawn farm effluent were 6.7 and 0.604 mg/l, respectively. This compares to mean concentrations for nitrate and nitrite of 1.99 mg/l and 0.2 mg/l for the remainder of the sites. The high COD values measured in the prawn farm effluent further indicated that a high nutrient load is being discharged into the estuary. The values above are very high when compared with water quality guidelines for coastal marine waters by DWAF (1995). DWAF (1995) indicated a natural level of >0.0014 and 0.038 mg/l for nitrite and nitrate, respectively.

The above trend is a complete opposite to that observed during the 1989-1991 sampling period, when highest nitrogen concentrations were recorded at Site 5 in the lower reaches. The nitrate concentrations recorded at the prawn farm outlet and at Site 3 during the 1999-2000 sampling period were less than nitrate concentrations recorded at Site 3 during the 1989-1991 sampling period. The nitrate concentration of 9.81 mg/l recorded during 1989-1991 was attributed to agricultural run-off since no industry or prawn farm was present during this period whilst the estuary was surrounded by sugarcane plantations. Since this high nitrate concentration was recorded once in autumn, it is possible that samples were taken after strong rains. Tovar, *et. al* (2000) reported a high concentration of nitrate recorded during winter in the San Pedro River impacted by the fishfarm. The samples were taken after a strong period of rain, which caused an increment in nitrate concentration due to runoff from agricultural land adjacent to the river. In the Mhlathuze estuary, Wepener and Vermeulen (1999) reported that a potential source of nitrates and nitrites was related to fixation of nitrogen by blue-green algae in the muds of mangrove areas, while sewage runoff from the surrounding rural settlements was another potential source.

Ammonia occurs not only in many effluents, but is produced in the natural degradation of nitrogenous matter and provides an essential link in the nitrogen cycle. Ammonia has two forms, ammonium (NH_4), which is an ionised form of nitrogen and the unionised dissolved ammonia gas (NH_3), which is more toxic than ammonium (Dallas, *et. al* 1994). The relative proportions of unionised ammonia (NH_3, toxic) and ionised ammonia (NH_4, non-toxic) depend mainly on the pH of the water. Dallas, *et. al* (1994) found that the higher the pH, the higher the proportion of unionised ammonia (NH_3). Since this form of ammonia is toxic, deleterious effects can occur. Increased temperature also causes an increase in relative proportion of unionised ammonia in aquatic systems, hence an increase in its toxicity to aquatic organisms (DWAF 1995). The autotrophic conversion of ammonia to nitrites and nitrates requires oxygen and so the discharge of ammonia and subsequent oxidation can reduce dissolved oxygen levels in estuaries with detrimental effects on biota (Sawyer & MacCarty 1989). The measurement of ammonium ion concentration is therefore useful in detecting point as well as non-point sources of pollutants, surveying nutrient levels in natural water bodies and, of particular importance in this case, for monitoring aquaculture projects for excessive waste concentrations. In the

Mlalazi estuary, ammonium concentrations were low (0.06 mg/l) during the 1989-1991 period whereas during the 1999-2000 period, concentrations were much higher, particularly at the prawn farm outlet. These high ammonia concentrations in the prawn farm effluent (0.7 mg/l) are further indications that the prawn farm is responsible for discharging a high nitrogen load into the estuary. The high ammonia concentrations recorded in the Mdloti and Tongati estuaries reflected severely eutrophic condition of these estuaries (Blaber, *et. al* 1984).

Phosphates and nitrogen are the main indicators of the nutrient status of a water body. Unusually high concentrations of nitrate and phosphate have been linked to agricultural drains discharging into the estuary (Scharler, *et. al* 1997). In the Choptank River estuary, a shallow estuary (>3 m) with 29 % of the catchment area forested and 66 % agricultural, agricultural practises were identified as the main source of the high nitrate content in the estuary (Staver, *et. al* 1996). Baker and Horton (1990) cited agricultural runoff as the main culprits introducing large quantities of nitrogen and phosphate into Chesapeake Bay, causing excessive growth of floating plants which in turn led to low oxygen in the bottom waters when plants eventually decompose. Similarly Birch (1982) reported that the excessive growth of benthic algae in the shallow Peel-Harvey estuary, western Australia, was positively correlated with the rate of fertiliser application in neighbouring fields. This seemed to be the case during the 1989-1991 period, whereas the prawn farm proved to be the primary source of phosphates and nitrogen during 1999-2000 sampling period.

Aquaculture practises in coastal areas are gradually changing the nature of environmental patterns since these areas are generally used as dumping grounds for effluents, generating a number of adverse situations. This is because effluents from aquaculture operations are typically enriched in suspended solids, nutrients and BOD (Paez-Osuna 2001). Tovar, *et. al* (2000) reported that up to 85 % of phosphorus and 52-95 % of nitrogen input into a marine fish culture system as feed may be lost into the environment through feed wastage. The major environmental concerns about aquacultures include eutrophication and sedimentation, excessive use of resources such as water and negative effects on fisheries and biodiversity (FAO 1995). Eutrophication of estuarine water is well documented to impact on water quality and eventually the biotic community (Emmel 1973. Scharler, *et. al* 1997, Ginkel, *et. al* 2000). The Dutch canal, which is the major water source for fish farming industry in Netherlands, has faced a problem of deterioration of its water quality (Corea, *et. al* 1995, in FAO 1999). Over the past decade many water quality parameters have changed towards unfavourable levels. Increase in sediment sulphides, nutrients and suspended solids were reported in the Dutch canal leading to detrimental effects of biotic community.

The pigment chlorophyll allows plants and algae to convert water and carbon dioxide to organic compounds in the presence of light (photosynthesis). To enhance their ability to capture light at different wavelengths, algae have evolved several kinds of chlorophyll with chlorophyll-a being the predominant kind (Brown 1992). The rate of primary production or photosynthesis in an aquatic system is usually closely related to the quantity of chlorophyll-a contained in phytoplankton in water column (Dallas, *et. al* 1994). Algae are abundant in aquatic environments as primary producers and are important components of water bodies, providing food for fish and other aquatic organisms (Koning, *et. al* 2000). Excess algae can, however, become a nuisance and interfere with the uses of a water body. The high chlorophyll-a concentrations recorded at Site 3 during the 1999-2000 period indicated a very high primary production rate in this area, which can probably be ascribed to higher nutrient concentrations in the prawn farm effluent. The results obtained by Trott and Alongi (2000) in the Muddy Creek estuary, Australia, indicated elevation of phytoplankton biomass and alteration of water quality characteristics by shrimp pond effluents discharged into the upper reaches of the estuary.

The concentration of oxygen in water is very important to the health of aquatic organisms. When oxygen is lost from the water column, water quality degradation usually follows. Waste material that contains oxygen-demanding materials can affect water in which these organisms live in. Chemical oxygen demand is thus a widely used measure of the pollutional strength of domestic and industrial effluents, which is related to the total quantity of oxygen required for oxidation of CO_2 and water (Sawyer & McCarty 1989). COD is helpful in indicating presence of biologically resistance substances. Although there was a gradual downstream increase in COD concentration in the Mlalazi estuary, the prawn farm effluent showed the highest concentrations compared to other sites. This is indicative of the high nutrient load in the prawn farm effluent, which is being discharged into the estuary.

Sulphur in water occurs largely in the sulphate (SO_4^{2-}) ion form and is derived predominantly from the sea (Day, *et. al* 1989). Sulphate rarely limits the growth or distribution of aquatic biota. Sulphates are of considerable concern because they are directly responsible for problems associated with the handling and treatment of wastewater. These include odour and sewer corrosion problems resulting from the reduction of sulphates to hydrogen sulphide (Sawyer & McCarty 1989). At high concentrations and under specific chemical conditions, dissolved sulphate results in the formation of sulphuric acid, a strong acid which can be detrimental to aquatic biota (Ketchum 1983). Under anaerobic conditions i.e. in the absence of oxygen, sulphate ion is reduced to sulphide ion, which establishes a equilibrium with hydrogen ion to form hydrogen sulphide. At pH levels above 8, most of the reduced sulphur exists as HS^- and S^2 ions and the amount of free H_2S is so small that odour problems do not occur (Day, *et. al* 1989). At pH levels below 8, the equilibrium shifts towards formation of un-ionised H_2S, which may cause odour problems whenever sulphate reduction yields a significant amount of sulphide ion. However, sulphate and pH levels in the Mlalazi estuary probably resulted in too little free H_2S therefore no odour problems occurred in the estuary. Although sulphate concentrations in the Mlalazi estuary showed a gradual increase from the head to the mouth, suggesting the sea to be a source of sulphate in the estuary, highest sulphate concentrations were recorded in the prawn farm effluent, with concentrations ranging between 6.9 to 297 mg/l. Toxic sulphide rich waters are often associated with anoxic conditions that are deleterious to benthic organisms (Sawyer & McCarty 1989). The Mlalazi estuary did not experience anoxic conditions as such, even though oxygen concentrations were slightly below 4 mg/l, especially in the middle reaches during both sampling periods. However the middle reaches were characterised by lower sulphate concentrations and as such a decrease in oxygen did not correspond to sulphate concentrations.

3.4.3. **Sediment**

Sediments constitute a major controlling factor in the ecology of estuaries. Sediment not only forms a habitat for many benthic plants and organisms, but also influences other life processes of organisms (Kennish 1986). For example, the high turbidity commonly encountered in estuaries is due in part to fine sediments held in suspension, which decrease light penetration thereby limit primary production. The morphology, feeding and dominance patterns of benthic species are a function of the substratum type (Kennish 1986). In addition, ecological relationships among benthos hinge on the characteristics of estuarine sediments.

The sediment within the estuary is usually very different from the adjacent sea and river (McLusky 1974). As tidal currents enter the estuary and slacken in speed, it will first deposit gravel then sand and finally fine particles. Thus the head of the estuary is dominated by coarse sediment, the middle reaches are often dominated by mud flats while sandbanks are commonly found at the mouth.

Sediments can be described by examining fundamental properties e.g. grain size and mineral composition (Kennish 1986). The sizes of sediment grains are frequently expressed on a geometric or log scale in which a continuous range of sediment sizes are subdivided into classes or grades (Kennish 1986). One of the major components of estuarine sediments is detrital particles and organic matter (Day 1981). Hill (1966) found the bottom sediments of the Mlalazi estuary to be either eroded to coarse sand with little silt or covered with mud containing 40 % silt. The muddy upper reaches extended to within 5 km from the sea and thereafter the middle and lower reaches constituted medium sand. The mouth contained clean medium to coarse sand. During the present study, the sediment pattern was different from that described by Hill (1966). During the 1989-1991 sampling period, the sediment ranged from medium sand in the upper reaches to silt in the middle reaches. The lower reaches were characterised by medium sand with the mouth region constituting fine sands. A comparison with the 1999-2000 sampling period indicated a decrease in the amount of silt in the middle reaches. The upper and lower reaches were characterised by coarse sand while the middle reaches were characterised by medium to fine sands. The mouth region constituted fine sediments.

Coarse sediments have been found to retain little organic matter (Gray 1981). This was also observed in the Mlalazi estuary during the 1999-2000 sampling period. Sites containing coarse sediment (Sites 1, 2, 5 and 6) had low to medium organic content (<1 - 2%). The middle reaches (Site 4) and the mouth region (Site 7), which contained fine to very fine sand were characterised by a high organic content (>2%). Fine sediments such as mud and silt have poor water circulation and often low oxygen tension (Gray 1981). This corresponds with the Mlalazi estuary sediment as sites with low dissolved oxygen content contained fine or silty sediment. In the adjacent Siyaya estuary sites with fine sediments also had low oxygen levels (Mackay 1996).

The sorting co-efficient of sediment represents the degree of the mixing of the different types of sediments. Well-sorted sediments tending towards homogeneity are typical of high wave and current activity (Gray 1981), whereas poorly sorted sediments are heterogeneous and are typical of low current activity. Sediments in the Mlalazi estuary were poorly to moderately sorted along the length of the estuary during both sampling periods. Sediments in the estuary were therefore not evenly distributed in terms of grain size except in the mouth region.

3.5. Summary and Conclusion

The water quality data discussed above made it possible to establish the physical conditions and water quality status of the Mlalazi estuary before the prawn farm was established and also eight years after the establishment of the prawn farm. The physical water quality data (e.g. salinity, temperature) indicated temporal and spatial variations that are typical for a subtropical estuary during both sampling periods. Chemical analysis suggest that agricultural run–off represented the main source of nutrients in the estuary during the 1989-1991 period. During the 1999-2000 period, effluents discharged from the prawn farm into the estuary contained concentrations of nutrients which exceeded that of other areas in the estuary, and which exceeded the acceptable standards for maintenance of the estuarine environment. This is considered to be a serious issue, which needs to be addressed, since high nutrient concentrations have a potential to create eutrophication problems and thus degradation of the estuarine environment. The results from this study serve as a valuable indicator of the current water quality status and associated problems in the Mlalazi estuary related to the prawn farm and should be used as a directive to initiate monitoring of the Mlalazi estuary.

CHAPTER 4

The Macrobenthic community of the Mlalazi estuary

4.1. Introduction

The study of macrobenthic organisms has received considerable attention due to the importance of benthic organisms as biological indicators of environmental change in aquatic ecosystems (Day 1981). Due to their sessile nature, macrobenthic organisms are regarded as representative of the habitat being sampled (Mackay & Cyrus 1999). Detection of distributional and abundance patterns of the macrobenthic community and the identification of factors or forces that govern these patterns, are often used as monitoring tools of spatial and temporal changes in response to perturbations (Flint, *et. al* 1986).

Although some estuaries in KwaZulu-Natal have received considerable attention, there are many that are poorly studied and for which little information is available. In Zululand, systems which are comparatively well studied include larger systems such as St. Lucia estuary (Day, *et. al* 1954, Boltt 1975, Blaber, *et. al* 1983, Owen & Forbes 1997) and the Mhlathuze estuary (Millard & Harrison 1954, Mackay & Cyrus 1999, Wepener & Vermeulen 1999), as well as smaller ones such as the Nhlabane (Vivier, *et. al* 1998) and Siyayi estuaries (Mackay 1996). However, apart from the study undertaken by Hill (1966), there is little information on the benthos of the Mlalazi estuary.

The fact that little is known about the biology of the Mlalazi estuary, the establishment of the prawn farm adjacent to the estuary and the concerns that have been expressed about the quality of water being discharged from the prawn farm into the estuary were the main reasons for this study being undertaken. This study was undertaken to determine the composition and abundance of the benthic fauna of the Mlalazi estuary prior to prawn farm establishment (1989-1991) and after the prawn farm was established (1999-2000). Comparison of the 1989-1991 data and the 1999-2000 data aims at revealing what changes occurred in the estuary since the prawn farm was established and whether the prawn farm activities had any effect on the macrobenthic fauna.

4.2. Materials and Methods

4.2.1. Sampling periods

Monthly sampling in the Mlalazi estuary was conducted over two sampling periods:

a) Prior to prawn farm development

From February 1989 to December 1989, monthly benthic samples were collected from Sites 1-4 along the length of the estuary. From January 1990 to February 1991, two additional sites, i.e. Sites 5 and 6 were sampled every month (Figure 2.2).

b) Eight years after prawn farm activities commenced

From August 1999 to July 2000, monthly benthic samples were collected from seven sites along the length of the estuary. The upper six sites (Sites 1-6) corresponded to the six sites sampled during the 1989-1991 sampling period, with an additional site (Site 7) being sampled at the mouth of the estuary (Figure 2.2).

4.2.2. Benthic sampling

Benthic samples were collected using a Zabalocki-type Ekman grab, which samples an area of 0.0236 m^2 to a depth of approximately 45mm. Five grabs were taken at each site and the contents of each washed through a 0.5 mm mesh sieve to ensure extraction of at least 95 % of the macrobenthic animals (Cyrus & Vivier 1994). The samples were then preserved in 4 % formaldehyde to which the vital dye phloxin was added to facilitate sorting in the laboratory. Animals in the benthic samples were then sorted, counted and identified to species level where possible under a stereo microscope. Descriptions and keys compiled by Day (1967a, 1967b, 1969), Kensely (1978) and Griffiths (1976) were used in the identification of different taxa.

4.3. Data analysis

Benthic data from the two sampling periods (1989-1991 and 1999-2000) were put into a matrix with species as rows and samples as columns. Counts were converted to densities (No.m^{-2}) following which the average of five grabs per site were calculated to obtain the mean monthly density of each species at each site. The monthly densities were then averaged into seasons where three months were grouped to represent a season as follows: -

September + October + November = Spring

December + January + February = Summer

March + April + May = Autumn

June + July + August = Winter

The macrobenthic densities per site were calculated by averaging densities over all the seasons for each sampling period. Macrobenthic densities per season were calculated as average densities over all sites. The macrobenthic densities were then graphically presented, spatially and temporally, for each sampling period. Percent contributions per site and season for each sampling period were calculated from the macrobenthic densities and graphically presented using pie charts.

The statistical package PRIMER (Plymouth Routines in Multivariate Ecological Research) was used for analysis of temporal and spatial changes in the benthic community (Clarke & Warwick 1994). The data were analysed using univariate, graphical and multivariate analysis as follows:

4.3.1. Univariate analyses

Diversity indices are mathematical expressions of three components of community structure, namely richness (number of species present), evenness (uniformity in the distribution of individuals among species) and abundance (total count of individuals present) (Dallas, *et. al* 1994). All three indices were used to describe species abundance relations and to reflect the response of a community to the quality of its environment.

i) **Shannon-Wiener diversity index**: expresses abundance and how evenly the individuals are distributed among different species:

$$H' = -\Sigma p_i (\log p_i)$$

ii) **Margalef's index**: a measure related to total number of species present:

$$d = (S-1)/ \log N$$

iii) **Pielou's evenness index**: expresses how evenly the individuals are distributed among different species:

$$J' = H' \text{ (observed)}/H'_{max}$$

4.3.2. Multivariate analyses

Classification and MDS

Densities were subjected to an appropriate transformation to conform to normality before being analysed. Hierarchical agglomerative clustering on a Bray-Curtis similarity matrix and Multi-Dimensional Scaling (MDS) were then used to delineate groups with distinct community structure. Two types of classification were used:

- by sample using all species. These multivariate routines were performed to allow analysis of spatial and temporal variations in biotic patterns between and within the two sampling periods.
- by species (inverse analysis). Inverse analysis was used to highlight those species that are responsible for determining patterns in the sample groupings. Indicator species of the two periods were identified using the program SIMPER which examine the contribution of individual species to the similarity measure used.

Non-metric Multidimensional scaling (MDS) was used where a stress of <0.2 gives a potentially useful 2-dimensional picture. At least 6 runs were undertaken to find the global minimum i.e. runs are done until two or more solutions with the same stress value are achieved.

Significance of any differences in benthic community structure

Non-parametric one-way analysis of similarity routine (ANOSIM) in the statistical package PRIMER was used to test for significant differences among sites and seasons of the Mlalazi benthic data. Two-way crossed ANOSIM was used to test for any significant differences ($p<0.05$) between the 1989-1991 and 1999-2000 sampling periods.

4.4. Results

4.4.1. Benthic community structure during 1989-1991 and 1999-2000

A total of 28 and 36 taxa were recorded from the Mlalazi estuary during the 1989-1991 and 1999-2000 sampling periods, respectively (Table 4.1 and 4.2). These included taxa such as polychaetes, amphipods, tanaids, isopods, brachyurans, gastropods and bivalves. Oligochaetes, hirudineans and cumaceans were also present, although in low numbers.

Polychaetes were well represented during both sampling periods with nine and fourteen taxa recorded during 1989-1991 and 1999-2000, respectively. During 1989-1991 there were three brachyuran and amphipod species while bivalves and gastropods were represented by five and two species each, respectively.

During 1999-2000, brachyurans were represented by three species. There were also four gastropod species, three bivalve species and two and three species of isopods and amphipods, respectively. Taxa that were recorded during the 1989-1991 period which were not found during the 1999-2000 period included the polychaete *Mercierella enigmatica*, the copepod *Pseudodiaptomus stuhlmanni* and the bivalves *Musculus (Brachidontes) virgiliae* and *Heterodonax ludwigii*. A number of polychaetes found during the 1999-2000 period were not recorded in the 1989-1991 period such as *Desdemona ornata* (Tables 4.1 and 4.2). Capitellids and *Prionospio spp*. dominated the polychaete component during both the sampling periods while the brachyurans were dominated by *Paratylodiplax blephariskios*.

Table 4.1. Macrobenthic densities (mean No/m^2 ± 1SE) of benthic taxa recorded from the Mlalazi estuary during 1989-1991 at Sites 1-6.

Species	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
POLYCHAETA						
Capitellidae	1564.08	2197.64	21807	99.33	3892.69	672.46
	(±373.79)	*(±383.83)*	*(±2579.7)*	*(±87.90)*	*(±1148.7)*	*(±379.30)*
Dendronereis arborifera	505.68	771.08	102.90	152.50	212.76	192.00
	(±110.27)	*(±154.74)*	*(±23.16)*	*(±24.05)*	*(±113.49)*	*(±57.80)*
Glycera convoluta	0.68	5.52	1.41	1.08	7.23	8.61
	(±0.06)	*(±2.66)*	*(±0.97)*	*(±0.78)*	*(±4.49)*	*(±4.23)*
Glycera longipinnis	19.48	16.68	13.95	52.25	22.38	23.07
	(±7.73)	*(±4.27)*	*(±5.16)*	*(±12.4)*	*(±8.25)*	*(±5.88)*
Glycera tridactyla	447.80	518.52	2.20	10.50	1.30	3.38
	(±85.27)	*(±350.17)*	*(±1.20)*	*(±6.48)*	*(±1.03)*	*(±2.10)*
Mercierella enigmatica	509.84	1326.44	405.25	1403.7	418.69	1981.46
	(±208.44)	*(±481.92)*	*(±191.260*	*(±383.50)*	*(±190.35)*	*(±639.20)*
Prionospio sp.	3334.48	3636.54	6461.90	2143.92	3468.07	3634.90
	(±797.87)	*(±961.75)*	*(±2155.1)*	*(±321.42)*	*(±1227.9)*	*(±784.94)*
Polychaete sp1	0.36	0.36	-	3.95	0.692	1.38
	(±0.03)	*(±0.03)*	-	*(±2.23)*	*(±0.069)*	*(±0.93)*
Polychaete sp2	-	-	-	-	-	2.61
	-	-	-	-	-	*(±1.62)*
HIRUDINEA	-	2.72	-	0.375	0.69	8.6
	-	*(±1.60)*	-	*(±0.175)*	*(±0.06)*	*(±5.2)*
CRUSTACEA						
TANAIDACEA						
Apseudes digitalis	3106.6	1440.40	3083	10.75	287.07	11.2
	(±769.25)	*(±468.17)*	*(±691.28)*	*(±7.05)*	*(±148.76)*	*(±4.56)*
AMPHIPODA						
Corophium triaenonyx	2585.76	2781.32	27.33	195.12	5.38	23.61
	(±1108.17)	*(±949.83)*	*(±14.96)*	*(±101.95)*	*(±3.31)*	*(±11.39)*
Grandidierella bonnieroides	508.68	506.04	234.5	56.32	74.46	296.15
	(±136.28)	*(±162.39)*	*(±68.85)*	*(±27.44)*	*(±35.71)*	*(±118.37)*
Amphipod sp1	8.96	17.44	0.375	1.08	3.30	2.69
	(±4.63)	*(±9.20)*	*(±0.175)*	*(±0.78)*	*(±2.64)*	*(±1.51)*
COPEPODA						
Pseudodiaptomus stuhlmanni	172.04	75.28	134.25	136.16	61.61	206.2
	(±58.46)	*(±23.06)*	*(±43.25)*	*(±53.66)*	*(±26.50)*	*(±104.54)*
ISOPODA						
Lepthanthura laevigata	98.04	239.12	72.08	2.87	62	17.69
	(±285.80)	*(±215.85)*	*(±16.99)*	*(±1.59)*	*(±59.22)*	*(±16.21)*
Isopod sp1	5.48	11	0.375	1.91	10.53	13.69
	(±5.11)	*(±8.92)*	*(±0.175)*	*(±1.22)*	*(±7.43)*	*(±8.65)*
BRACHYURA						
Brachyuran larvae (megalopa)	1.72	10.92	259.29	20.02	7.23	23.53
	(±1.39)	*(±5.8)*	*(± 182.29)*	*(±9.62)*	*(±4.90)*	*(±13.6)*
Hymenosoma orbiculare	7.68	3.12	0.375	1.45	1.35	2.07
	(±2.06)	*(±1.85)*	*(±0.175)*	*(±1.130)*	*(±0.9)*	*(±1.09)*
Paratylodiplax blephariskios	7.2	76	3269.54	0.375	1081.7	19.76
	(±2.83)	*(±75.25)*	*(±502.27)*	*(±0.175)*	*(±320.62)*	*(±13.44)*
BIVALVA						
Dosinia hepaticca	1.04	0.72	3.58	1.79	0.69	5.76
	(±0.04)	*(±0.49)*	*(±2.57)*	*(±1.02)*	*(±0.06)*	*(±2.45)*
Eumercia pauperculata	-	-	0.375	2.5	4	32.23
	-	-	*(±0.175)*	*(±1.4)*	*(±1.84)*	*(±9.8)*

Heterodonax ludwigii	-	0.36	-	-	22.9	23.76
	-	*(±0.06)*	-	-	*(±9.78)*	*(±7.78*
Macoma littoralis	10.44	12.24	20.70	10.5	90.46	256.76
	(±4.41)	*(± 4.42)*	*(±14.21)*	*(±4.76)*	*(±20.89)*	*(±100.29)*
Musculus (Brachidontes)	1851.04	265.68	755.91	6.79	0.69	2.6
virgiliae	*(±884.26)*	*(±140.93)*	(±752.43)	*(±3.77)*	*(±0.06)*	*(±1.77))*
GASTROPODA						
Assiminea capensis	-	-	-	6.41	-	1.30
	-	-	-	*(±3.46)*	-	*(±0.03)*
Nassarius sp.	1.4	0.36	2.87	2.20	-	4.07
	(±0.8)	*(±0.03)*	*(±2.05)*	*(±1.20)*	-	*(±1.59)*
CUMACEA	14.64	5.12	16.7	11.75	-	4.69
	(±13.20)	*(±2.30)*	*(±14.04)*	*(±9.19)*	-	*(±2.7)*

Spatial and temporal differences in number of taxa

Figure 4.1 shows the number of taxa recorded at each site and season in the Mlalazi estuary during the 1989-1991 and 1999-2000 sampling periods. During the 1989-1991 period, taxa numbers ranged from 23 at Site 3 to 26 at Site 4 and 6. During 1999-2000, the number of taxa was highest at Site 6 (28) with the lowest number of taxa recorded at Site 4 (20) in the middle reaches.

Seasonally, the highest number of taxa was recorded in spring and winter (26, each) during the 1989-1991 period. The lowest number of taxa was recorded in summer and autumn (24). The number of taxa declined from spring through to winter during the 1999-2000 sampling period. The highest number of taxa (30) was recorded in spring, declining to a minimum of 25 taxa in winter.

Table 4.2. Macrobenthic densities (mean No/m^2 ± 1SE) of benthic taxa recorded from the Mlalazi estuary during 1999-2000 at Sites 1-7.

Species	Site1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
POLYCHAETA							
Ancistrocyllis parva	-	-	-	0.7	0.7	10.5	157.3
	-	-	-	*(±0.17)*	*(±0.7)*	*(±9.0)*	*(±39.7)*
Capitellidae	1873.2	1442.7	1458.8	709.8	24.5	7	20.6
	(±526.4)	*(±337.7)*	*(±347.3)*	*(±132.7)*	*(±15.2)*	*(±4.4)*	*(±18.2)*
Ceratonereis keiskamma	740.6	304.5	139.3	-	7.7	0.7	-
	(±279.7)	*(±92.9)*	*(±57.4)*	-	*(±7.0)*	*(±0.7)*	-
Dendronereis. arborifera	662.2	702.1	1019.2	15.4	16.1	19.6	68.0
	(±132.7)	*(±124.8)*	*(±325.3)*	*(±5.6)*	*(±6.8)*	*(±7.5)*	*(±34.4)*
Desdemona ornata	413	304.5	120.4	8.4	421.4	102.9	3.8
	(±157.1)	*(±91.4)*	*(±80.5)*	*(±7.0)*	*(±221.2)*	*(±58.6)*	*(±2.6)*
Glycera spp	4.9	23.1	21.7	2.1	39.9	37.8	29.0
	(±3.9)	*(±6.5)*	*(±9.0)*	*(±1.5)*	*(±12.5)*	*(±11.7)*	*(±9.5)*
Glycinde capensis	-	0.7	0.7	-	-	0.7	4.6
	-	*(±0.17)*	*(±0.17)*	-	-	*(±0.7)*	*(±3.1)*
Heteromastus filiformis	-	0.7	2.8	2.1	3.5	8.4	80.9
	-	*(±0.7)*	*(±1.7)*	*(±1.5)*	*(±2.4)*	*(±3.7)*	*(±17.0)*
Magelona cincta	-	-	-	-	-	2.8	15.3
	-	-	-	-	-	*(±1.6)*	*(±11.3)*
Phyllodocea sp	0.7	-	-	0.7	-	1.4	1.5
	(±0.7)	-	-	*(±0.7)*	-	*(±0.9)*	*(±1.5)*
Prionospio spp	767.9	969.5	901.6	829.5	1917.3	1099	291.7
	(±334.3)	*(±413.4)*	*(±359.1)*	*(±148.5)*	*(±415.6)*	*(±186.3)*	*(±81.8)*
Tharyx spp	-	-	-	-	3.5	293.3	196.3
	-	-	-	-	*(±2.2)*	*(±111.6)*	*(±47.9)*
Polychaete sp1	-	-	-	-	-	1.4	-
	-	-	-	-	-	*(±0.4)*	-
Polychaete sp2	-	-	-	-	0.7	-	1.5
	-	-	-	-	*(±0.7)*	-	*(±1.5)*
OLIGOCHAETA	2.8	-	-	-	0.7	-	0.8
	(±1.8)	-	-	-	*(±0.7)*	-	*(±0.6)*
HIRUDINEA	21	-	0.7	-	1.4	-	-
	(±1.5)	-	*(±0.7)*	-	*(±0.4)*	-	-
CRUSTACEA							
TANAIDACEAE							
Apseudes digitalis	368.9	63.7	120.4	422.8	4.9	7.7	1.5
	(±204.2)	*(±36.5)*	*(±58.2)*	*(±250.5)*	*(±3.3)*	*(±4.4)*	*(±0.5)*
AMPHIPODA							
Corophium triaenonyx	758.8	100.1	2.8	1.4	4.2	2.1	0.8
	(±561.2)	*(±66.3)*	*(±1.58)*	*(±0.9)*	*(±2.8)*	*(±1.5)*	*(±0.4)*
Grandidierella lignorum	133	21.7	1.4	28.7	4.9	12.6	16.8
	(±57.9)	*(±12.8)*	*(±0.9)*	*(±21.5)*	*(±2.8)*	*(±8.6)*	*(±15.2)*
Talorchestia australis	0.7	-	-	-	-	-	-

	(±0.7)	-	-	-	-	-	-
ISOPODA							
Excirolana natalensis	49	4.9	-	-	-	-	-
	(±22.7)	*(±2.4)*	-	-	-	-	-
Leptanthura laevigata	132.3	23.1	2.8	11.9	1.4	2.1	
	(±55.8)	*(±8.2)*	*(±1.2)*	*(±11.2)*	*(±0.9)*	*(±1.1)*	
BRACHYURA							
Brachyuran larvae	0.7	16.8	-	7	4.2	1.4	-
	(±0.7)	*(±12.7)*	-	*(±4.2)*	*(±2.8)*	*(±0.9)*	-
Hymenosoma orbiculare	0.7	2.8	-	-	-	0.7	-
	(±0.7)	*(±1.9)*	-	-	-	*(±0.7)*	-
Paratylodiplax blephariskios	19.6	4.9	134.4	1016.4	25.2	2.8	10.7
	(±14.6)	*(±2.8)*	*(±97.6)*	*(±88.0)*	*(±24.4)*	*(±1.8)*	*(±7.0)*
Scylla serrata	-	-	-	-	0.7	-	-
	-	-	-	-	*(±0.7)*	-	-
BIVALVA							-
Dosinia hepatica	4.9	2.8	2.1	4.2	6.3	56.7	138.2
	(±2.4)	*(±1.6)*	*(±1.1)*	*(±3.0)*	*(±2.9)*	*(±22.4)*	*(±19.8)*
Eumercia pauperculata	0.7	0.7	0.7	-	3.5	3.5	2.3
	(±0.07)	*(±0.7)*	*(±0.7)*	-	*(±1.2)*	*(±1.2)*	*(±1.2)*
Macoma sp	1.4		0.7	3.5	2.1	15.4	28.3
	(±0.94)		*(±0.7)*	*(±2.2)*	*(±1.2)*	*(±9.0)*	*(±7.2)*
GASTROPODA							
Assiminea ovata	-	0.7	18.2	11.2	46.9	24.5	19.9
	-	*(±0.7)*	*(±7)*	*(±4.4)*	*(±20.6)*	*(±11.9)*	*(±9.4)*
Melanoides turbeculata	1.4	-	2.1	-	2.1	0.7	4.6
	(±0.94)	-	*(±1.5)*	-	*(±1.5)*	*(±0.7)*	*(±2.6)*
Nassarius sp.	1.4	1.4	2.8	29.4	9.8	38.5	65.7
	(±0.94)	*(±0.9)*	*(±1.2)*	*(±7.3)*	*(±2.9)*	*(±10.1)*	*(±14.0)*
Solen capensis	4.9	1.4	2.8	-	1.4	2.1	-
	(±4.2)	*(±0.15)*	*(±1.6)*	-	*(±0.9)*	*(±1.5)*	-
STOMATOPODA	4.9	-	-	-	-	-	-
	(±4.1)	-	-	-	-	-	-
CUMACEA	-	14.7	4.9	7.7	6.3	6.3	-
	-	*(±7.5)*	*(±3.6)*	*(±5.6)*	*(±4.3)*	*(±3.6)*	-
SIPUNCULID	-	-	0.7	-	-	-	-
	-	-	*(±0.7)*	-	-	-	-

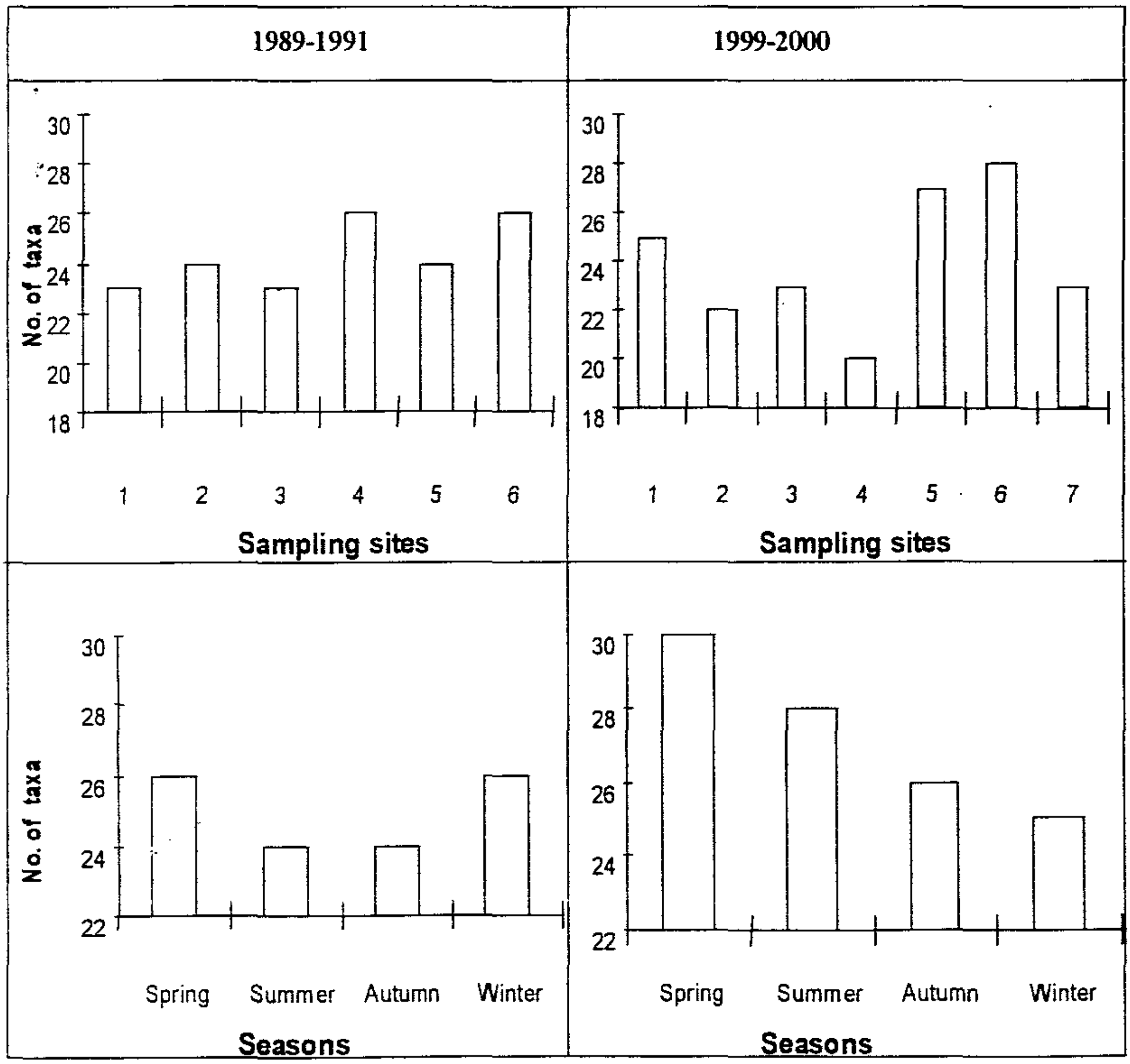

Figure 4.1. Number of taxa recorded at each site and season from the Mlalazi estuary during the 1989-1991 and 1999-2000 sampling periods.

Spatial and temporal differences in macrobenthic densities

Figure 4.2 presents the mean macrobenthic densities per site and per season from the Mlalazi estuary during the 1989-1991 and the 1999-2000 sampling periods. There were large differences in densities between the two periods with higher densities obtained during the 1989-1991 period.

The highest densities during the 1989-1991 period were recorded at Site 3 (143 523 No/m^2). The densities at this site were almost triple that recorded at any other site during all four seasons. Site 4 contained the lowest macrobenthic densities throughout the study period with a mean density of 17 344 No/m^2. Seasonally, densities ranged from 138 721 No/m^2 in spring to 53 273 No/m^2 in autumn. Highest seasonal macrobenthic densities were recorded at Site 3 (45 804 No/m^2) in spring with lowest densities being recorded at Site 4 (2 155 No/m^2) in summer.

During the 1999-2000 sampling period species density decreased from Site 1 to Site 7. Throughout the 1999-2000 period, the highest benthic densities were recorded at Site 1 (23 789 No/m^2). Macrobenthic organisms at this site were most abundant in summer (7 650 No/m^2) and winter (7 132 No/m^2), decreasing in autumn (4 337 No/m^2). The densities during summer and winter at Site 1 were almost double that recorded at any other sites. Site 7 was the most impoverished site throughout the year, with a mean density of only 4 609 No/m^2. Relative high densities were recorded at Sites 2, 3 and 5 during winter (Site 2 =6 899, Site 3 = 6 958, Site 5 = 4 558 No/m^2) with much lower densities found at these sites during the rest of the year. Highest densities at Site 4 were attained in spring (3 878 No/m^2) with autumn showing lower densities. Overall, the highest macrobenthic densities were recorded during winter (31 157 Nom^2), with the lowest densities being recorded in autumn (15 884 No/m^2, Figure 4.2).

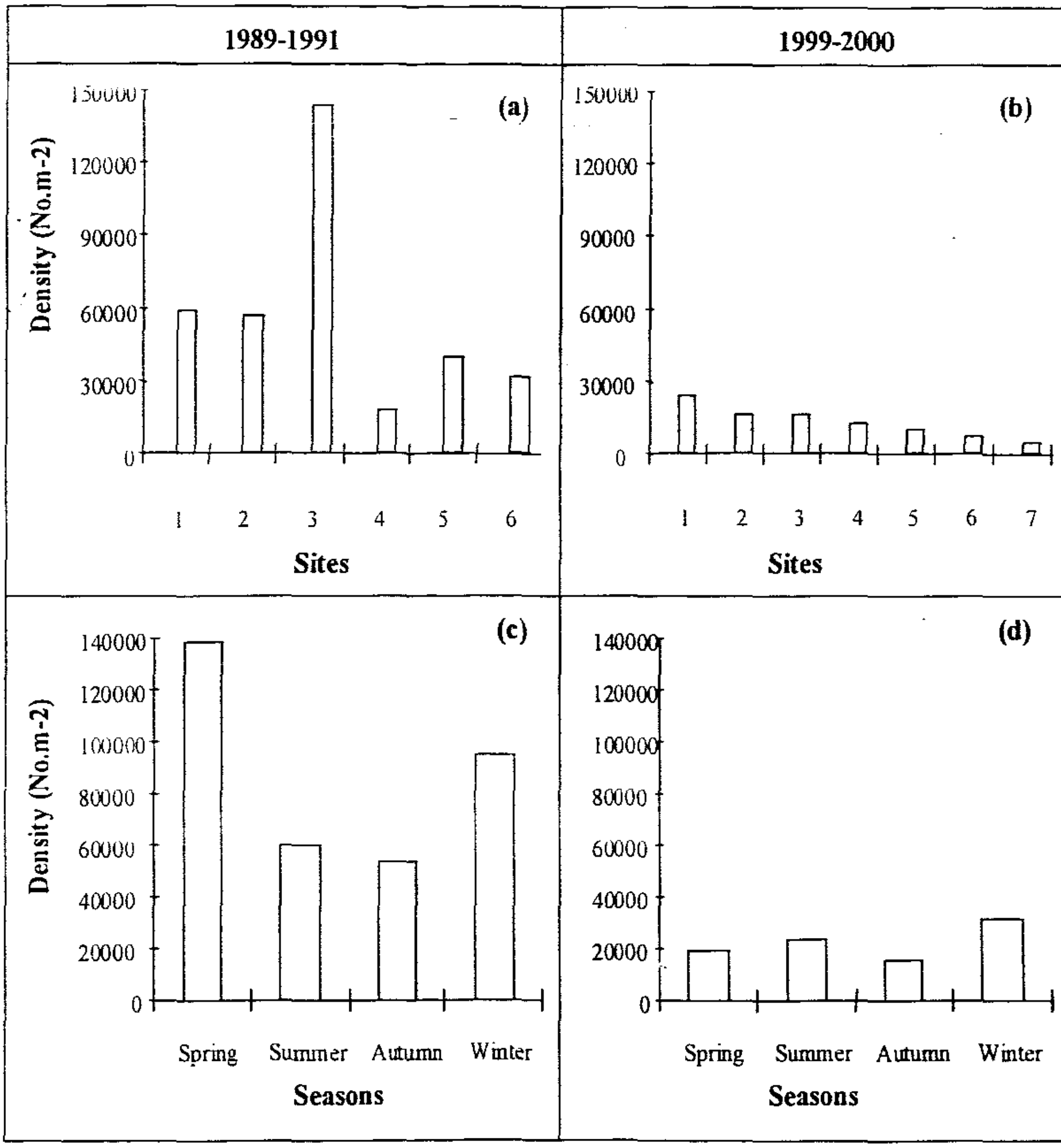

Figure 4.2. Mean macrobenthic densities (a+b) averaged over the sampling period for each site, and averaged over all sites for each season (c+d) recorded from the Mlalazi estuary during the 1989-1991 and 1999-2000 sampling periods.

4.4.2. **Benthic composition at different sites during 1989-1991 and 1999-2000**

Figures 4.3 and 4.4 present the dominant species per site recorded during the 1989-1991 and 1999-2000 sampling periods from the Mlalazi estuary, respectively. Those species contributing less than 2 % of the abundance were grouped together as 'others'. Generally, both periods were dominated by capitellids and *Prionospio spp.* polychaetes at all the sites and during all seasons.

During the 1989-1991 period, *Prionospio spp.* and capitellids dominated all sites. Other numerically important taxa included the tanaid *A. digitalis*, the amphipod *C. triaenonyx*, the bivalve M. *virgiliae* (Site 1), the polychaetes *D. arborifera* (Site 2) and *M. enigmatica* (Site 4 and 6), the brachyuran *P. blephariskios* (Site 3).

During the 1999-2000 period, the polychaetes, *Prionospio spp.* and capitellids were the dominant taxa almost at all sites. The capitellids were most dominant at Sites 1,2 and 3, but were absent at Sites 5 and 6, while at Site 7 it only contributed 2 % of the benthic fauna. At Sites 5 and 6 the above polychaetes were replaced by *Tharyx sp.* (17 %), the bivalve *D. hepatica* and the gastropod *Assiminea ovata*. *Prionospio spp.* were most dominant at Sites 5 and 7 but were also present in some numbers at Sites 1-3. Site 4 was the exception with the brachyuran *P. blephariskios* dominating this site. The capitellid polychaetes and *Prionospio spp.* were also present at this site, but in low densities.

4.4.3. **Seasonal composition of the benthic community during 1989-1991 and 1999- 2000**

Figures 4.5 and 4.6 represent the seasonal percentage contributions of benthic taxa during the 1989-1991 and 1999-2000 sampling periods. The two polychaetes, *Prionospio spp.* and capitellids were the dominant taxa during both sampling periods. These two polychaetes were particularly abundant in autumn and winter seasons.

The polychaetes, capitellids and *Prionospio spp.* were generally the dominant taxa at all four seasons during the 1989-1991 sampling period (Figure 4.5). *Prionospio spp.* was dominant in spring and winter while capitellids were more abundant in summer, autumn

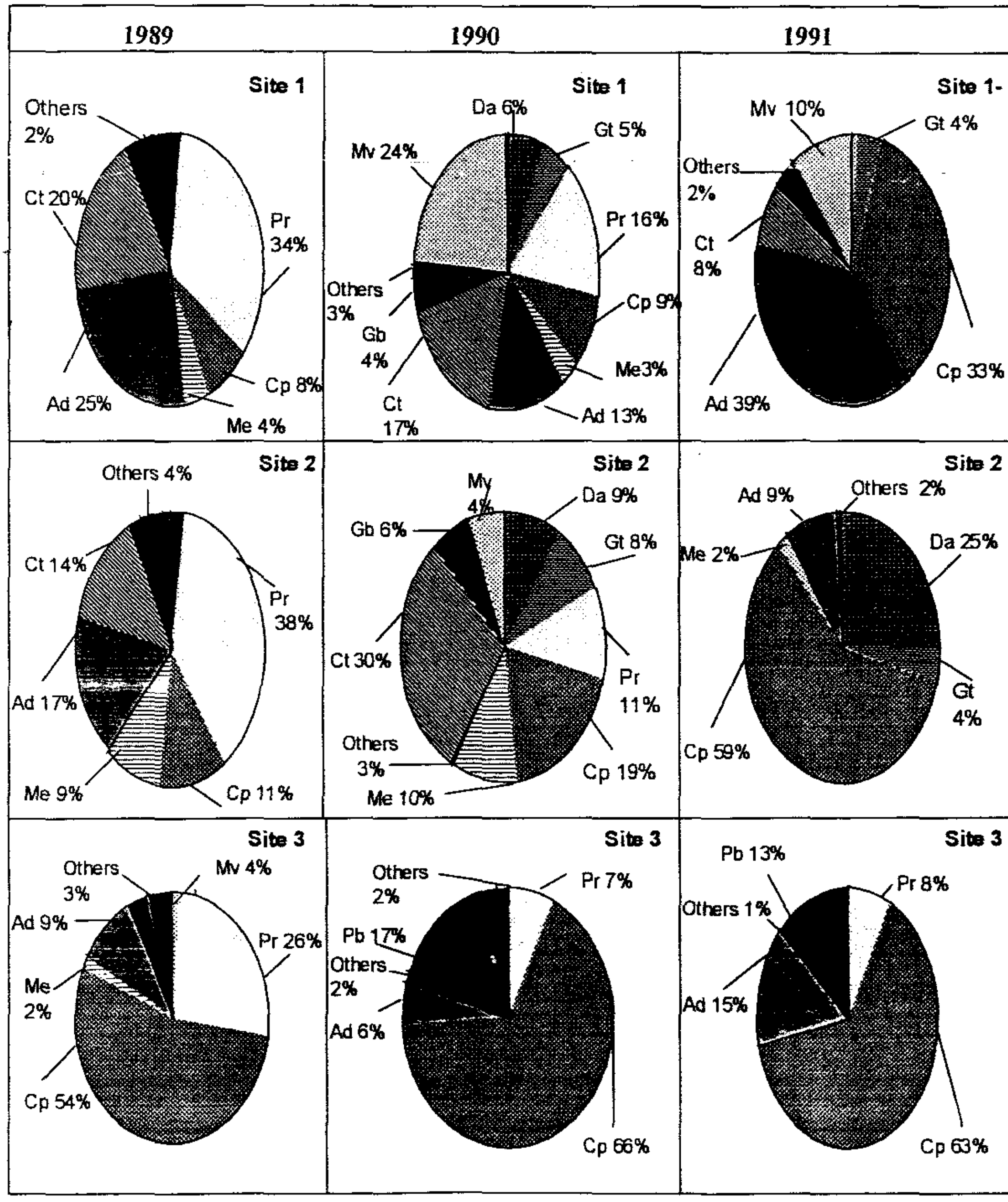

Figure 4.3. Species distribution and the percentage contribution per site (1-3) sampled from the Mlalazi estuary the during 1989-1991 period. (See Fig. 4.3 cont. for key to taxon abbreviation).

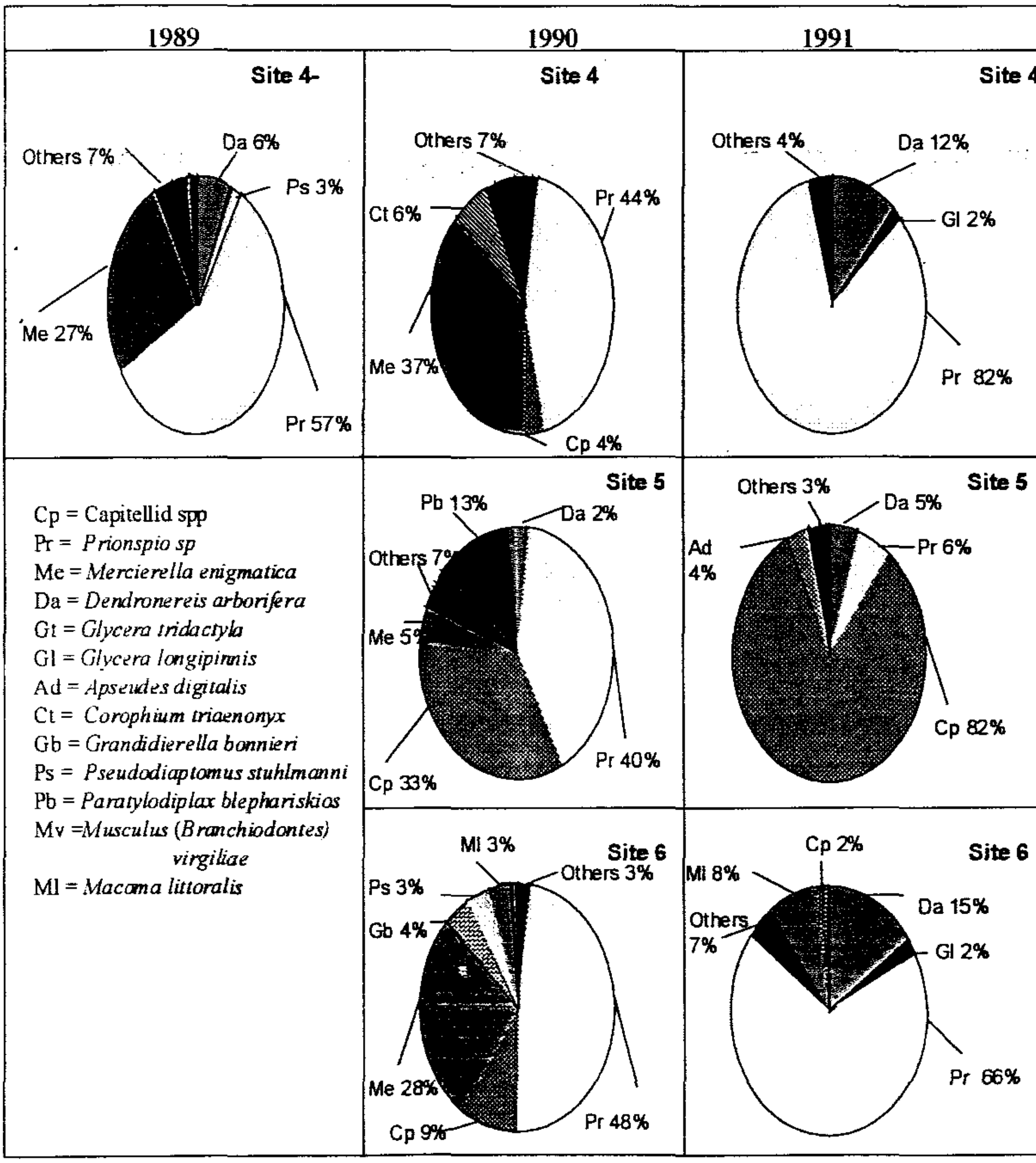

Figure 4.3 (cont.). Species distribution and the percentage contribution per site (4-6) sampled from the Mlalazi estuary during the 1989-1991 period.

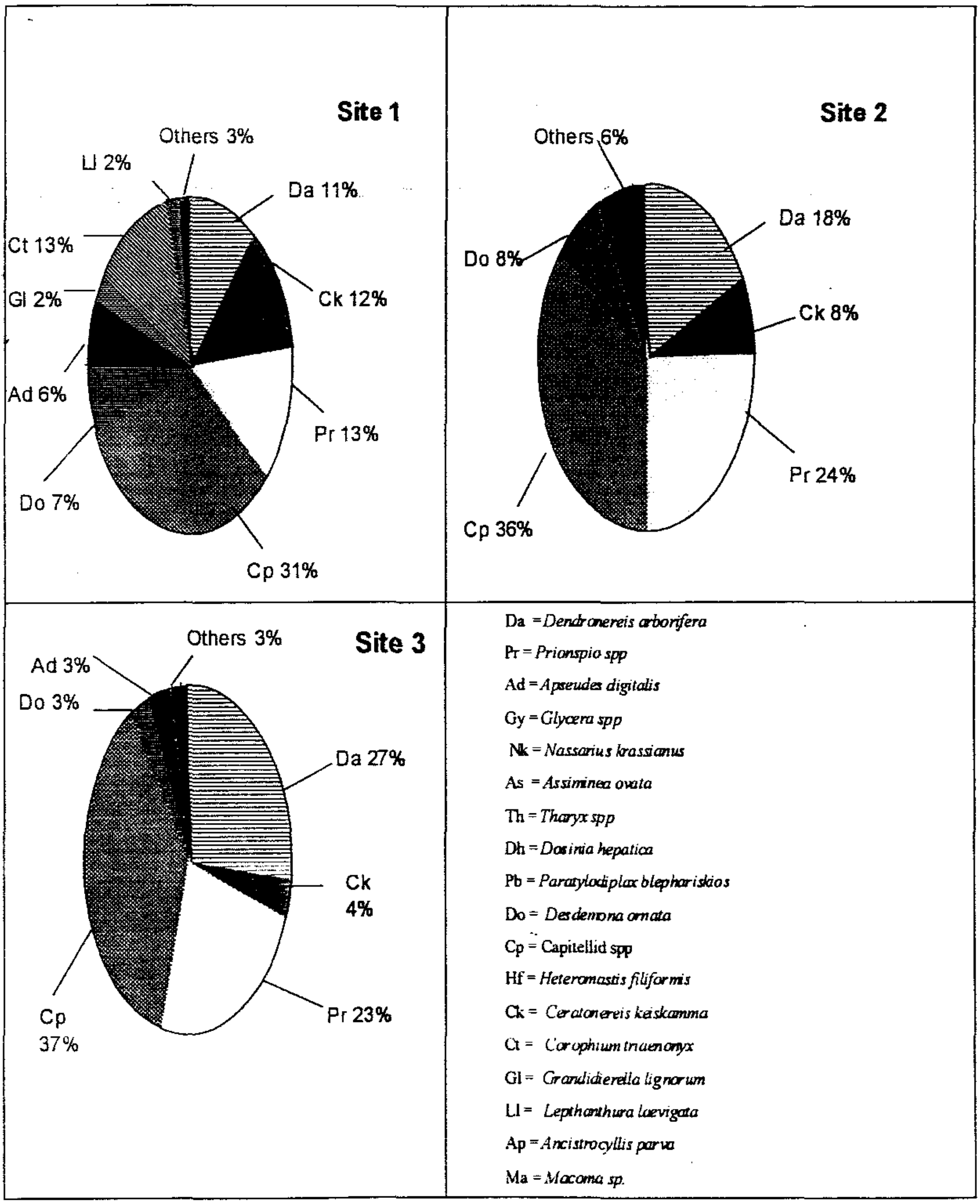

Figure. 4. 4. Species distribution and the percentage contribution per site (1-3) sampled from the Mlalazi estuary during the 1999-2000 period.

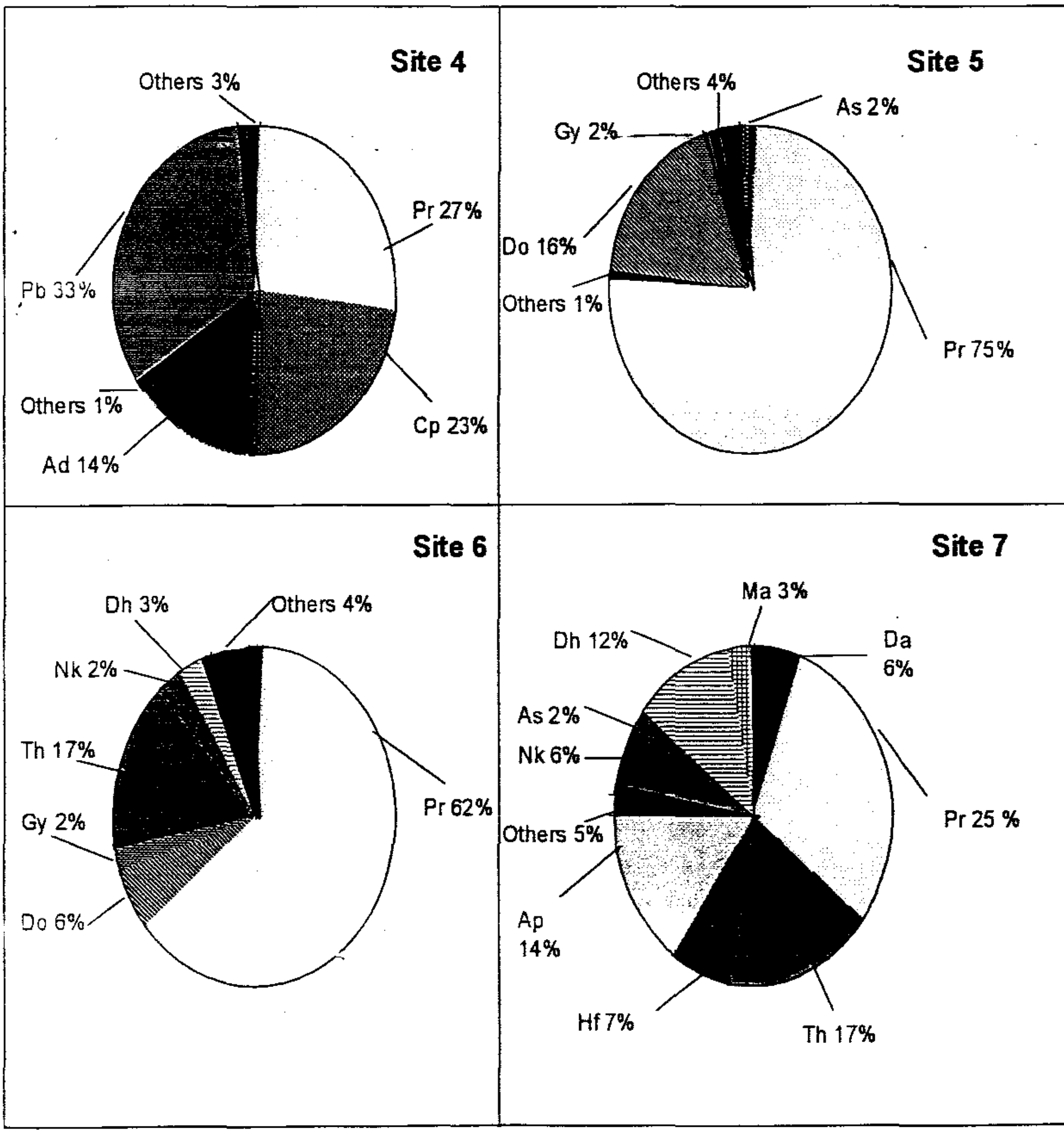

Figure. 4. 4. (cont.) Species distribution and the percentage contribution per site (4-7) sampled from the Mlalazi estuary during the 1999-2000 period. (See Figure 4.4 for key to taxon abbreviation).

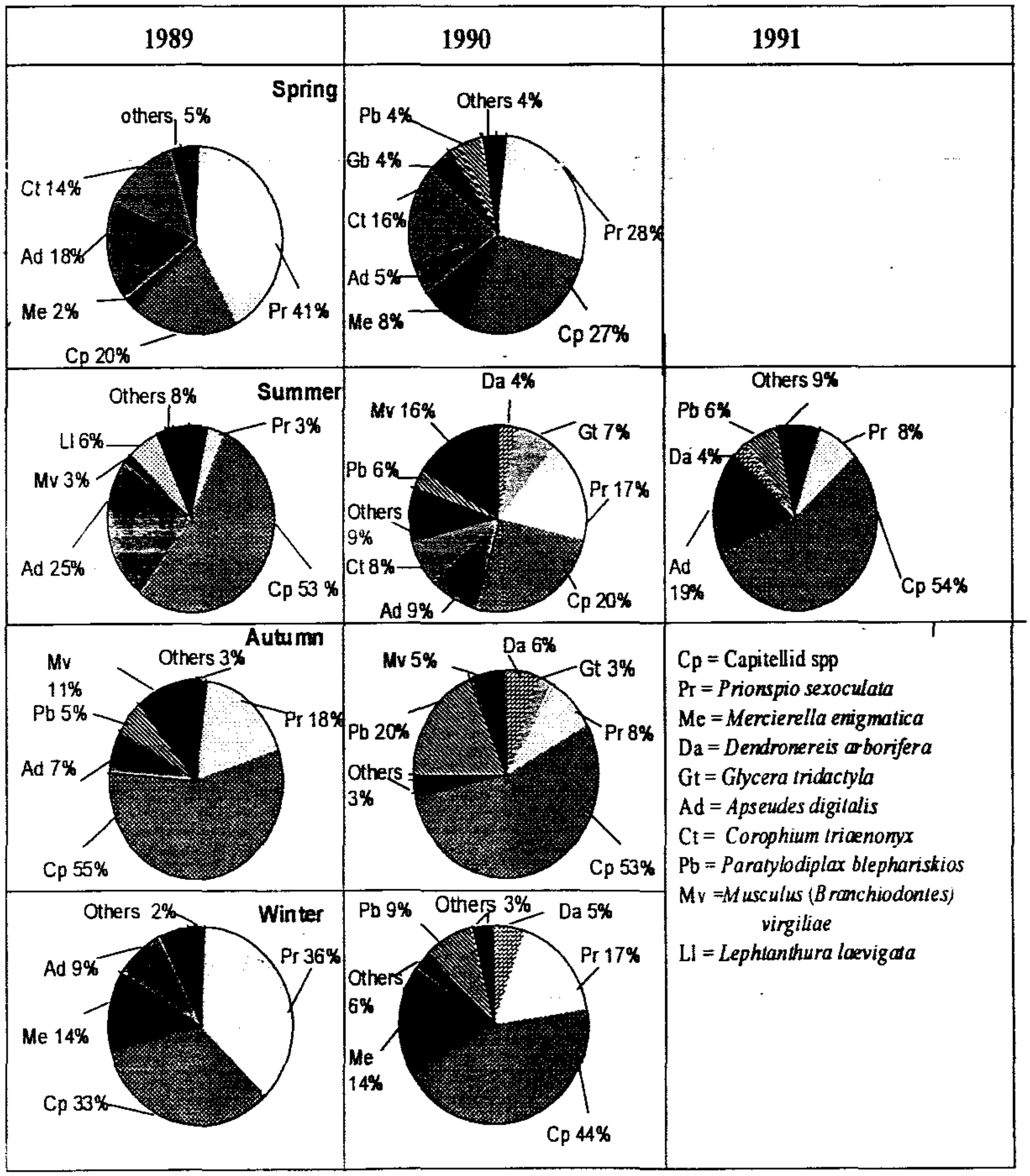

Figure 4.5. Seasonal percentage contribution of dominant species from the Mlalazi estuary during the 1989-1991 period.

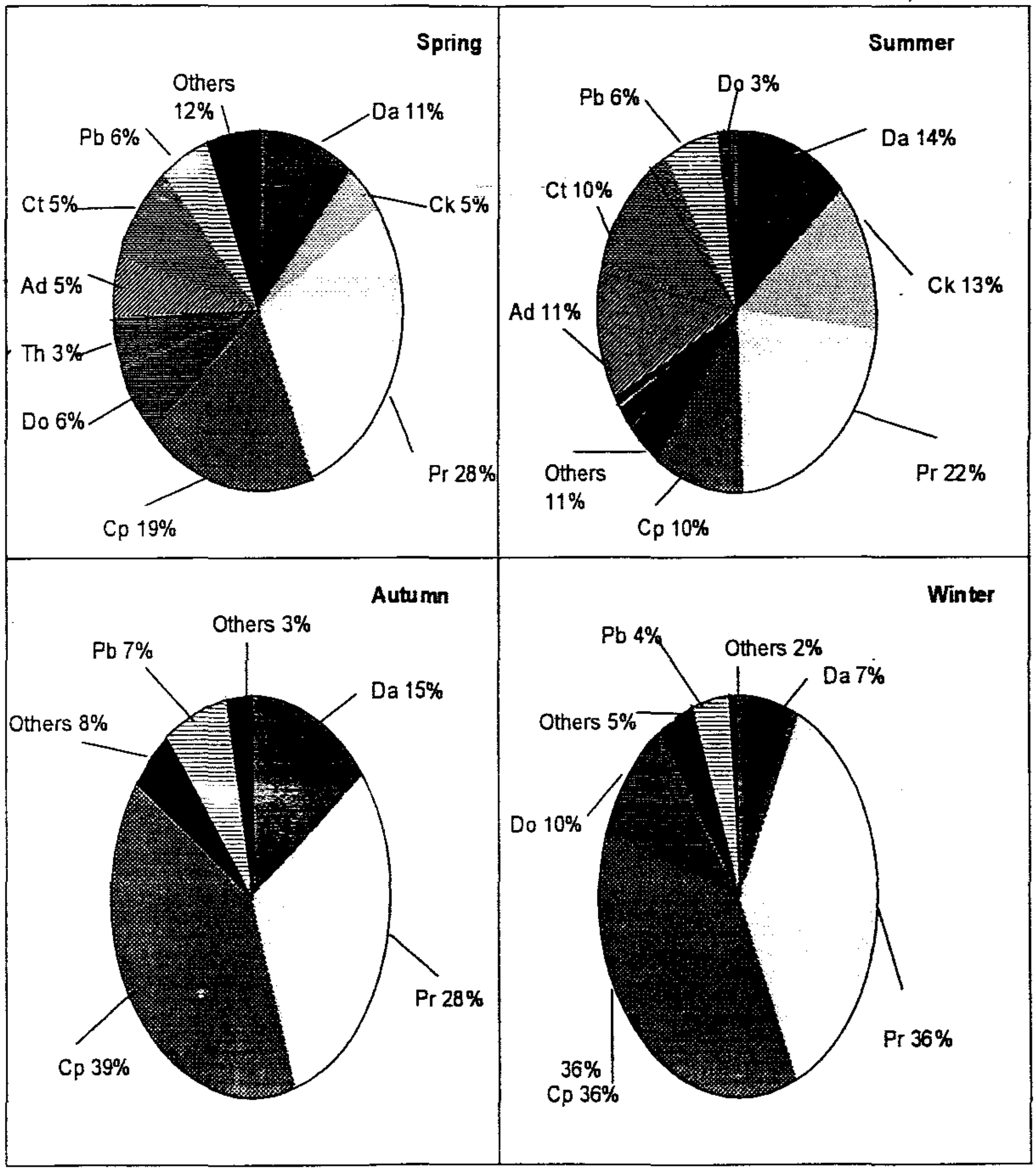

Figure 4.6. Seasonal percentage contribution of dominant species from the Mlalazi estuary during the 1999-2000 period. (See Figure 4.5 for key to taxon abbreviation).

and winter. The tanaid *A. digitalis* was numerically important in summer, *P. blephariskios* and M. *virgiliae* in autumn, and *M. enigmatica* in winter. During all four seasons in the 1999-2000 period, the polychaetes, *Prionospio spp. and* capitellids were the dominant taxa (Figure 4.6). These two taxa were particularly abundant during autumn and winter with capitellids forming 39% and 36% of the benthos during autumn and winter respectively. During spring and summer, *Prionospio spp.* contributed 28% and 22%, respectively, followed by capitellids which contributed 19% and 10%, respectively.

4.4.4. **Species Diversity**

Spatial variations

Figures 4.7 and 4.8 illustrate the diversity indices applied to the 1989-1991 and 1999-2000 benthic data.

Diversity

The diversity pattern that was found in the 1989-1991 period was different to that found during the 1999-2000 sampling period. Diversity was high in the upper reaches (Sites 1 and 2) decreasing markedly in the middle reaches (Sites 3 and 4) after which it gradually increased again towards the lower reaches (Sites 5 and 6) during the 1989-1991 period. During the 1999-2000 period, Sites 5 and 6 (lower reaches) showed the lowest diversity while Site 7, in the mouth region, showed the highest diversity compared to the other sites sampled.

Species richness

Sites 4 and 6 showed the highest species richness during the 1989-1991 period. Low species richness was recorded at Sites 3 and 5. During the 1999-2000 period, Site 7 showed the highest species richness, while Site 4 showed the lowest species richness compared with the other sites.

Species evenness

During the 1989-1991 period, Sites 1 and 2 showed a more evenly distributed benthic community compared with the other sites. Low evenness scores were recorded at Sites 3 and 4, which then increased slightly towards Sites 5 and 6. During 1999-2000, Site 7 reflected a more evenly distributed community compared with the other sites, with Sites 5 and 6 showing the lowest evenness scores. Evenness, during the 1999-2000 period, decreased from the mouth of the estuary to the lower reaches and then increasing again in the middle and upper reaches.

Temporal variations

Seasonal variations in species diversity during the 1989-1991 and 1999-2000 periods are represented in Figures 4.9 and 4.10. A trend was observed in that diversity was high during warm seasons and decreased in cooler seasons, with species richness following a similar trend to that of diversity.

During the 1989-1991 sampling period, the highest diversity scores were recorded during summer and spring with a low diversity recorded in autumn and winter. A similar pattern to diversity was apparent with species richness. The benthic community was evenly distributed in terms of species composition during summer, autumn and winter compared with spring. During 1999-2000, diversity was highest in spring followed by summer with lowest diversity being recorded in autumn (Figure 4.10). Species richness followed the same trend as diversity with high values for spring and low values for autumn. The trend seen in the evenness index corresponded with diversity and species richness.

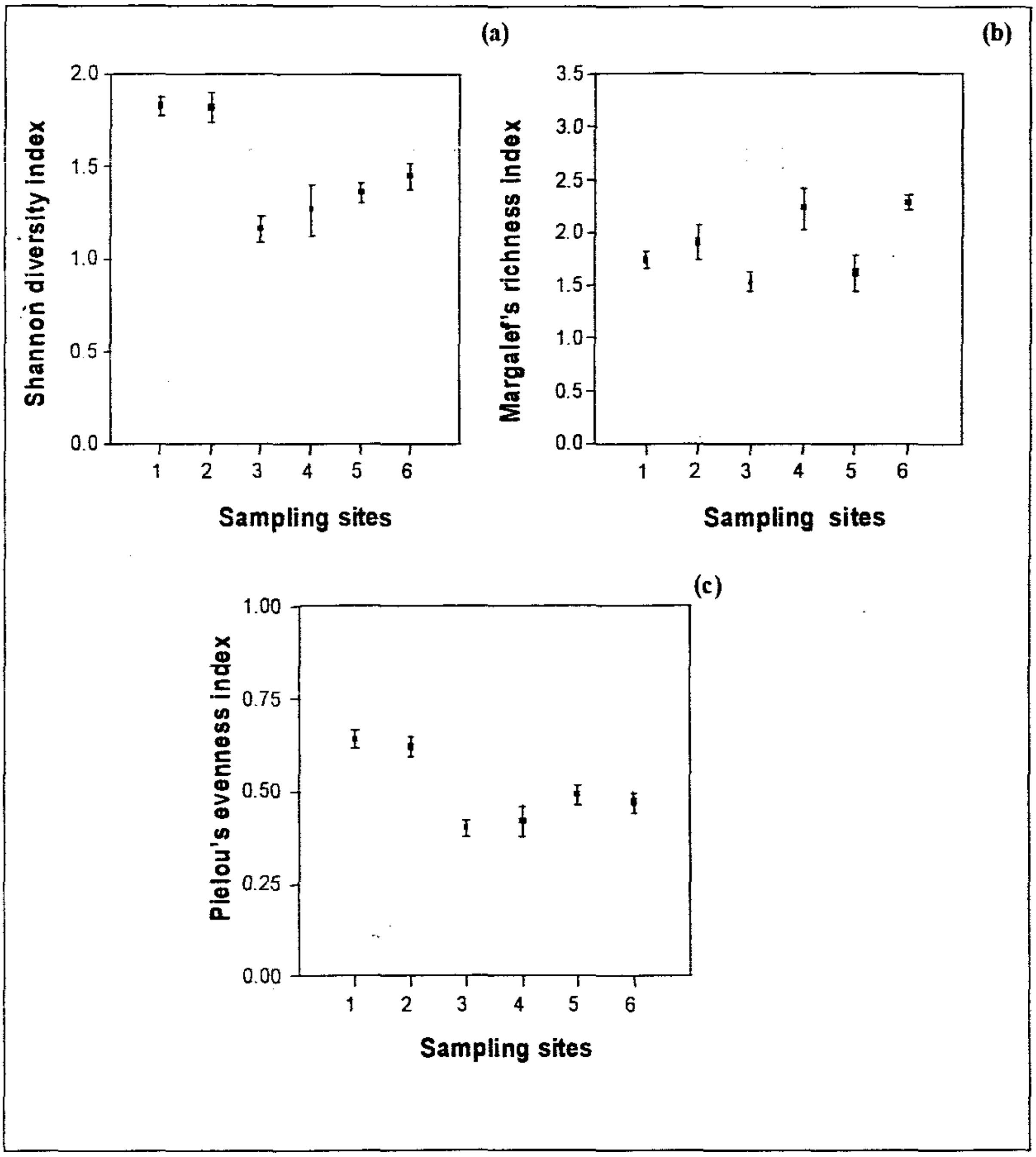

Figure 4.7. Mean (a) diversity, (b) richness and (c) evenness scores as recorded at Sites 1-6 from the Mlalazi estuary (1989-1991). (The bars indicate the mean ±1SE).

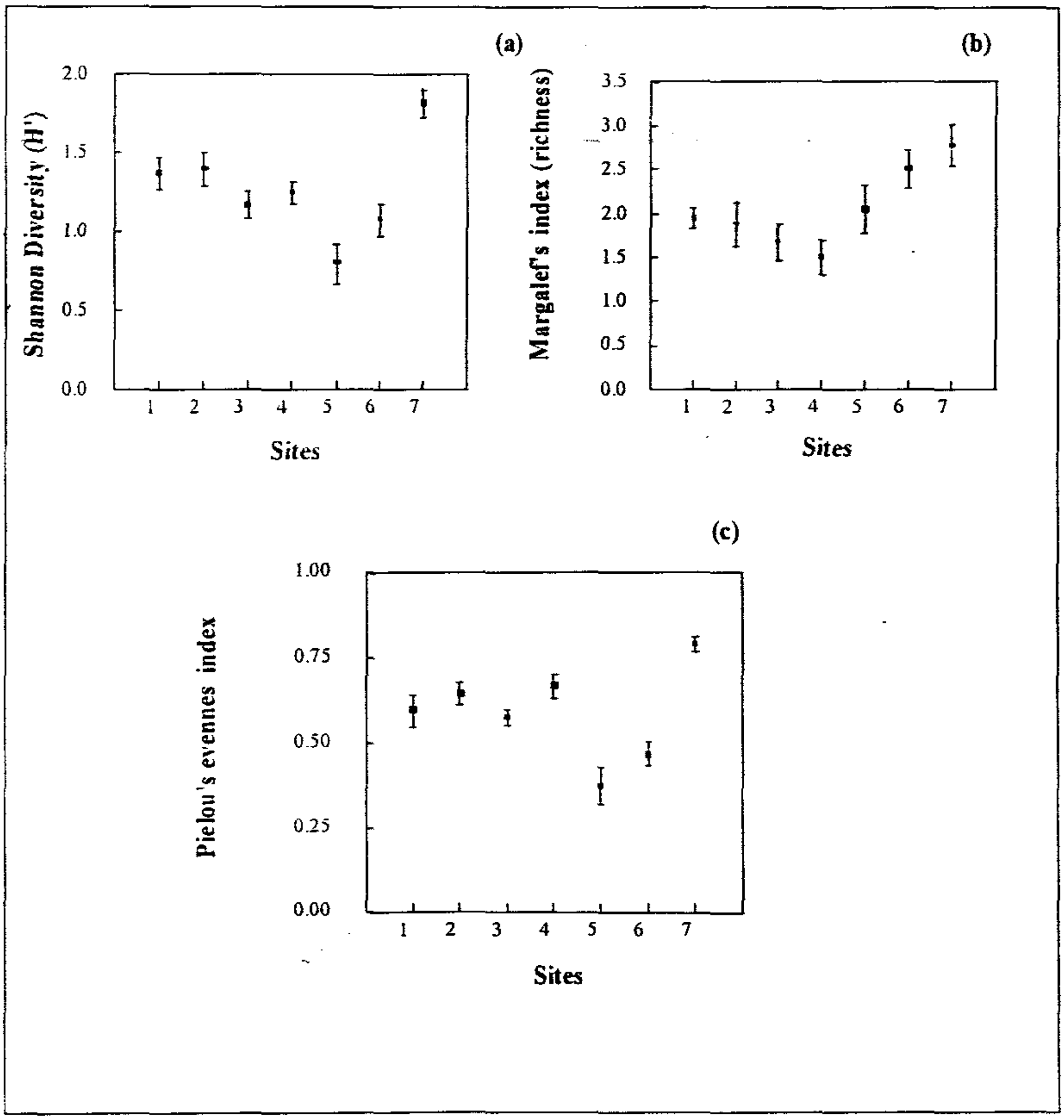

Figure. 4.8. Mean (a) diversity, (b) richness and (c) evenness scores as recorded at Sites 1-7 from the Mlalazi estuary (1999- 2000). (The bars indicate the mean ± 1SE).

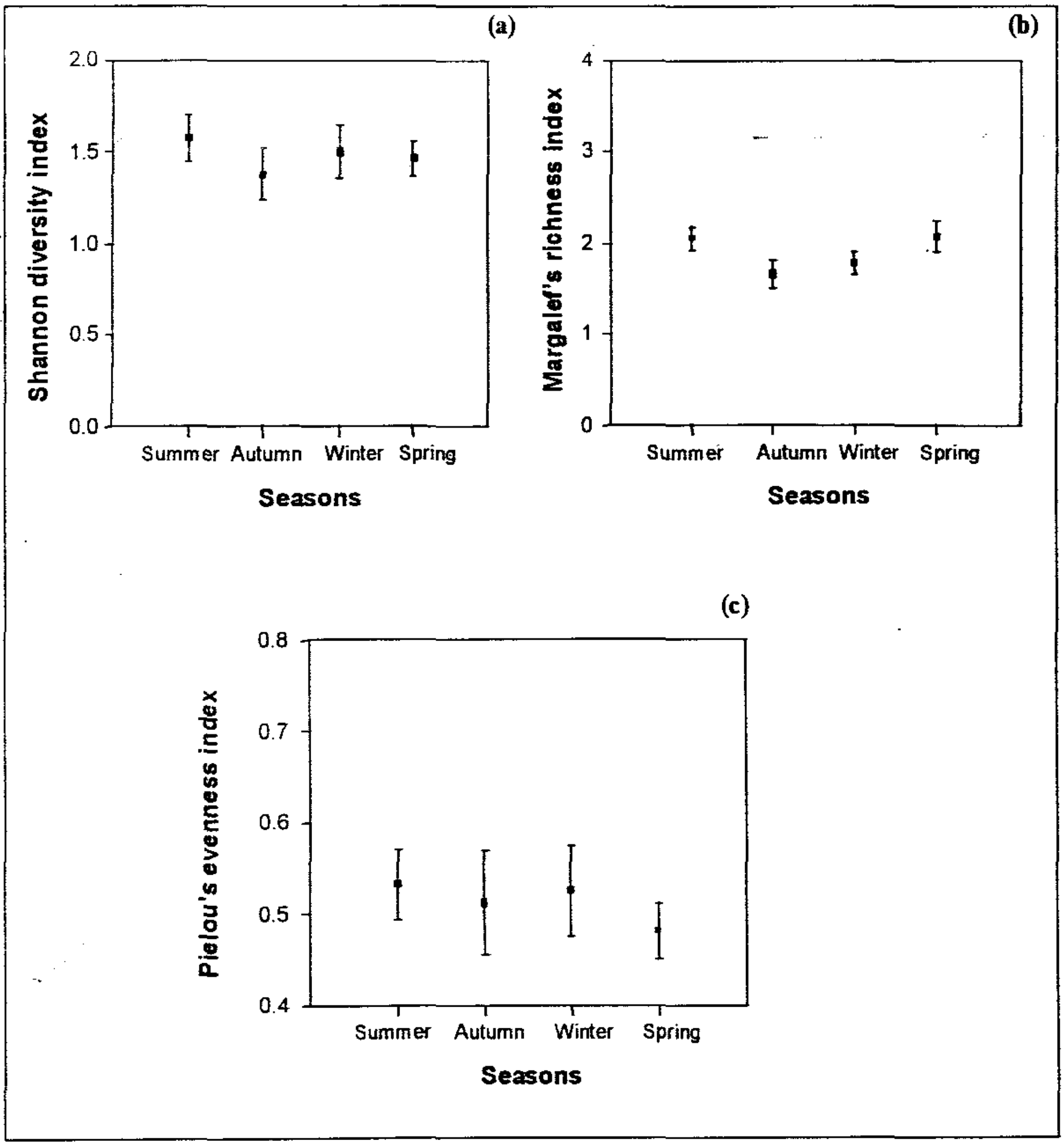

Figure. 4. 9. Mean seasonal differences in (a) diversity, (b) richness and (c) evenness indices from the Mlalazi estuary (1989-1991). (Bars indicate the mean ± 1SE).

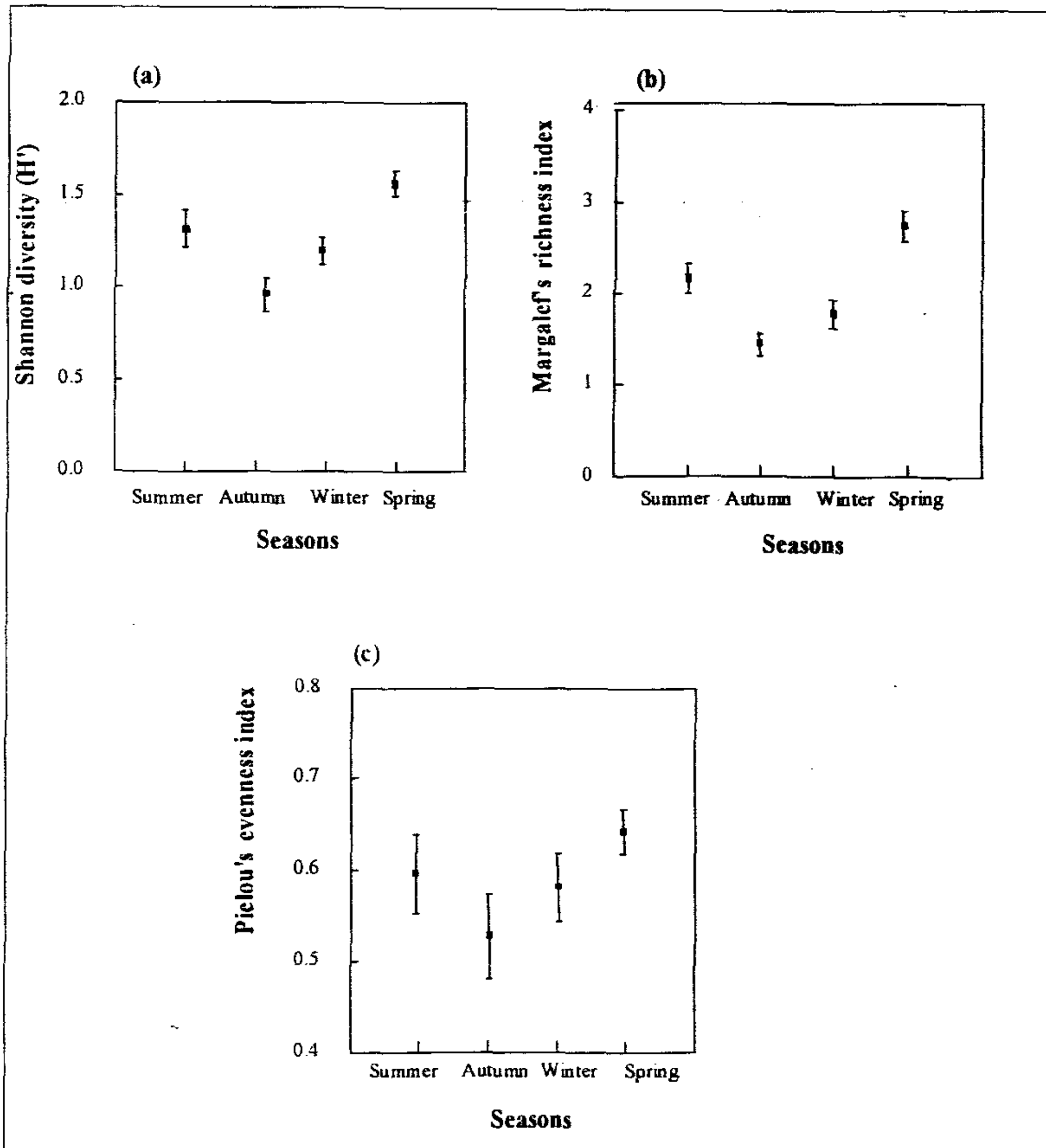

Figure. 4. 10 Mean seasonal differences in (a) diversity, (b) richness and (c) evenness indices from the Mlalazi estuary (1999-2000). (Bars indicate the mean ± 1SE).

4.4.5. Multivariate analysis

Classification and Ordination within each sampling period

Figures 4.11 and 4.12 represent the classification and MDS plots of the seasonal benthic data during the 1989-1991 and the 1999-2000 sampling periods, respectively. Five groups were identified on the basis of community structure during the 1989-1991 period at a similarity level of 68%. During the 1999-2000 sampling period, four community groups were identified at 55 % similarity level. The same general picture for the sampling periods is apparent in the MDS plots. The MDS analysis showed that the community groups separated out according to sites rather than according to season.

The groups delineated during the 1989-1991 sampling periods were: Group I consisted of Sites 1 and 2 that are in the upper reaches of the Mlalazi estuary. Group II comprised samples collected at Sites 5 and 6 in the lower reaches of the estuary. Group III consisted mostly of Site 3 samples with one Site 5 sample. Group IV comprised Site 4 samples and group V consisted of Site 6 samples. The general picture is that sites in the upper reaches (Sites 1 and 2) were grouped together, while those in the lower reaches (Sites 5 and 6) were grouped together. Sites in the middle reaches (Sites 3 and 4) formed their own groups (Groups III and IV, Figure 4.11). The analysis of similarity (ANOSIM) results indicated that there was a significant difference both among sites ($p < 0.05$) and among seasons ($p < 0.05$) although no clear trend was observed in the MDS analysis between seasons (Table 4.3).

The groups delineated during the 1999-2000 sampling period were: Group I which contained two Site 5 samples. Group II was made up of Sites 1, 2 and 3 which are sites in the upper and middle reaches of the estuary. Site 4 samples, found in the middle reaches, formed the third group (group III). Group IV was characterised by mostly Sites 6 and 7 samples with a few Site 5 samples. These sites were found in the lower and mouth regions of the estuary. The ANOSIM analysis showed that during the 1999-2000 sampling period, there were significant differences among sites ($p < 0.05$) while there were no significant differences among seasons (Table 4.3).

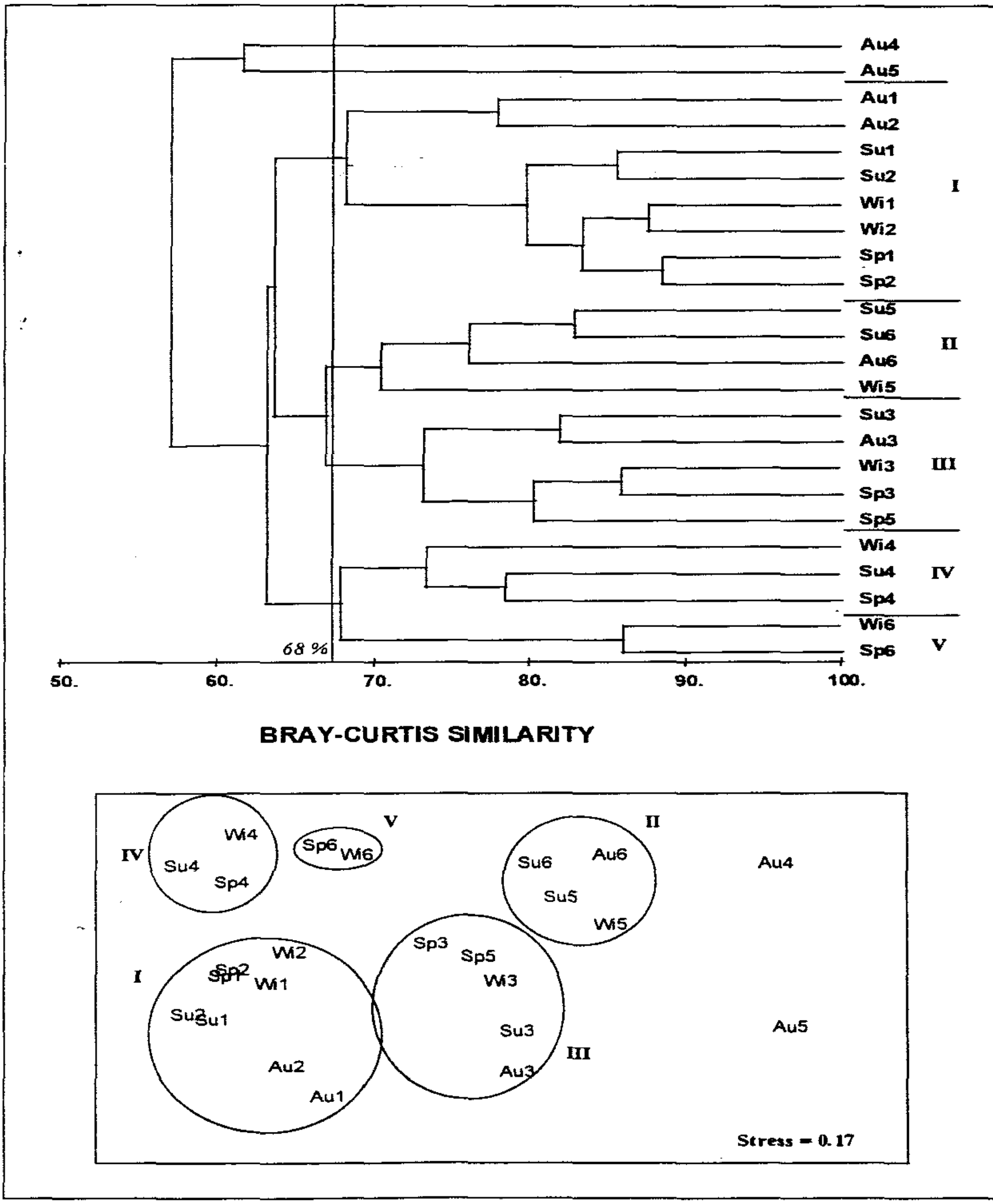

Figure. 4. 11. Bray-Curtis ranked similarity classification and MDS ordination of log transformed samples based on mean abundance collected at six sites from the Mlalazi estuary during the 1989 –1991 sampling period.

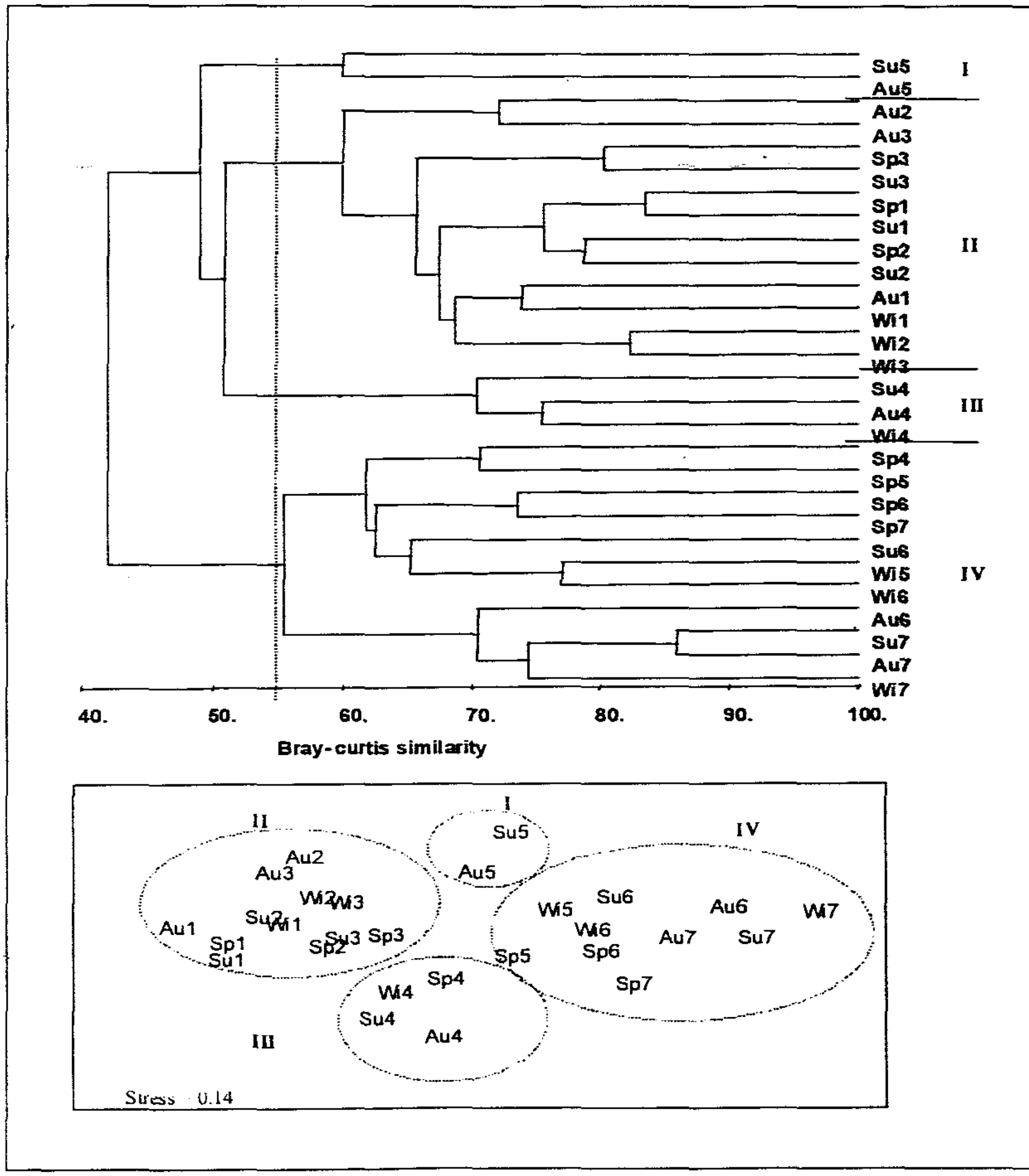

Figure. 4. 12. Bray-Curtis ranked similarity classification and MDS ordination of log transformed samples based on mean abundance collected at seven sites from the Mlalazi estuary during the 1999 –2000 sampling period.

Table 4.3. Analysis of Similarity (ANOSIM) test statistics (R) and significance level (p) of differences between sites and seasons for ranked sample similarities during the 1989-1991 and the 1999-2000 sampling periods. Levels marked with an asterisk indicate a significant difference ($p < 0.05$)

Period	**Site**		**Season**	
	R	p	R	p
1989-1991	0.585	0.01*	0.273	0.02*
1999-2000	0.685	0.01*	0.116	0.1

In order to get a clearer picture of the spatial change in the benthic community during the two sampling periods, and to test the hypothesis that the prawn farm effluent would have a marked impact on the benthic fauna at Site 3, classification and MDS ordination analyses were performed on the seasonally averaged data for each sampling period.

During both sampling periods, there was a very clear spatial gradient in the benthic community, from Site 1 in the upper reaches through to the mouth of the estuary (Figures 4.13 and 4.14). The benthic community at Site 3 did not appear to be markedly affected by the prawn farm effluent and fitted into the spatial gradient mentioned above. As such, the prawn farm effluent did not have such a large effect on the benthic fauna as what was expected in view of the poor water quality recorded at Site 3. This was further supported when an ANOSIM analysis was done to compare the benthic community at Site 3 with that from the other sampling sites. The results indicate that during both sampling periods, the Site 3 benthic community did not differ significantly from that recorded at the other sampling sites (Table 4.4). Surprisingly, the Site 4 benthic community was found to be most notably different from that observed at the other sites. This was found to be the case during both sampling periods.

During the 1989-1991 sampling period, Sites 1 to 3 were grouped together at a similarity level of about 80%. Sites 5 and 6 formed their own group while Site 4 was plotted far apart from the rest of the sites as indicated in the MDS plot (Figure 4.13).

During the 1999-2000 sampling period, Sites 1 to 3 formed a distinct group while Sites 5 to 7 were also grouped together. The MDS plot again indicated that Site 4 was quite dissimilar from the rest of the sites being plotted far from the other sites (Figure 4.14).

Comparison between the pre- and post prawn farm periods

In order to make a comparison between the sampling periods and to determine the extent of change in the benthic community between the two sampling periods, cluster, MDS ordination and ANOSIM analyses were performed. For this purpose, the averaged seasonal data from the two sampling periods were combined.

The clustering and MDS analysis clearly separated the pre-and post prawn farm periods indicating a change in the macrobenthic community between the two sampling periods (Figure 4.15). The ANOSIM results showed that the benthic communities of the two sampling periods were significantly different ($p < 0.01$, Table 4.4). The ANOSIM analysis also indicated that, with the exception of Site 2, there were significant differences in the benthic community between the pre- and post prawn farm benthic fauna at all the other sites ($p < 0.01$, Table 4.4). The MDS analysis also revealed that the different sites during the 1989-1991 period were plotted closer together than was the case during the 1999-2000 period. This indicated larger differences between sites and thus more variability in the 1999-2000 benthic community.

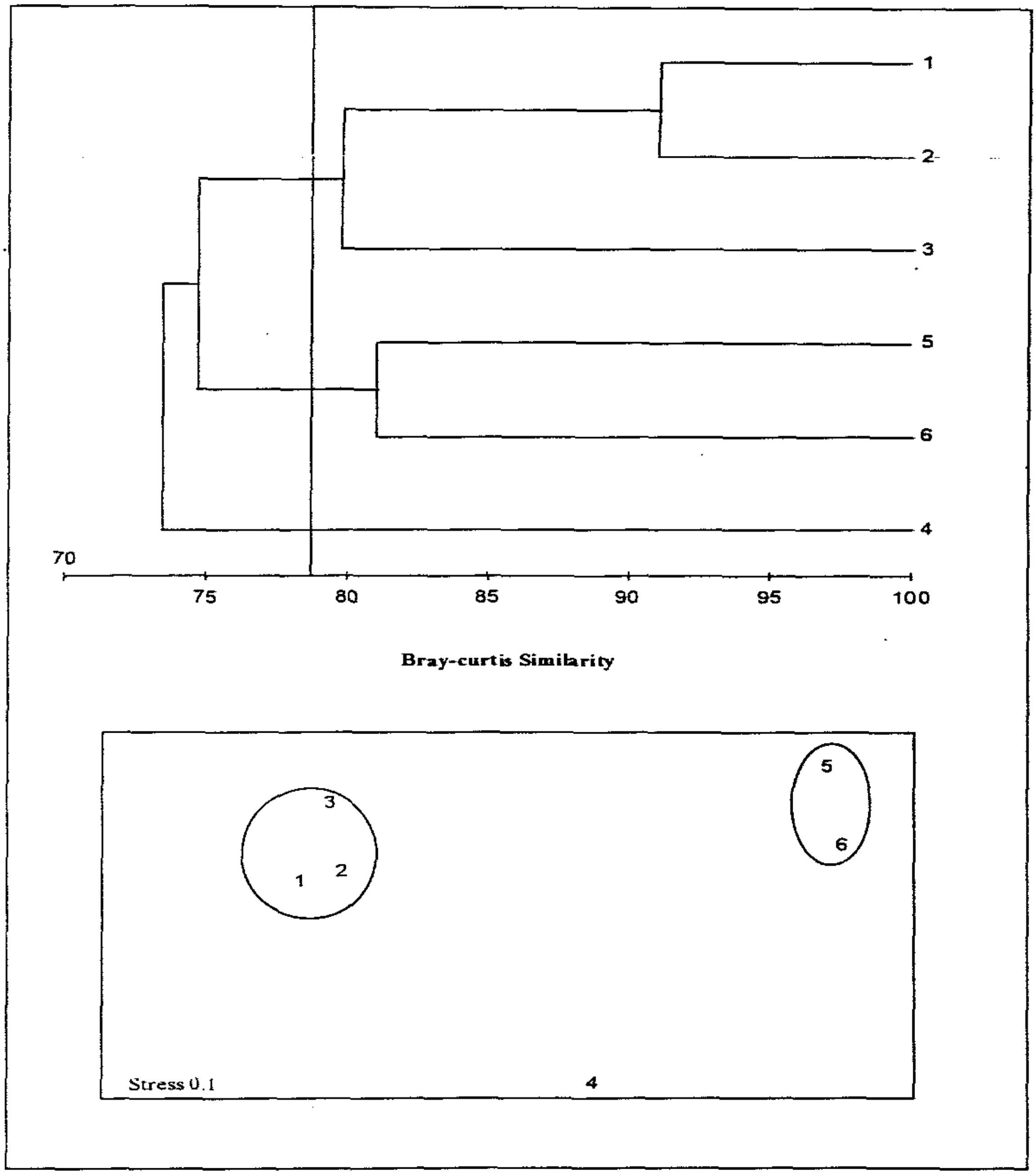

Figure 4.13. Bray-Curtis ranked similarity classification and MDS ordination of log transformed samples based on averaged data collected at six sites from the Mlalazi estuary during the 1989–1991 sampling period.

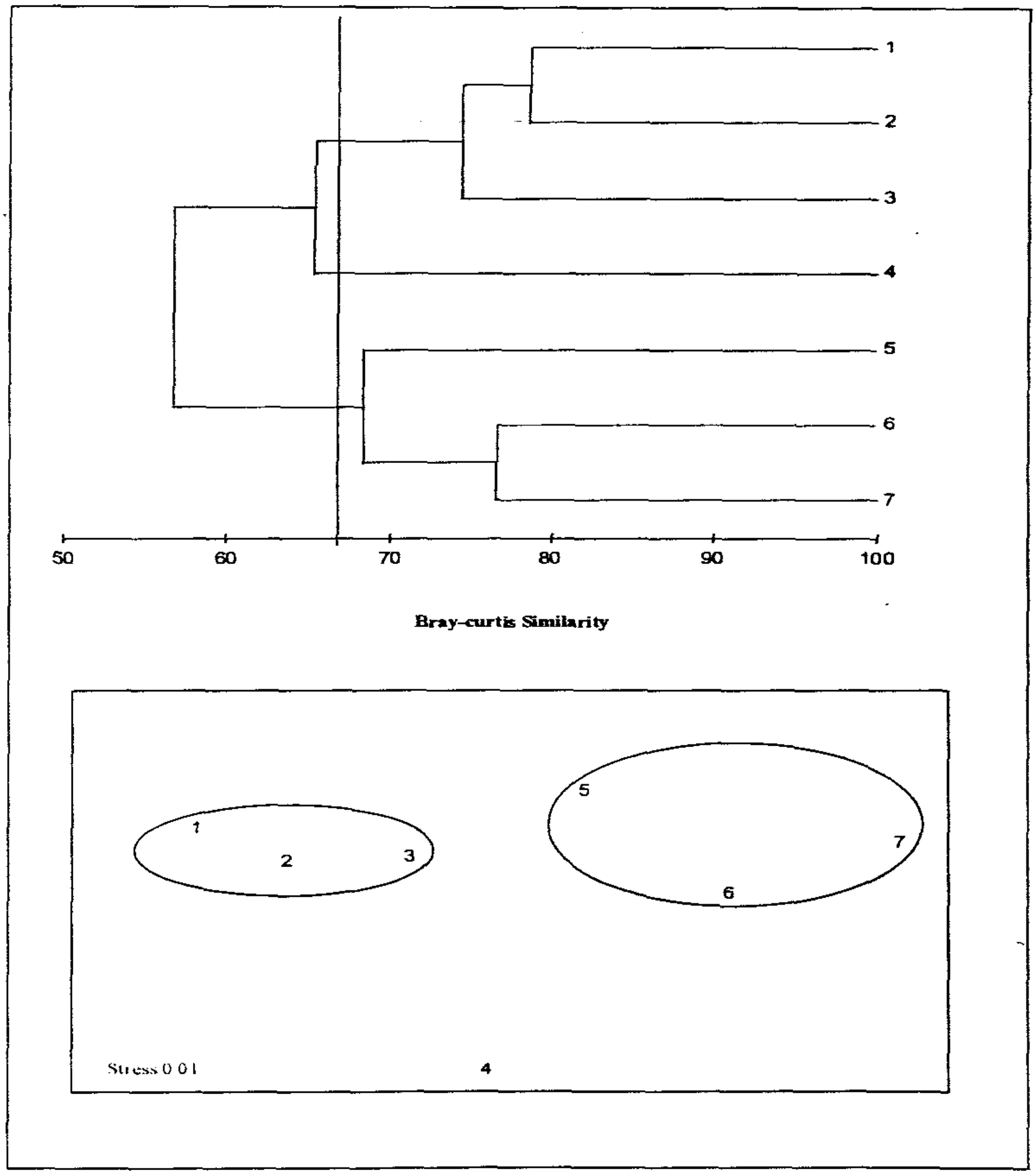

Figure 4.14. Bray-Curtis ranked similarity classification and MDS ordination of log transformed samples based on averaged data collected at seven sites from the Mlalazi estuary during the 1999–2000 sampling period.

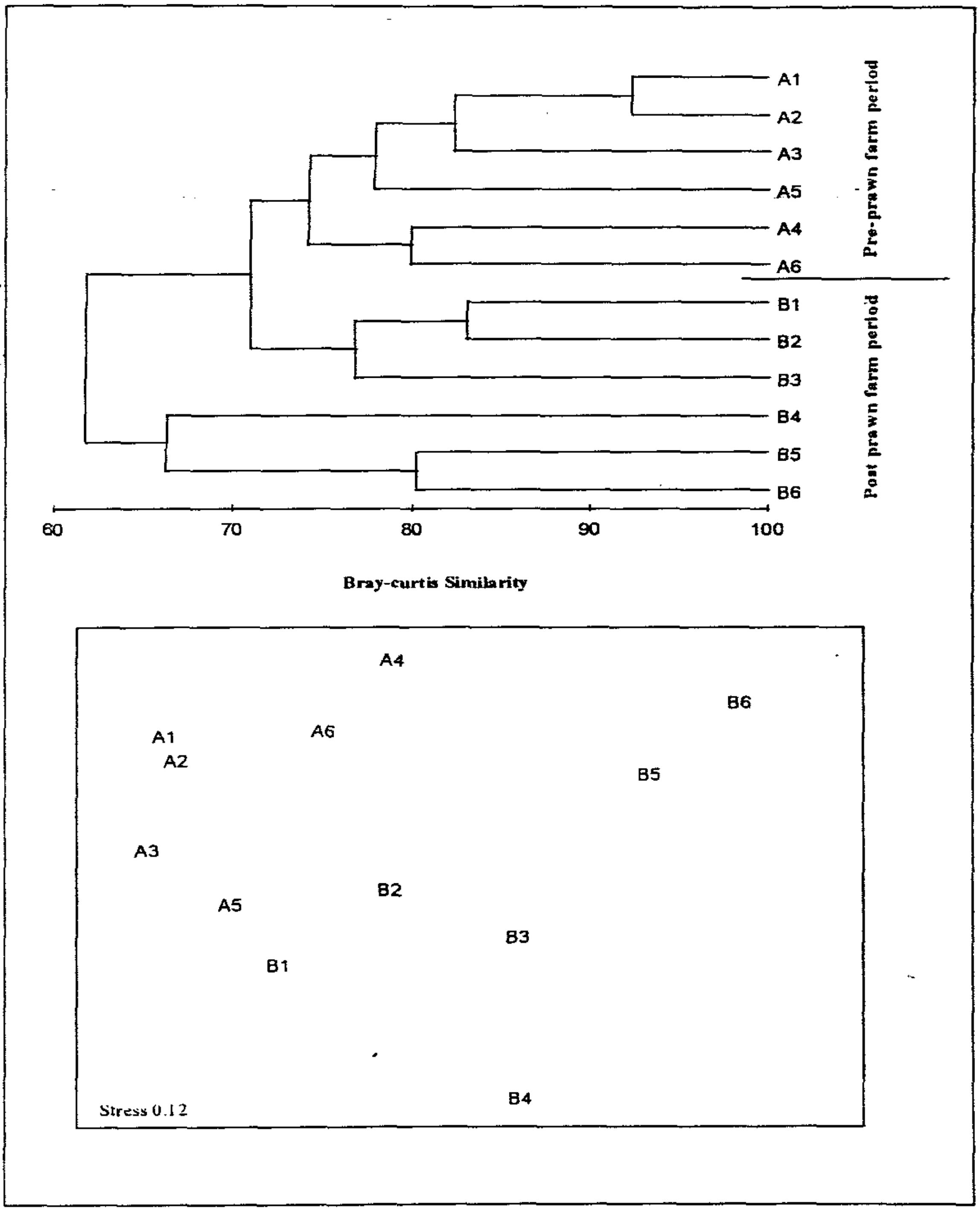

Figure 4.15. Bray-Curtis similarity classification and MDS ordination of log transformed samples based on average data collected from the Mlalazi estuary during the pre- and post prawn farm periods at six sites.

Table 4.4. Analysis of Similarity (ANOSIM) test statistics (R) and significance level (p) of differences between pre- (1989-1991) and post (1999-2000) prawn farm periods and within each period for ranked sample similarities. Levels marked with an asterisk indicate a significant difference ($p < 0.05$).

Between periods	**R**	**p**
Site 1 pre vs Site 1 post PF	0.8	0.003*
Site 2 pre vs Site 2 post PF	0.573	0.057
Site3 pre vs Site 3 post PF	0.99	0.03*
Site 4 pre vs Site 4 post PF	0.99	0.03*
Site 5 pre vs Site 5 post PF	0.086	0.003*
Site 6 pre vs Site 6 post PF	0.99	0.003*
All sites pre vs all sites post PF	0.489	0.004*
Within each period		
Site 3 vs rest of the sites ('99-2000)	0.154	0.94
Site 3 vs rest of the sites ('89-1991)	-0.003	0.45

Inverse analysis

Figures 4.16 and 4.17 represent the classification and MDS plots of the inverse analysis during the 1989-1991 and 1999-2000 sampling periods. Five groups were identified at a similarity level of 30 % during the 1989-1991 period, whereas four groups were identified at the 7 % similarity level in the 1999-2000 period. The organisms in each group were coded for easier interpretation. The species comprising each group and which site they came from are shown in Table 4.5 for the 1989-1991 period and Table 4.6 for the 1999-2000 period.

During 1989-1991, the MDS of the similarity matrix delineated five groups as follows (Figure 4.16): Group I consisted of capitellids and the brachyuran *P. blephariskios,* which were recorded at all sites but were most abundant at Site 3. Group II comprised of Polychaeta sp1 and *A. capensis*, these taxa being virtually absent at Site 3. Group III consisted of the polychaete, *G. tridactyla,* the isopod *L. laevigata* and the bivalve *M. virgiliae*.

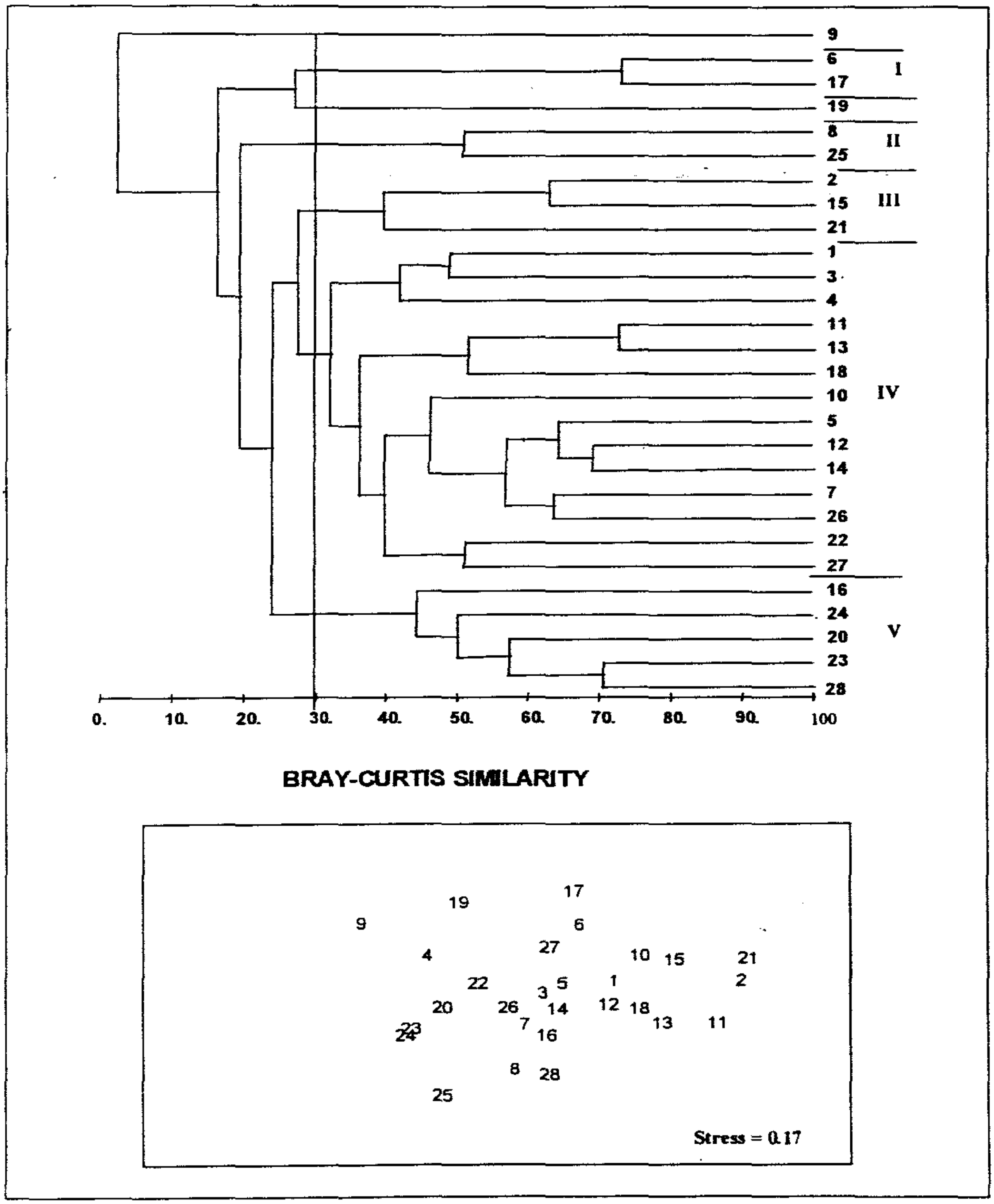

Figure 4.16. Inverse analysis showing a Bray-Curtis classification and MDS ordination plots based on samples collected at six sites during the 1989-1991 period.

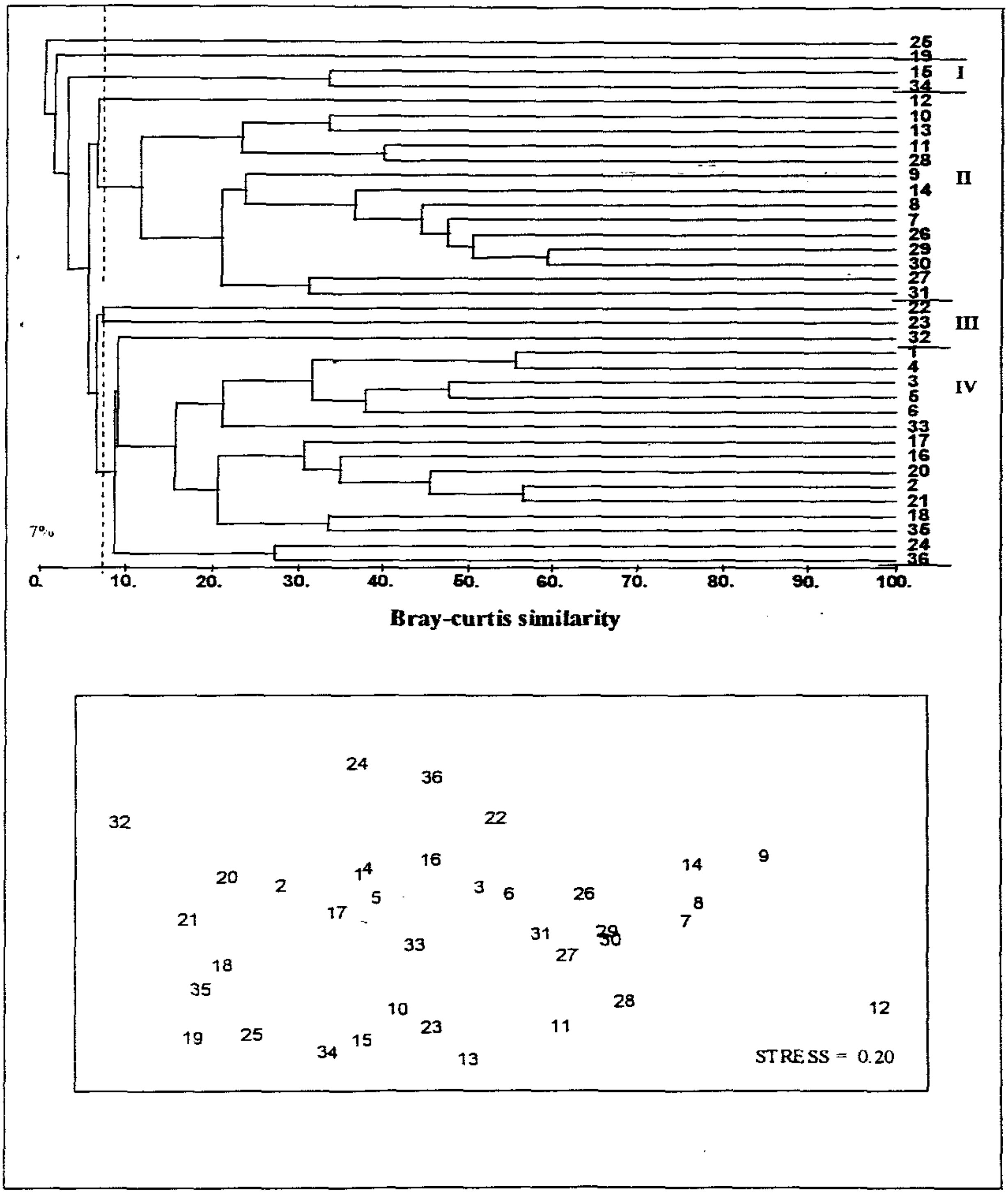

Figure 4.17. Inverse analysis showing a Bray-Curtis classification and MDS ordination plots based on samples collected at seven sites during the 1999-2000 period.

Table 4.5. Species groups distinguished by inverse analysis based on data collected during the 1989-1991 period. Species numbers refer to those used in the classification and MDS plots (see Figure 4.13).

Group	Code	Species	Site
1	6	Capitellid sp.	1-6
	17	*P. blephariskios*	
2	8	Polychaeta sp1	1,2,4,5,6
	25	*A. capensis*	
3	2	*G. tridactyla*	1-6
	15	*L. laevigata*	
	21	*M. virgiliae*	
4	1	*D. arborifera*	1-6
	3	*G. longipinnis*	
	4	*G. convoluta*	
	5	*Prionospio sp*	
	7	*M. enigmatica*	
	10	*A. digitalis*	
	11	*C. triaenonyx*	
	12	*G. bonnieri*	
	13	Amphipod sp.	
	14	*P. stuhlmanni*	
	18	*H. orbiculare*	
	22	*E. pauperculata*	
	26	*Nassarius sp*	
	27	*Cumacea sp.*	
5	20	*M. littoralis*	3,4,5,6
	23	*D. hepatica*	
	24	*H. ludwigii*	
	28	Hirudinea	

These were recorded at all sites but were more abundant in the upper reaches. Representative taxa of Polychaeta, Tanadacea, Amphipoda, Isopoda, Branchyura, Bivalvia and Gastropoda were found in group IV except for those taxa found in group III. Group V comprised of the bivalves *M. littoralis*, *D. hepatica* and *H. ludwigii*, these taxa were mostly recorded in the lower reaches although also found elsewhere in the estuary.

During the 1999-2000 period, the inverse MDS plot of the benthic community revealed four groups mainly according to sites (Figure 4.17). Group I comprised oligochaetes and sipunculids, i.e. taxa recorded at Sites 1, 5 and 7. Group II consisted of polychaetes,

Table 4.6. Species groups distinguished by inverse analysis based on data collected during the 1999-2000 period. Species numbers refer to those used in the classification and MDS plots (see Figure 4.14).

Group	Code	Species	Site
1	15	Oligochaeta sp1	1,5,7
	34	Sipunculid	
	7	*Tharyx sp.*	5,6,7
2	8	*H. filiformis*	
	9	*M. cincta*	
	10	*Phyllodocea sp.*	
	11	*G. capensis*	
	13	Polychaeta sp2	
	14	*A. parva*	
	26	*Nassarius sp*	
	27	*Assiminea ovata*	
	28	*Melanoides tuberculata*	
	29	*D. hepatica*	
	30	*Macoma sp.*	
	31	*E. pauperculata*	
	22	*P. blephariskios*	4
3	23	Brachurian larvae	
	1	*D. arborifera*	1,2,3
	2	*C. keiskamma*	
	3	*Prionospio spp.*	
	4	Capitellids.	
4	5	*D. ornata*	
	6	*Glycera spp.*	
	16	*A. digitalis*	
	17	*G. lignorum*	
	18	*C. triaenonyx*	
	20	*L. laevigata*	
	21	*E. natalensis*	
	32	Stomatopoda	
	33	Cumacea sp.	
	35	Hirudinea	
	36	*S. capensis*	

gastropods and bivalves, taxa recorded at Sites 5, 6 and 7, which represented the lower and mouth regions of the estuary. Group III consisted of brachyurans found mostly at Site 4, predominantly *P. blephariskios* and Brachurian larvae. Group IV was dominated by polychaetes with amphipods and isopods also making a contribution. These organisms were found mostly at Sites 1, 2 and 3, which represent the middle and upper reaches of the estuary.

4.5. Discussion

The invertebrate macrofauna form one of the most abundant biotic components in estuaries and has been widely used as indicators of water quality and status of the estuary (Dauer 1993, Harvey, *et. al* 1998). These organisms are generally in the size range of >500µm (Brown & McLachlan 1990). They include representatives of taxa such as Polychaetes, Tanaidaceae, Amphipoda, Isopoda, Brachyura, Bivalva and Gastropods. The polychaetes, molluscs and crustaceans usually predominate the benthic community of estuaries (Day 1981).

A measure of the abundance and diversity of macrobenthos in an estuary provides information on the status of that system (Hynes 1972). The presence or absence and relative abundance of macrobenthic species can be used to assess disturbance events that occurred before sampling. Water chemical samples, which give instantaneous record of prevailing conditions, do not provide such information (de Moor, *et. al* 2000). Macrobenthic species, because of their relatively sedentary nature, are vulnerable to ecological disturbances. Certain species may be eliminated from sections of the estuary for a considerable period as a result of ecological disturbance, therefore, species composition of macrobenthic communities can provide information on the nature of disturbances e.g. chemical or organic pollution.

4.5.1. Community structure

A total of 28 and 36 benthic taxa were recorded in the Mlalazi estuary during the 1989-1991 and 1999-2000 sampling periods, respectively. These relatively low numbers of taxa in the Mlalazi estuary are a poor reflection of a subtropical estuary since subtropical estuaries are known to be highly diverse. Hill (1966) recorded 84 macrobenthic taxa in the Mlalazi estuary, which he suggested to be poor for a subtropical estuary in relation to the 113 taxa recorded by Mackay and Cyrus (1999). Day (1974) recorded 378 taxa in the Morrumbene estuary.

During both the sampling periods, the polychaete *Prionospio spp.*, capitellids and *D arborifera,* and the ocypodid crab *P. blephariskios* were recorded throughout most of the estuary. There were no marked differences in benthic composition between the two sampling periods as the ANOSIM analysis indicated although some taxa were only recorded during one of the two sampling periods 1999-2000 period. The polychaete *D. ornata* was only recorded during the 1999-2000 period while taxa such as *M. enigmatica, P.stuhlmann,* and *M. virgiliae* were recorded only during the 1989-1991 period. Other taxa recorded during both periods such as Hirudinea, Sipunculids and Cumacea comprised either a single record or occurred at very low densities at only one location.

The polychaete worms are generally abundant members of an estuarine macrofauna communities and are represented by more genera than any other group of organisms (Brown & McLachlan 1990). Mackay and Cyrus (1999) reported that the polychaetes were the most abundant macrobenthos in the Mhlathuze estuary. This was true for the Mlalazi estuary where the polychaetes were the dominant group in terms of density and the number of genera during both sampling periods. The capitellids and *Prionospio spp.* numerically dominated the Mlalazi estuary throughout the study period. Other species that made significant contributions were the polychaete *D. arborifera,* the amphipod *C. triaenonyx*, the tanaid *A. digitalis* and the brachurian *P. blephariskios*. The polychaete *M. enigmatica* was numerically important during 1989-1991 while *D. ornata* was numerically important during 1999-2000 period.

Prionospio spp and capitellid polychaete, are well documented as being highly opportunistic organisms capable of rapid colonisation of substrata subjected to perturbations or organic enrichment (Gray 1981, Brown & McLachlan 1990). The Tongati estuary was reported to be dominated by *Prionospio spp.* with the zoobenthos being impoverished (Blaber, *et. al* 1984). This system was classified as degraded due to anthropogenic influences, which lead to low salinity and oxygen tension. Opportunistic species usually thrive in such conditions. Mackay and Cyrus (1999) reported dominance of *Prionospio spp.* and capitellids at selected sites in the Mhlathuze estuary. These authors suggested that the abundance of these opportunistic polychaetes was due to these sites being more disturbed or that the sediments characterising these sites were more fine-grained. Large increases in the density of opportunistic species at sites where there is material disposal have also been observed in marine and freshwater environments (Harvey, *et. al* 1998). The Mlalazi estuary was dominated by these two opportunistic polychaetes which were particularly abundant at sites that consisted of silt or fine sediments (Sites 3 and 4 during 1989-1991, Sites 4 and 7 during 1999-2000) and during the winter season. However the capitellids were more abundant during the 1989-1991 period whereas *Prionospio* polychaetes were most dominant during the 1999-2000 period. During winter, estuaries in the subtropical region usually experience reduced freshwater inflow due to low rainfall, therefore the system does not get flushed properly. Due to such conditions, the nutrients discharged from rivers and agricultural runoff, accumulate in the estuary resulting in high nutrient loads. Although the source of a high nutrient load in estuary during the 1989-1991 period is not known, it is speculated that agricultural runoff were the potential source. Mann, *et. al* (1996) reported that sugar cane was cultivated along the upper and middle reaches of the estuary and could pose problems in the form of runoffs. The chemical results of the present study indicated that the prawn farm is discharging high concentrations of nutrients into the estuary and this coupled with reduced freshwater inflow led to accumulation of nutrients resulting in high nutrient concentration during autumn-winter season. Elevation of nutrients is usually accompanied by oxygen tension and opportunistic species thrive in such conditions (Harvey, *et. al* 1998). The macrobenthic data indicated dominance of opportunistic species during autumn and winter seasons of the 1999-2000 period, with an apparent decrease in species diversity.

Perturbed environments are dominated by r-selected (opportunistic) species with k-selected (rare) species being less favoured. The densities of opportunistic species are usually higher due to adaptive strategies that allow rapid local recruitment by such species in disturbed habitats (Dauer 1993). The accumulation of nutrients due to potential agricultural run off and prawn farm effluent in the estuary during 1989-1991 and 1999-2000 sampling periods is suggested to have resulted in the proliferation of the opportunistic species *Prionospio spp.* and capitellids. This may have been the case during the 1989-1991 period, however, MDS ordination analysis of the 1999-2000 data has indicated that the prawn farm had no visible effect on the abundance and distribution of the macrobenthic community. Indications are that it was mostly variations in physical conditions such as substrate type and salinity that were responsible or played a key role in structuring the community patterns of the two sampling periods. The capitellids are classified as first stage colonists while *Prionospio* polychaetes are classified as the second stage colonists (Day 1967b, Mackay 1996). The capitellids were more abundant than *Prionospio* polychaetes during the 1989-1991 period while the latter polychaete became dominant during the 1999-2000 period than the former. The decline in abundance of the first stage colonists (Capitellids) during the 1999-2000 period seem to have presented the opportunity for the second stage colonists (*Prionospio*) to proliferate. This also seemed to decrease the overall density of fauna as compared to the 1989-1991 period. Although it is not clear why there was a decline in capitellid densities during the 1999-2000 period, however, due to the high dominance of the first stage colonists during the 1989-1991 period it seemed like this period was more perturbed than the 1999-2000 period. Generally, the abundance of these taxa indicated perturbed conditions during both the sampling periods since an ecological survey of the estuary by Hill (1966) revealed no capitellids or *Prionospio spp.* in the Mlalazi estuary.

The polychaete *D. ornata* is also regarded as an opportunistic species since it has been suggested that the presence of this species in brackish estuaries may be associated with a high organic content in the sediment (Mackay 1996). However, in the Mlalazi estuary during the 1999-2000 period, this polychaete was found in the upper and lower regions of estuary. These regions were characterised by coarse sediment with a low to moderately low organic content. In areas characterised by high organic content, i.e. Sites 4 and 7, densities of *D. ornata* were reduced. Perturbations that occurred in the estuary in winter seemed to have resulted in an increased density of *D. ornata*, confirming the opportunistic nature of this species. The absence of this polychaete during 1989-1991 is not clearly understood. However, due to the high dominance of first and second stage opportunists, the lack of available areas for colonisation might have been the cause. As such, *D. ornata*, can be regarded as a third stage coloniser which also proliferates in disturbed environments.

Day (1981) reported that the polychaete *M. enigmatica* is known from warm temperate estuaries where it is found in compact mud. Its occurrence in southern Africa is sporadic where it is reported as abundant in Kosi Bay while absent in the Morrumbene estuary. Hill (1966) reported that this polychaete was present in fairly high numbers in the Mlalazi estuary. This was also the case during the 1989-1991 period, however *M. enigmatica* was not recorded during 1999-2000 period. The lack of compact mud as the habitat might have caused the disappearance of this polychaete during 1999-2000 since the estuary contained mainly course to fine sediments with no compact mud or silt as was the case during the 1989-1991 period.

Amphipods are often regarded as dominant organisms in most estuaries (Stoner & Acevedo 1990). The amphipod, *G. lignorum,* is found in estuaries on the East coast of southern Africa and in certain coastal lakes such as Lake Sibaya in Zululand and Swartvlei in the Knysna district (Bishop 1973). Owen and Forbes (1997) also reported the occurrence of *G. lignorum* in Lake St Lucia while it was also found in the Nhlabane estuary (Vivier, *et. al* 1998). Another amphipod recorded in the Mlalazi estuary together with *G. lignorum* was *C. triaenonyx*. Boltt (1969) recorded both these amphipods as part of the relict estuarine fauna of Lake Sibaya. These two amphipods are found in a range of sediments but are common in sandy-mud where shallow burrows are made. Throughout the study period, *C. triaenonyx and G. lignorum* were restricted to the low salinity regions in the upper reaches of the Mlalazi estuary where mainly sandy sediment occurred. Although these two amphipods were found inhabiting the same habitat, *C. triaenonyx* was more abundant than *G. lignorum*. This is in contrast to what was reported by Mackay (1996) in the Siyaya estuary where *G. lignorum* was more abundant than *C. triaenonyx*. Mackay (1996) concluded that an element of competition had arisen between the two amphipods in the Siyaya estuary. In the Mlalazi estuary it is not clear why *C. triaenonyx* seemed more successful of the two, however, *C. triaenonyx* is able to occupy substrates that have a muddy consistency as compared to *G. lignorum* (Stoner & Avecedo 1990).

Blaber, *et. al* (1983) have reported the occurrence of increased densities of *A. digitalis* in muddy substrata. They concluded that increased densities were due to greater availability of a detrital food source. This confirms the abundance of *A. digitalis* in the middle reaches of the estuary. This region was characterised by muddy substrata that were organically rich.

The burrowing ocypodid crab, *Paratylodiplax blephariskios,* is endemic to the subtropical East coast of southern Africa where it forms an important component of benthos in soft-bottom substrata (Owen, *et. al* 2000). Hill (1966) also reported that *P. blephariskios* was common in the middle reaches of the Mlalazi estuary which consisted mainly of soft mud. Owen, *et. al* (2000) found that the distance from the mouth and the combination of median particle size and sorting coefficient were the main variables affecting the distribution of *P. blephariskios* in the St. Lucia and Mhlathuze estuaries. During the study period this ocypodid crab was found only in the muddy middle reaches of the estuary (Site 3 and 4) where it was highly abundant compared to other species. This area of the estuary was also characterised by a high organic content. The combination of muddy substrate and a high organic content seemed to be the reason for the abundance of this crab in the middle reaches of the Mlalazi estuary.

Gastropods and bivalves were restricted to the lower and mouth regions of the Mlalazi estuary, except for *M. virgiliae*. This species, recorded only during 1989-1991, was found along the length of the estuary. The bivalve *M. virgiliae* was also found in sediments that consisted of sand, mud, small stones and boulders in the Kariega estuary (Hodgson 1986). The high tolerance of this bivalve to different sediment types perhaps explains its wide distribution throughout the Mlalazi estuary. The restriction of other gastropods and bivalves to the mouth region seemed to be related to their sediment and salinity preference, since they are mostly adapted to soft sediments and high salinity. However, it is also possible that restriction in the mouth region may be due to their feeding mode since these species are filter feeders. Branch and Branch (1981) reported that the bivalve *Mactra sp.* occurred in high densities in areas where currents were strong (mouth region) and were surely partly responsible for clarity of water in the Langebaan Lagoon. The bivalve *Macoma littoralis* is also found at marine salinities in sandy sediments, however this bivalve has been recorded in high densities in the middle reaches of the Swartkop and Kariega estuaries (McLachlan & Grindley 1974, Hodgson 1986). In the Mlalazi estuary during 1989-1991, *M. littoralis* was found in the middle and lower reaches. During the 1999-2000 period, densities of *Macoma sp.* were greatly reduced when compared to the 1989-1991 period. A decrease in abundance of this species during the 1999-2000 period is not clearly understood, however, it is suggested that competition with and dominance of the crab *P. blephariskios* in the middle reaches might have reduced the density of this bivalve. McLachlan and Grindley (1974) claimed that competition with *Upogebia* restricted the distribution of *M. littoralis* in the Swartkops estuary.

4.5.2. <u>**Species diversity**</u>

Increased abundance of benthic organisms with a corresponding decrease in diversity and evenness are generally considered to be caused by an increasing level of stress or disturbance (Clarke & Warwick 1994). High densities of opportunistic polychaetes at Site 3 and 4 (middle reaches) during the 1989-1991 sampling periods resulted in these sites being less diverse than the rest of sites whereas Site 4 during the 1999-2000 period had lower species richness than the other sites. Low diversity, species richness and evenness scores indicated that this site consisted of a few species with numerical distribution among species confined to a few dominant taxa while other species made small or insignificant numerical contributions. Dominance of opportunistic polychaetes along with a decline in species diversity usually indicates a disturbed area. Low numbers of fauna were observed in Sta. Rite River estuary which is affected by fishpond effluents, where opportunistic species dominated over other groups in the areas of the estuary characterised by high organic matter and nutrients (Galope-Bacaltos & Diego-McGlone 2002).

The physical conditions, agricultural run-off and the prawn farm as the potential sources of anthropogenic disturbances in the estuary were considered to have increased the levels of stress in the middle reaches of the estuary. The general pattern of species diversity, richness and evenness in the Mlalazi estuary indicated a slightly more diverse benthic community during the 1999-2000 period with higher species richness and evenness than during the 1989-1991 period. Recent theories on the influence of disturbance or stress on diversity suggest that in situations where disturbance is minimal, species diversity is reduced because of competitive exclusion between species while at a slightly increased level of disturbance, competition is relaxed resulting in an increased diversity (Scharler, *et. al* 1997). This might have been the case in the Mlalazi estuary where it is speculated that agricultural run-off caused disturbance during 1989-1991, which allowed dominance of opportunistic capitellids. The competition for habitat resulted in exclusion of some species in favour of the capitellids resulting in a decline in species diversity. During the 1999-2000 period, it is speculated that the level of disturbance increased due to physical factors such as nature of the substrata and nutrient rich effluents being discharged from the prawn farm into the estuary, which resulted in reduced competition for habitat between species. Such an increase in level of disturbance may have caused a slight increase in diversity during the 1999-2000 compared to the 1989-1991 period. However since the prawn farm was shown to have little or no effect, the macrofaunal patterns that were noted seemed to be a function of the response of species to such factors as sediment characteristics and salinity regime.

4.5.3. Changes in benthic community

Classification and ordination performed on the benthic samples showed distinct groups, which indicated a change in the benthic community along the length of the estuary and between the two sampling periods. The spatial gradient in benthic communities of the both periods is clearly indicated in the MDS plots. The clustering and MDS plots further indicated that the prawn farm activities had little or no effect on the macrofaunal community of the 1999-2000 period. Due to the close proximity of Site 3 to the prawn farm outlet during the 1999-2000 period, it was expected that this site would clearly show the impact of the prawn farm effluent. However cluster analysis indicated that Site 4 and not Site 3 was dissimilar from the rest of the sites. This is confirmed by the fact that ANOSIM results indicated that there were no significant differences between Site 3 and the rest of the sites during the 1999-2000 period. The reason why Site 4 was dissimilar from the rest of the sites during both the sampling periods is thought to lie on physical conditions which in turn were responsible for the abundance and composition of the benthic community at this site.

Cyrus, *et. al* (1999) also reported changes in the macrobenthos of the Mhlathuze estuary due to abiotic factors. Changes in community structure that was reported by Vivier, *et. al* (1998) in the Nhlabane estuary were also due to physical factors. The community structure was affected by low salinity conditions due to drought and prolonged closure of the estuary. Owen and Forbes (1997) reported changes in the community structure of the St Lucia estuary that indicated the variable nature of benthic communities under different salinity conditions. Considerable differences in densities of organisms according to nature of the substratum were also reported by Blaber, *et. al* (1983) in the St. Lucia estuary.

Substrate type and salinity were the important factors which determined the Mlalazi benthic community structure during both the sampling periods. During the pre-prawn farm period Site 4 was dominated by opportunistic species which were however reduced in density compared to other sites, especially Site 3, which had a similar substrate type. Highest turbidity and salinity values were however recorded at Site 4 as compared to the other sites and such conditions coupled with a silty substrate are known to adversely affect benthic fauna leading to a decrease in abundance. During the post prawn farm period Site 4 was characterised by fine mud and a high organic content as compared to the rest of the sites. The ocypodid crab *P. blephariskios* dominated this site while it was virtually absent from the rest of the sites. The inverse analysis also produced species groups corresponding with sites demonstrating that species groups occurred along an increasing salinity gradient. The above discussion thus indicates that the principal factors that seemed responsible for a change in benthic community of the two sampling periods in the Mlalazi estuary were abiotic in nature. The importance of environmental factors in influencing biotic patterns has been greatly discussed in benthic studies and the effects of the physical factors on the macrobenthos of the Mlalazi estuary are discussed in detail in Chapter 5.

The benthic community structure of the 1989-1991 period was not different compared to the community structure of the 1999-2000 period as indicated by two-way crossed ANOSIM. This was attributed to the fact that most of the species recorded were common to both sampling periods and the pattern of species grouping according to the inverse analysis was also similar to both the sampling periods.

4.6 Conclusion

Macrobenthic organisms are representative of location being sampled, allowing monitoring of spatial and temporal changes in response to natural and anthropogenic disturbances. The macrobenthic community of the Mlalazi estuary during both the sampling periods was dominated by opportunistic taxa, which are indicative of variable conditions. This indicated that both the periods were seemingly disturbed which lead to the conclusion that the estuary is not in the same pristine condition as reported by Hill (1966) and Mann, *et. al* (1996). Physical conditions and anthropogenic disturbances have been reported to adversely affect the estuarine environment and consequently the estuarine community. This has been greatly discussed above whereby the results indicated disturbances in the macrofaunal community of the two sampling periods were due to environmental factors. The prawn farm, did however, not appear to adversely affect the benthic community in the region adjacent to the effluent discharge during this study. Even though the abundance, composition and distribution of macrobenthos in the Mlalazi estuary during the two sampling periods were largely due to the physical conditions with no apparent effect from the prawn farm effluent, there is a potential that the nutrient rich effluent from the farm may impact the estuary in future if the discharge is allowed to continue. The potential impact on the estuary cannot be ignored, therefore, there seemed to be a need to continue monitoring the estuary. Such monitoring of the estuary would add information on the changes in the biotic community associated with physical conditions and also enable early detection of the prawn farm activities in the estuarine environment.

CHAPTER 5

Relation between the benthic community and physico-chemical characteristics of the estuary

5.1. Introduction

Biological surveys on benthic communities usually result in complex biotic and environmental data from which patterns and relationships need to be extracted. A major area of practical community studies is the attempted explanation of community patterns by linking the biotic analysis to physical or contaminant data from the same set of samples. These data could be natural variables describing the physical properties of the aquatic system, e.g. median particle size, salinity, temperature, turbidity etc., or variables such as concentrations of ammonia, nitrate, nitrite, etc. The challenge in such an analysis is to examine the extent to which the physico-chemical data is related to the observed biological patterns (Clarke & Warwick 1994).

Biotic data is often best described by a multivariate summary, such as an MDS ordination. The relation between the biotic community and environmental variables can be visualised by representing the values of environmental variables as symbols of differing size and superimposing these symbols on the biotic ordination of the corresponding samples. This is effective in noting any consistent differences in environmental variables between biotic clusters. This chapter aims at determining the effects of physico-chemical parameters on the macrobenthic community of the Mlalazi estuary.

5.2. Materials and Methods

The environmental data collected during 1989-1991 and 1999-2000 were used to highlight changes in the benthic community of the Mlalazi estuary. For the 1989-1991 period, only physical variables (salinity, temperature, oxygen, turbidity and depth) were used for superimposition onto the seasonal biotic MDS ordination. Since sediment was collected only once during the 1989-1991 period, a separate biotic MDS ordination was used for the superimposition of sediment data. This was done in order to perform the BIO-ENV procedure, which requires that the biotic matrix and the environmental data should correspond, i.e. equal number of samples. For the 1999-2000 period, physical and chemical water quality data were used, with each variable being superimposed onto the seasonal biotic MDS ordination.

The relation between the biotic community and physico-chemical character of the Mlalazi estuary was investigated in two ways. Firstly, the environmental data was superimposed onto the biotic ordination using the CONPLOT procedure in the PRIMER statistical package. The premise here is that if a certain environmental variable was responsible for structuring the biotic community, the abiotic MDS ordination would be similar to the biotic MDS ordination. Secondly, in an attempt to find a suite of environmental variables that best describe changes in the benthic community of the Mlalazi estuary during the study period, the program BIO-ENV in the statistical package PRIMER was used. The non-parametric weighted Spearman (Harmonic) rank correlation (p_w) was used to test the correlations between the biotic and abiotic similarity matrices determined from the BIO-ENV procedure (Clarke and Warwick 1994). The weighted Spearman rank correlation is given as:

$$P_w = 1 - \frac{6}{N\,(N-1)} \quad \Sigma_{i=1}^{N} \; \frac{(r_i - s_i)}{r_i + s_i}$$

P_w lies in the range (-1,1) with values around zero corresponding to the absence of any match between the two patterns.

5.3. Results

5.3.1. Physical variables during the 1989-1991 period

Figures 5.1 to 5.2 show the superimposing of the 1989-1991 physical data on the biotic data.

The physical parameters that best matched the observed patterns in benthic community of the 1989-1991 sampling period were salinity, oxygen, turbidity and median particle diameter. Salinity is known to be an important feature structuring benthic community in estuaries and it typically increases from the head to the mouth region of the estuary. This is clearly shown in the salinity - superimposed plot where group I was distinguished from the rest of the groups. Group I consisted of samples from Sites 1 and 2 in the upper reaches of the estuary. Sites 1 and 2 were characterised by low salinity values that increased gradually towards the middle reaches (Sites 3 and 4) and lower reaches (Sites 5 and 6).

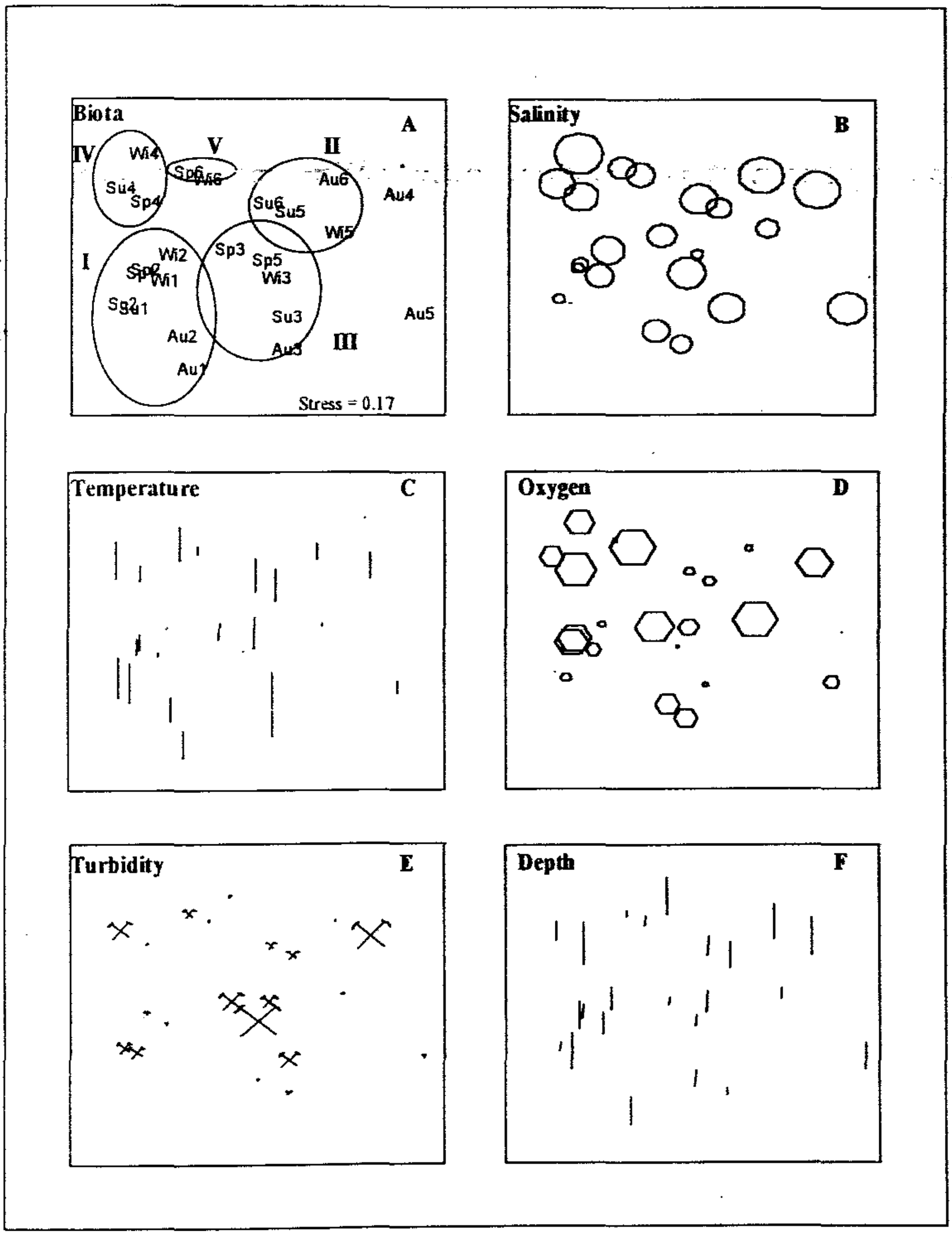

Figure 5.1. MDS plots of environmental data superimposed on the benthic data at Sites 1-6 from the Mlalazi estuary during the 1989-1991 sampling period. The biotic community showing groupings in the data (**A**). Superimposed environmental data (**B**)-(**F**) with symbols representing salinity, temperature, oxygen, turbidity and depth.

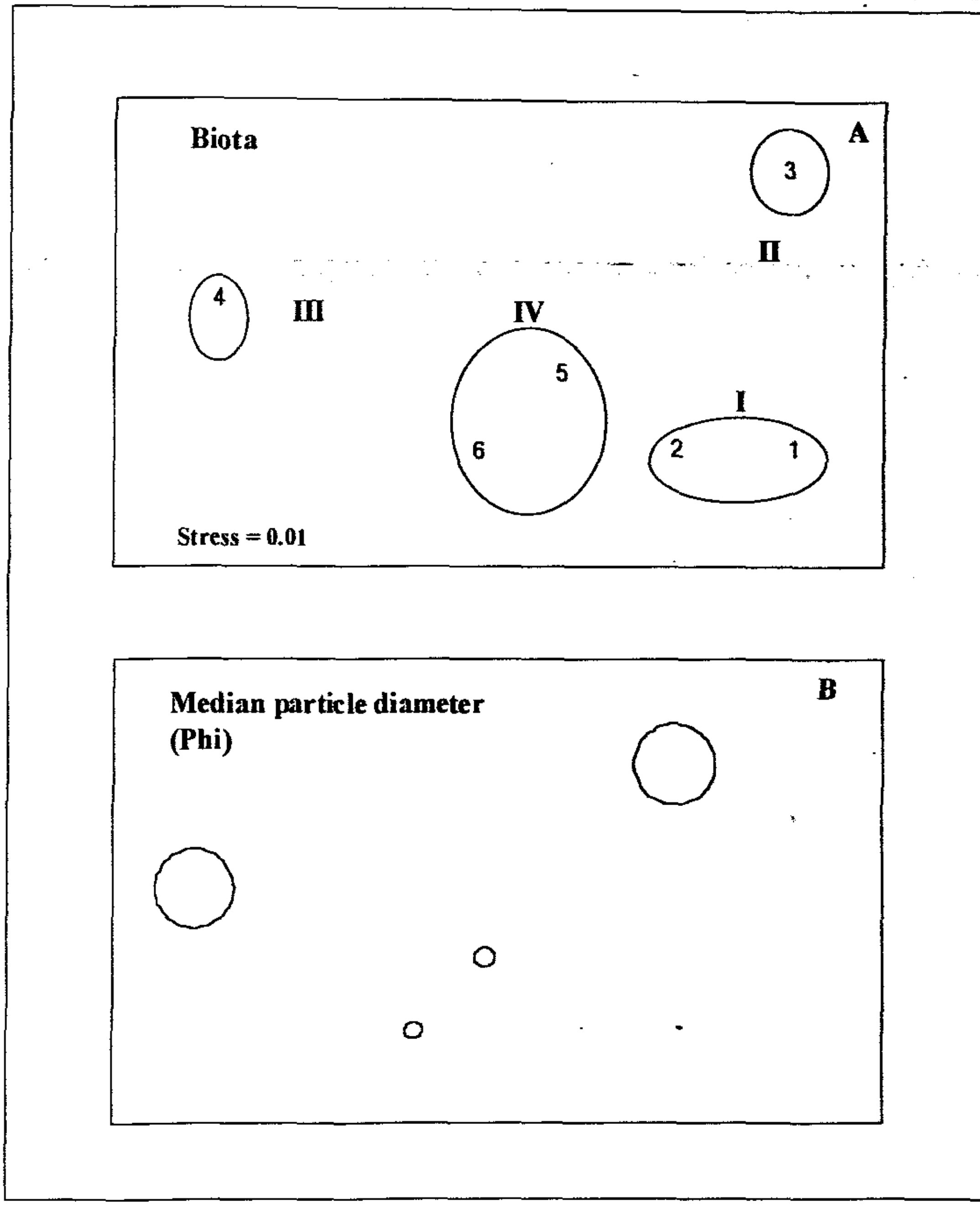

Figure 5.2. MDS plot of environmental data superimposed on benthic data at Sites 1-6 from the Mlalazi estuary during the 1989-1991 sampling period. The biotic community showing groupings in data (**A**). Superimposed environmental data (**B**) with symbols representing the median particle diameter.

The middle reaches of the Mlalazi estuary was expected to show higher turbidity values than the other reaches due to the muddy substratum. During the 1989-1991 period, turbidity values were high at Sites 3 and 4 (middle reaches) and this is evident in the turbidity superimposed plot where groups III and IV are clearly distinguished from groups I, II, and V. Groups III and IV consisted of Sites 3 and 4, which are sites that were high in turbidity compared to the rest of the sites. In the sediment particle size plot, groups I and IV were separate from groups II and III during the 1989-1991 period. Group I consisted of Sites 1 and 2 whereas group IV consisted of Sites 5 and 6. These sites (Sites 1, 2, 5, and 6) were characterised by medium sand whereas groups II and III constituted silty sediments.

Temperature, oxygen and depth did not seem to contribute much to the biotic patterns due to the fact that these variables showed no marked changes along the estuary. Oxygen concentration, did however, increase from the upper reaches to the middle reaches and then decrease again in the lower reaches.

5.3.2. <u>Physico-chemical variables during the 1999-2000 period</u>

Figures 5.3 to 5.6 show the superimposition of the physico-chemical variables measured during the 1999-2000 sampling period on the biotic data. During the 1999-2000 sampling period, the physical variables that seemed most responsible for structuring the biotic community were salinity, mean particle diameter and sediment organic content.

In terms of salinity, groups II and III were distinguishable from groups I and IV. Groups II and III are composed of Sites 1, 2, 3 and 4 that are in the upper and middle reaches of the estuary. Salinity in a typical estuary increases from the upper reaches towards the mouth. This pattern was observed in the salinity plot with salinity increasing from Sites 1, 2 and 3 (upper and middle reaches) towards Sites 5, 6 and 7 (lower and mouth reaches). In terms of the median particle size, there seemed to be an increase from the upper reaches towards the middle reaches, decreasing in the lower reaches and increasing again in the mouth reaches. Groups I, II and IV were separated from group III. Group II consisted of Sites 1, 2, and 3 while groups I and IV consisted of Sites 5 and 6. These sites (Sites 1, 2, 3, 5, and 6) were characterised by coarser sediments compared to Sites 4 and 7 (Groups III and part IV), which were characterised by fine sands. High organic content in estuaries is usually measured in fine sediments compared to medium or coarse grained sediments. This was apparent

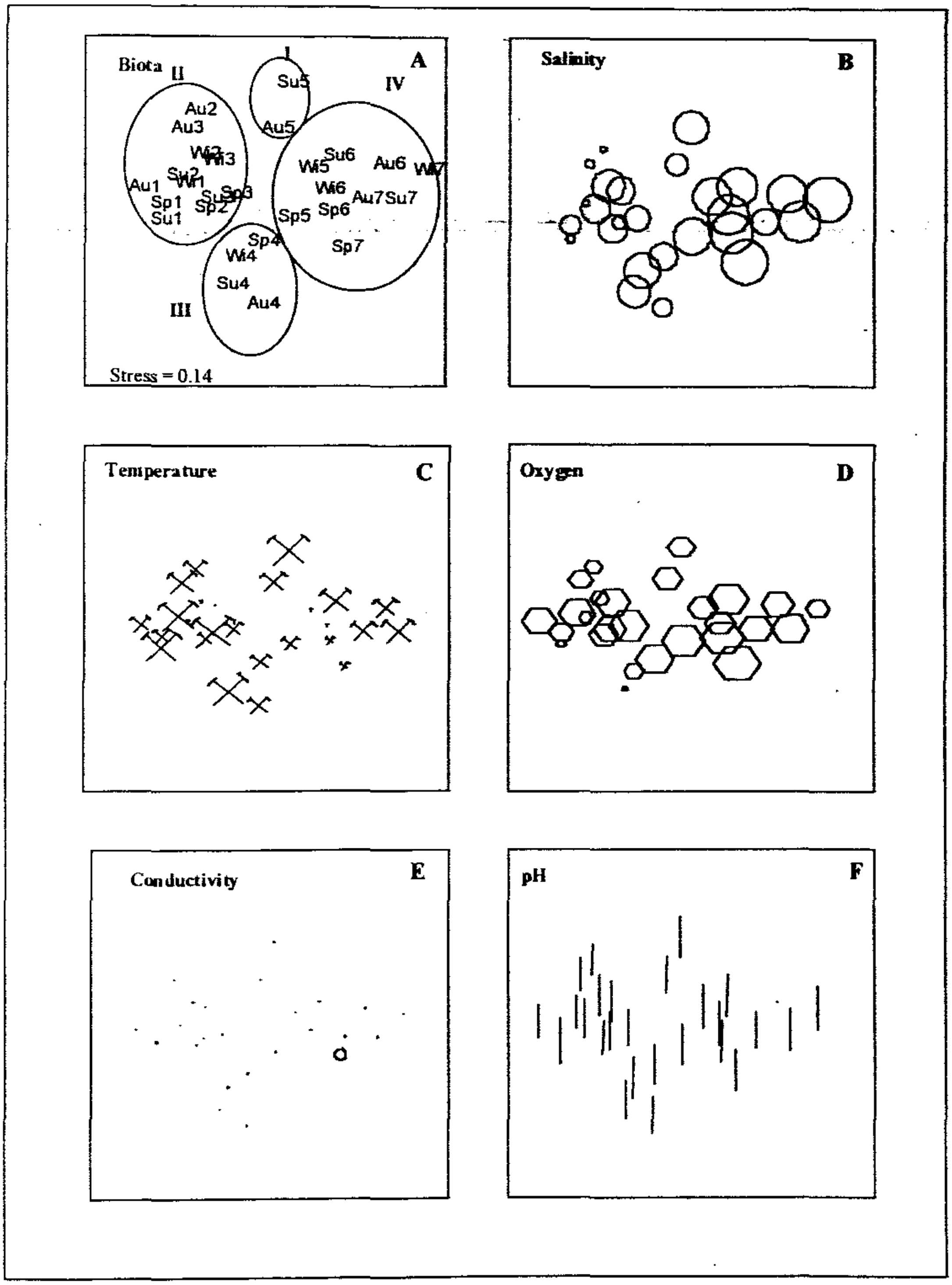

Figure 5.3. MDS plot of environmental data superimposed on benthic data at Sites 1-7 from the Mlalazi estuary during 1999-2000. The biotic community showing groupings in data (**A**). Superimposed environmental data (**B**)-(**F**) with symbols representing salinity, temperature, oxygen, conductivity and pH.

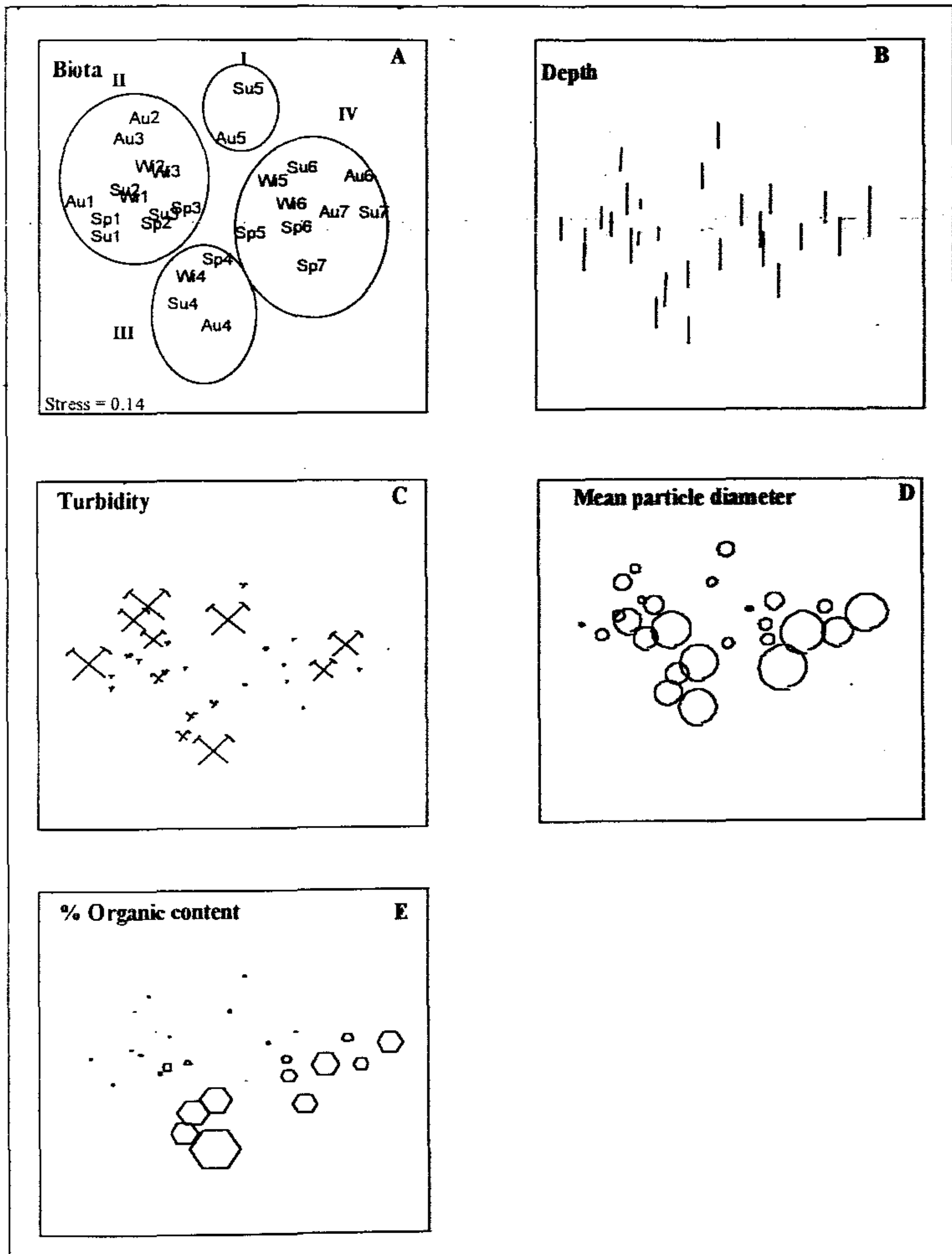

Figure 5.4. MDS plot of environmental data superimposed on benthic data at Sites 1-7 from the Mlalazi estuary during 1999-2000. The biotic community in groupings in data (**A**). Superimposed environmental data (**B**)-(**E**) with symbols representing depth, turbidity, mean particle diameter and percentage organic content.

in the organic content plot where group III was distinguishable from the rest of the groups. Group III consisted of Site 4, which was characterised by fine sands with high organic content than the other sites.

Temperature, pH, oxygen, turbidity and depth did not contribute much towards the observed biotic patterns. However, relating any single environmental variable to the observed biotic pattern does not provide a very successful match. This is because the community structure found in an estuary usually result from a combination of environmental variables.

Ammonium, nitrate, nitrite and phosphates were responsible for distinguishing group II from groups I, III, and IV. Group II, which consisted of samples from Sites 1, 2 and 3, had higher concentrations of these nutrients with the concentrations decreasing in the lower and mouth reaches (differences in symbol sizes in the graphs denotes low or high levels of nutrient concentration). Concentrations of these nutrients were notably higher at Site 3, which is closest to the prawn farm outlet. Group IV, in the sulphate superimposed plot, was different from groups I, II, and III. This groups (IV), containing samples from the lower reaches, were characterised by high concentrations of sulphate which decreased towards the middle and upper regions. The chlorophyll-a levels distinguished group II, from group IV. Group II contained sites from the upper and middle reaches which had high chlorophyll-a levels compared to sites in group IV. Site 3 had the highest chlorophyll-a concentration than the other sites.

5.3.4. **BIO-ENV procedure**

Tables 5.1, 5.2 and 5.3 present the BIO-ENV results of the 1989-1991 and 1999-2000 sampling periods where the environmental variables combinations were considered at increasing levels of complexity (k = 1, 2, 3,.... x). The premise here is that a combination of environmental variables that best explains the biotic patterns is being sought.

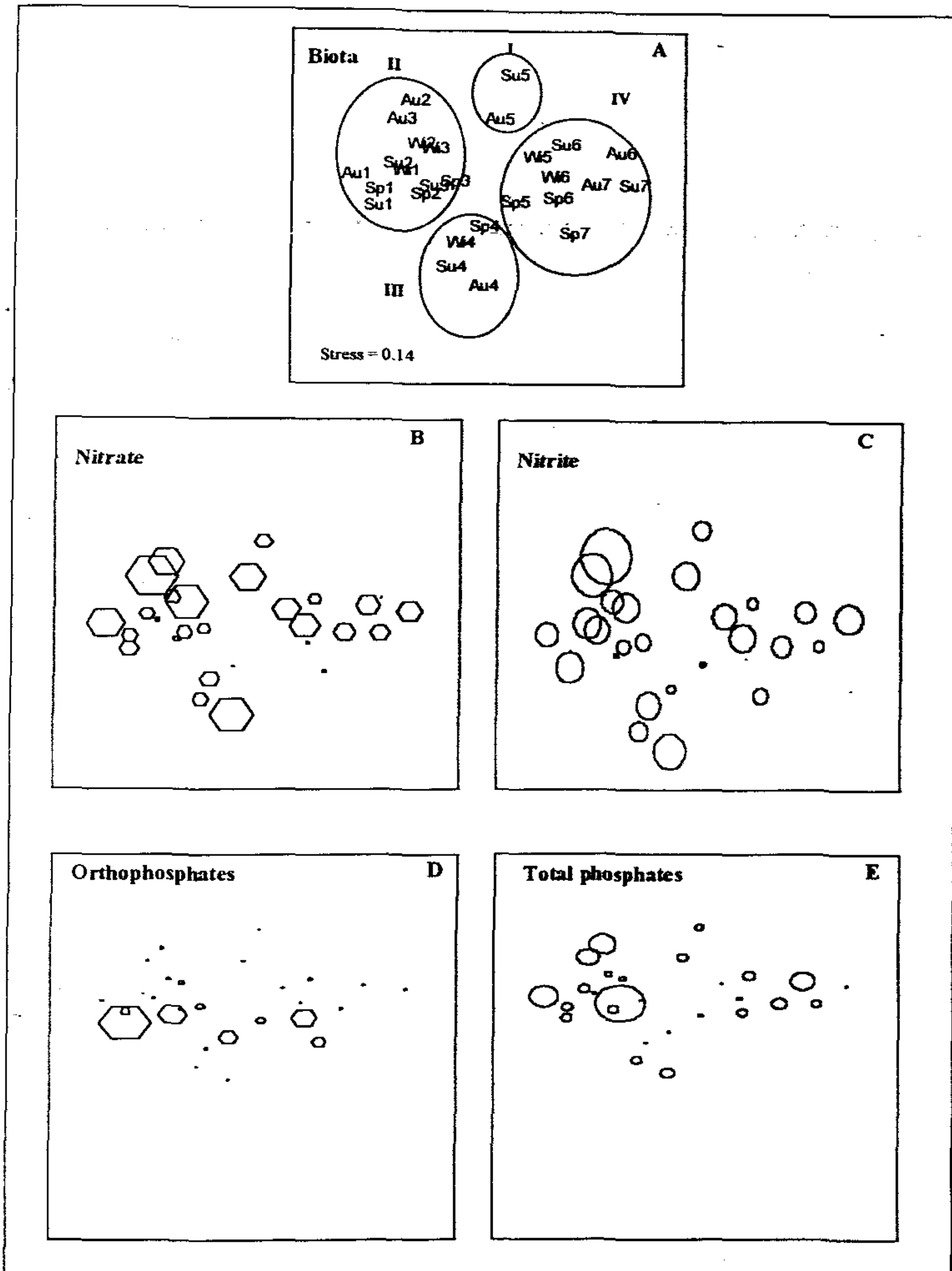

Figure 5.5 MDS plot of environmental data superimposed on benthic data at Sites 1-7 from the Mlalazi estuary during 1999-2000 sampling period. The biotic community groupings in data (**A**). Superimposed environmental data (**B**)-(**E**) with symbols representing nitrate, nitrite, orthophosphates, total phosphates

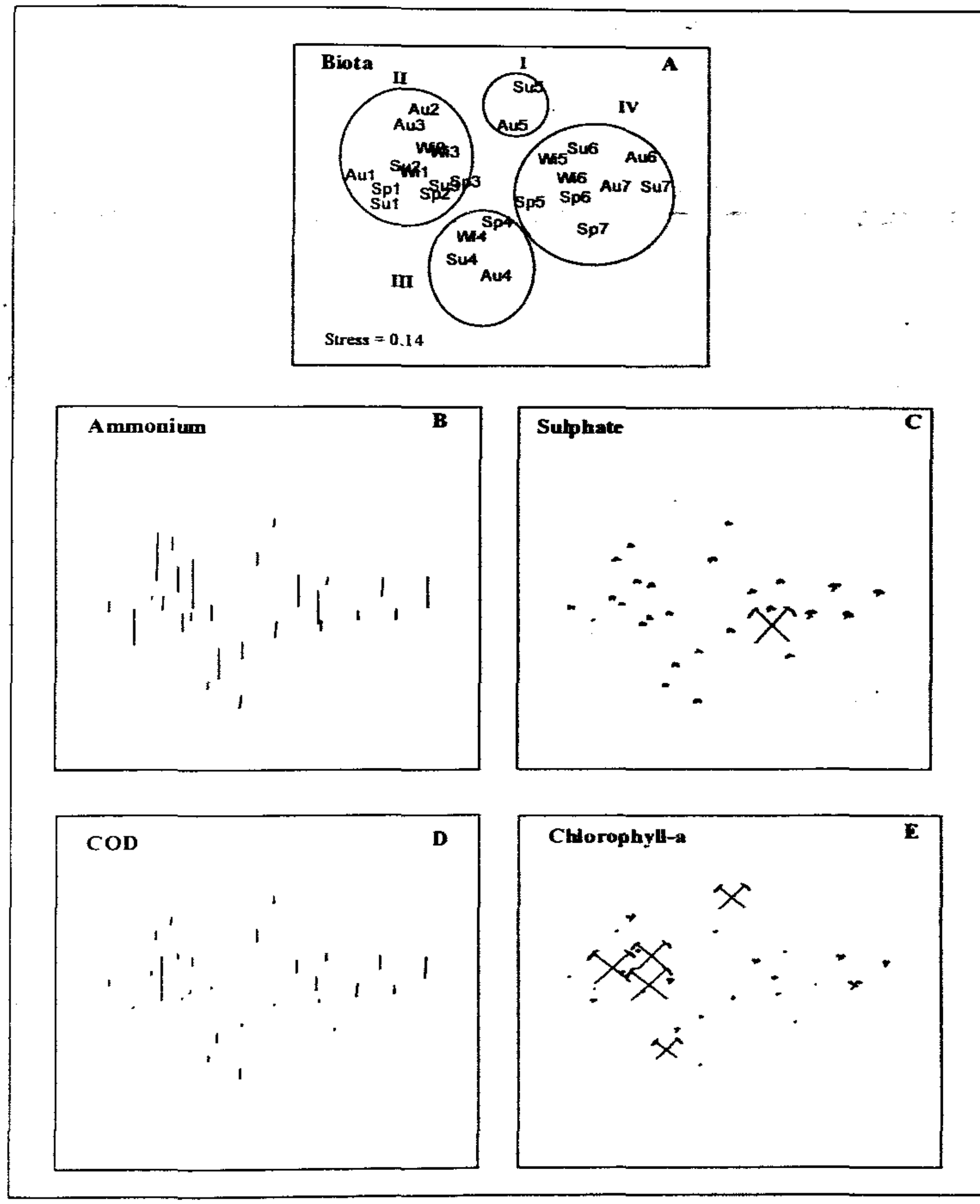

Figure 5.6. MDS plot of environmental data superimposed on benthic data at Sites 1-7 from the Mlalazi estuary during 1999-2000 sampling period. The biotic community groupings in data **(A)**. Superimposed environmental data **(B)**-**(E)** with symbols representing ammonium, sulphate, COD and chlorophyll-a.

The single physical variable which best explained the changes in the benthic community during the 1989-1991 period, was median particle size (Phi) (p_w = 0.505), followed by turbidity (p_w = 0.459). The combination of physical variables which best explained the changes in benthic community (**p_w = 0.874**) included salinity, oxygen, turbidity, depth and median particle diameter.

During 1999-2000, salinity (p_w = 0.292) was the physical variable best explaining the changes in the benthic community, followed by pH (p_w = 0.206) and percentage organic content (p_w = 0.194). Changes in the benthic community were, however, best explained by a two variable combination, i.e. salinity and percentage organic content (**p_w = 0.404**). The best 3 variable combination contained salinity, pH and percentage organic content (p_w = 0.396).

Chemical variables measured during the 1999-2000 sampling period were also subjected to the BIO-ENV procedure (Table 5.3). The single chemical variable which best explained the community structure was sulphate, with a p_w of **0.364**. This was followed by COD (p_w = 0.155) and nitrate (p_w = 0.020). The best two variable combination included sulphate and COD (p_w = 0.204).

Table 5.1. Combination of 6 physical variables, taken during the 1989-1991 sampling period, yielding the best matches of the biotic and abiotic similarity matrices for each k, as measured by weighted Spearman rank correlation p_w. Bold type indicates overall optimum.

k	Best variable combination (p_w)		
1	Phi (0.505)	Tb (0.459)	D (0.343
2	D, Phi (0.766)	Sal, Phi (0.735)	Sal, Tb (0.605)
3	Sal, Tb, D (0.842)	Sal, D, Phi (0.830)	Sal, Tb, Phi (0.695)
4	Sal, Tb, D, Phi (0.852)	Sal, O_2, D, Phi (0.794)	
5	**Sal, O_2, Tb, D, Phi** **(0.874)**		
6.	Sal, Temp, O_2, Tb, D, Phi (0.567)		

Table 5.2. Combination of the 9 physical variables, taken during the 1999-2000 sampling period, yielding the best matches of biotic and abiotic similarity matrices for each k, as measured by weighted Spearman rank correlation p_w. Bold type indicates overall optimum.

k	Best variable combination (p_w)		
1	Sal (0.292)	pH (0.206)	%OC (0.194)
2	**Sal, %OC (0.404)**	pH, %OC (0.336)	Sal, O_2 (0.300)
3	Sal, pH, %OC (0.396)	Sal, O_2, %OC, (0.380)	Sal, Cond, %OC (0.374)
4	Sal, pH, O_2, %)C (0.391)	Sal, O_2, D, %OC (0.372)	
5	Sal, pH, O_2, D, %OC (0.386)	Sal, O_2, Tb, D, %OC (0.367)	
6	Sal, pH, O_2, Tb, D, %OC (0.372)	Sal, pH, Cond, O_2, D, %OC (0.349)	
7	Sal, Tem, pH, O_2, Tb, D, %OC (0.346)		
8	Sal, Tem, pH, O_2, Tb, D, Phi, %OC (0.343)		
9	Sal, Tem, pH, Cond, O_2, Tb, D, Phi, %OC (0.314)		

Sal = salinity

%OC = percentage organic content

Tem = temperature

Cond = conductivity

O_2 = oxygen

Tb = turbidity

D = depth

Phi = median particle diameter

Table 5.3. Combination of 8 chemical variables, taken during the 1999-2000 sampling period, yielding the best matches of biotic and abiotic similarity matrices for each k, as measured by weighted Spearman rank correlation p_w. Bold type indicates overall optimum.

k	Best variable combination (p_w)		
1	**SO_4^{-2} (0.364)**	COD (0.155)	NO_3 (0.020)
2	SO_4^{-2}, COD (0.204)	NO_3, COD (0.130)	NO_2, COD (0.100)
3	NO_3, SO_4^{-2}, COD (0.123)	NO_2, SO_4^{-2}, COD (0.098)	SO_4^{-2}, TP, COD (0.093)
4	NO_3, NO_2, SO_4^{-2} (0.052)	NO_3, SO_4^{-2}, TP, COD (0.044)	
5	NO_3, NO_2, TP, COD, Chlr-a (0.006)	NO_3, NO_2, SO_4^{-2}, OP, COD (0.001)	
6	NO3, NO3, OP, TP, COD, Chlr- (-0.005)		
7	NH_3, NO_3, NO_2, OP, TP, COD, Chlr-a (-0.027)		
8	NH_3, NO_3, NO_2, SO_4^{-2}, OP, TP, COD, Chlr-a (-0.059)		

NH_3 = ammonium

NO_3 = nitrate

NO_2 = nitrite

SO_4^{-2} = sulphate

OP = ortho-phosphate

TP = total phosphate

COD = chemical oxygen demand

Chlr-a = chlorophyll -a

5.4. Discussion

In an estuary there are spatial and temporal variations in physical and chemical conditions which often range from freshwater to marine conditions. Biotic species thrive optimally in water with particular combinations of physical and chemical attributes (Ketchum 1983). Since optimal conditions differ for each species, the biota is spatially zoned along this gradient.

The superimposition of physico-chemical variables on the biotic data clearly indicated the existence of a significant relationship between the biological and environmental variables. This augmented the fact that variations of physico-chemical conditions of an estuary play a major role in structuring the biotic communities (Ketchum 1983).

5.4.1. Physical variables

The most important environmental variables considered to influence the distribution and abundance of benthic organisms in estuaries are salinity and sediment characteristics (grain size and organic content) (Warwick, *et. al* 1991). The superimposition and BIO-ENV results of the physical variables show that the biotic patterns of the 1989-1991 and 1999-2000 sampling periods were best governed by salinity, median particle size and organic content. The benthic biota in the Mlalazi estuary seemed to be zoned along the salinity and median particle diameter gradient. However one cannot expect a single environmental variable to be solely responsible for determining the distribution of benthic community. This is because reactions of organisms to one environmental variable may be confounded by other prevailing environmental variables. For example, Moore (1966, in Knox 1986) found that the reaction of estuarine mud inhabitants to salinity is complicated by the effect of varying nature of the substratum and their ability to retreat into burrows and so avoid adverse salinities and other unfavourable conditions. This seemed to be the case during the 1989-1991 period where Site 4 in the middle reaches had reduced densities compared to Site 3. Both these sites were characterised by the same sediment type however Site 4 had higher salinity values compared to Site 3. The combination of a muddy substrate and high salinity appeared to have caused a decrease in organism density as compared to Site 3.

Many authors have reported changes in benthic community abundance in response to variable nature of salinity in estuaries (McLachlan & Grindley 1974, Cyrus 1988). In a typical estuary, such as the Mlalazi estuary where salinity increases from the head towards the mouth, those organisms that tolerate low salinities are usually confined to the head and upper reaches while those that tolerate higher salinities (stenohaline) are found in the lower and mouth regions. However, euryhaline species, will be expected to inhabit areas with a wide range of salinities along the length of the estuary. For example, during the 1989-1991 and the 1999-2000 sampling periods, the amphipods *C. triaenonyx* and *G. lignorum* were restricted to the head and upper reaches (low salinity) whereas the bivalves *D. hepatica* and *H. ludwigii*, the gastropods *A. capensis* and *Nassarius sp.* were found at high salinity regions (lower and mouth reaches). The opportunistic polychaetes *Prionospio. spp.* and the capitellids were found from the head to the mouth of the estuary. Wu and Richards (1981) reported that changes in the structure of the benthic communities in a subtropical estuary showed a good correlation with the prevailing gradient of increasing salinity. Owen and Forbes (1997) reported changes in community structure of the St. Lucia Narrows between 1989 and 1994 due to different salinity conditions. The bivalve *Solen cylindraceus* was present during the high salinity period with increased abundance being associated with marine salinities while being rare in low salinities.

Benthic surveys have indicated that sediment type plays a substantial role in forging faunal diversities and abundance (Gray 1981, Flint, *et. al* 1986). In the Mhlathuze estuary the sediment load deposited into the upper reaches and fine dredge spoil sediments in the mouth of the estuary played a major role in changing faunal diversities, abundance and distribution (Mackay & Cyrus 1999, Wepener & Vermeulen 1999). The nature of the substratum was an important determining factor in the distribution and abundance of the crab, *P. blephariskios* in the St. Lucia and the Mhlathuze estuaries (Owen, *et. al* 2000). This crab was only recorded from the muddy sites in both estuaries. In the Mlalazi estuary, during the 1989-1991 and 1999-2000, *P. blephariskios* was also recorded only in muddy areas of the estuary while it was virtually absent in sandy areas.

Coarse sediments usually contain little organic matter with the fauna being somewhat impoverished. Fine muddy sediments, in contrast, are usually characterized by a high organic content with a corresponding increase in organism diversity and abundance due to food availability. Blaber, *et. al* (1983) found the amphipod *G. lignorum*, usually more common in sandy substrata, to be more common on mud than sand in the St. Lucia estuary due to greater availability of a detritus food source. During the two sampling periods in the Mlalazi estuary, a higher abundance of organisms were found at sites characterized by silt and fine sediments (Sites 3 and 4) when compared to sites that constituted coarse-grained sediments (Sites 1, 2, 5 and 6). It was concluded that increased benthic densities at these sites were due to the nature of the substrata and high organic content.

It has been argued that increased inputs of organic matter should result in an increase in benthic diversity (Hockin 1983). However, there have been studies that showed that enrichment of ecosystems sometimes leads to disappearance rather than enhancement of species (Riebesell 1974, Schratzberger & Warwick 1998). The increased input of organic matter in fact allow fewer species to coexist since high levels of organic enrichment often alter environmental conditions to such an extent that only tolerant species are able to survive. Even though areas in the Mlalazi estuary with a high organic content were densely populated, these areas were inhabited predominantly by a few relatively tolerant taxa such as the opportunistic capitellids and *Prionospio* polychaetes.

Reduction in light penetration due to increased water turbidity has been known to act directly and indirectly on the fauna of estuaries (Cyrus 1988). In areas of high turbidity reduction in species diversities have been reported (Cyrus & Blaber 1987). The middle reaches of the Mlalazi estuary, even though densely populated during the 1989-1991 sampling period, showed a lower biotic diversity when compared with rest of the estuarine reaches. The nature of the substrate, high organic content and high turbidity seemed to have created conditions that led to colonization of the middle reaches by highly abundant opportunistic species. The opportunistic *Prionospio spp.* and capitellids, which dominated the middle reaches, are well documented as taxa that are indicative of disturbed or organically enriched environments (Day 1967a, Brown & McLachlan 1990, Mackay & Cyrus 1999). The increasing densities of opportunistic species usually lead to elimination of other organisms due to lack of available or favorable habitat. This results in low diversity. In most benthic studies undertaken, it has been shown that increased turbidities due to factors such as landuse practices and sedimentation have deleterious effects on estuarine fauna (Johnson 1981, Hay 1985, Mackay & Cyrus 1999). Decreases in numbers and diversity of benthic communities have been reported to be caused by continuously high turbidities

(Dallas, *et. al* 1994). The absence of the suspension feeding bivalves and all other suspension feeding invertebrates from the Severn estuary, Britain, was almost entirely due to the high turbidity (Warwick, *et. al* 1991).

Temperature and depth did not appear to have much influence on changes in the biotic pattern during the 1989-1999 period, while the same can be said of temperature, depth, turbidity and pH during the 1999-2000 period. The Mlalazi estuary is relatively shallow and even though the sites differed in depth and temperature along its length, there were no marked differences. The pH values during the 1999-2000 period were also not substantially different between sites with the pH-superimposed plot showing no clear pattern or trend. What is interesting is that the BIO-ENV procedure placed it as the single variable second to salinity in structuring the biotic patterns. The reason for this is unclear since pH was 'normal' for estuarine waters, spatially and temporally.

5.4.2. **Chemical variables**

Man's activity (e.g. sewage and effluent disposal) has resulted in a high input of nutrients into many coastal areas, raising the nutrient load compared to natural levels (Schratzberger & Warwick 1998). Elevated concentrations of nutrients have been reported to cause eutrophication problems with high environmental stress in estuaries (Rosenberg, *et. al* 1996). Such conditions may lead to deleterious effects on benthic organisms.

The nutrients, nitrate, nitrite, phosphates, sulphates and chlorophyll-a were all partially responsible for the observed biotic patterns during 1999-2000. The middle reaches (Site 3) showed high concentrations of nitrate, nitrite, phosphates and chlorophyll-a compared to the rest of the estuary. Although much of the nutrient load in estuaries is brought in by river discharge and runoffs it was unlikely that such events were responsible for the high nutrient concentrations measured in the middle reaches of the Mlalazi estuary during the 1999-2000 sampling period. This is because low nutrient levels were recorded in the upper and lower reaches of the estuary with very high nutrient concentration being recorded in the middle reaches (Site 3) and in the prawn farm effluent. Elevated nutrient concentrations usually lead to a decrease in diversity of benthic fauna (Birch 1982, Baker & Horton 1990). The benthic fauna of the Friefjord showed large-scale reductions due to a high nutrient load (Rosenberg *et. al* 1996). Site 3 in the Mlalazi estuary, where high concentrations of nutrients were recorded, was expected to be characterized by low faunal diversity compared to the other sites. However, this was not the case, which meant that other parameters, and not the prawn farm effluent, played a role in faunal abundance and distribution in the Mlalazi estuary.

What is interesting is that sulphate was the single chemical variable that best explained the biotic pattern during the 1999-2000 period. Even though sulphate concentrations were higher in the prawn farm effluent, Site 3, which is closest to the prawn farm outlet, did not exhibit high sulphate concentrations. The general picture was an increase in sulphate concentration from the mouth to the upper reaches which suggested the sea as the source of sulphate. Day, *et. al* (1989) stated that sulphur in estuarine waters is derived predominantly from the sea, thus supporting the above statement. However, it is not clear to what extent sulphate was responsible for structuring the Mlalazi biotic community. High sulphate concentrations are often associated with anoxic conditions and odour problems in estuaries, which are detrimental to aquatic organisms (Sawyer & McCarty 1989). High sulphate concentrations coupled with changes in pH can also have devastating effects on aquatic biota (Sawyer & McCarty 1989). The pH changes were minor throughout the study period while anoxic and odour problems were not encountered in the Mlalazi estuary. This leads to suggest that even though sulphate was chosen as being responsible for biotic patterns its concentrations were not as high as to affect the distribution of biota.

COD was placed second as the single variable best explaining biotic patterns. This confirmed the presence of oxygen consuming chemicals in the estuary. From the superimposed plots it is observed that COD is high in areas where nutrient concentration is high (Site 3) indicating that these nutrients decompose by oxygen consuming processes which may lead to depletion of oxygen. High chlorophyll concentrations at Site 3 are signs of increased nutrient load discharged into the Mlalazi estuary from the prawn farm. Trott and Alongi (2000) reported a significant increase in chlorophyll-a concentrations in the Muddy Creek estuary during effluent discharge from the shrimp farm which exceeded levels observed in the estuary prior to pond discharge.

Nitrogen and phosphorus are plant nutrients and when present in excessive amounts they can set off an explosive growth or bloom of plants such as green and blue-green algae. The excess plants die and sink to the bottom of the system where they are decomposed by oxygen-consuming bacteria leading to depletion of oxygen supply (Dallas, *et. al* 1994). Such events lead to deterioration of the aquatic system and if the nutrients continue to flood in, the water may become foul and almost devoid of organisms. The Tongati estuary, regarded as degraded by Blaber, *et. al* (1984) experienced eutrophication problems due to sewage effluent entering the estuary. This estuary was reported to be rich in phosphates and nitrates which promoted rapid growth of large quantities of water hyacinth (*Eichornia crassipes*) leading to low oxygen levels. Such conditions led to impoverished zoobenthos of the Tongati estuary. The continuous discharge of nutrients from the prawn farm has the potential to create conditions similar to that experienced in the Tongati estuary if the nutrient load coming from the prawn farm is not properly monitored and managed.

Aquaculture developments should ideally be placed adjacent to a large water source such as the sea or an estuary; since water is required not only to support the organisms but also to provide oxygen and removal of effluents (Beveridge 1996). The effluents from the aquaculture operations are then discharged into these aquatic bodies. Due to the operational nature of aquacultures it is estimated that as much as 60 % of total phosphates and 80 % of total nitrogen wastes end up in these receiving aquatic systems (Wallin & Hakanson 1991). This is accompanied by apparent changes in dissolved oxygen, chemical oxygen demand, turbidity and associated eutrophication problems (Beveridge 1996). Wallin and Hakanson (1991) studied the impacts of Swedish and Finnish coastal fish farming and found strong correlations between fish farm effluents and nutrient levels in surface waters. Studies conducted in the Baltic, a highly brackish area, showed enhanced growth and production of algae and changes in community structure due to fish farming wastes being discharged into this area (Beveridge 1996). Changes in the macrobenthic structure were positively correlated with waste discharge and accumulation, COD increases and sediments becoming increasingly anaerobic. At the area of organic enrichment, exceptionally high numbers of opportunistic species (*Capitella capitata*) were found and biodiversity in this area was characteristically low Galope-Bacaltos and Diego-McGlone (2002) reported a decrease in species diversity and presence of opportunistic species in areas affected by fishpond effluents in the Sta. Rita River estuary.

The above discussion clearly indicates that effluents from aquaculture, such as the prawn farm, lead to the input of nutrients into the estuarine environment, which can potentially create an unbalanced ecosystem and lead to large scale changes in biotic communities if not properly controlled. Farmers have to consider the effects of the farm discharge on the environment. In the long-term perspective, farms cannot indiscriminately discharge nutrient rich waters without causing changes to the very water medium necessary for the operation. The solutions to the problem may be based on minimizing the nutrient loading and employing appropriate feeding strategies (Vigneswaran, Ngo & Wee 1999). The environmental management of anthropogenic inputs to brackish-marine environments also requires knowledge of the effects of different intensities and frequencies of input in relation to the nature of the aquatic body and receiving assemblages of organisms. Availability of scientific information can help to minimize the effects, for example by selection of the most appropriate disposal sites and by controlling the frequency and intensity of the input of effluents (Schratzberger & Warwick 1998). Sound scientific information on different potential adverse impacts of aquacultures on the aquatic environment remains scarce, which indicate the need for more investigations. The present study thus serves as such a baseline study to provide information on the biota of the Mlalazi estuary and the extent to which effluent discharge from the prawn farm affects the water quality of the estuary and ultimately biotic diversity.

5.5. Conclusion

Results from the present study indicated that variations in and combination of physical and chemical factors had a major influence in structuring the benthic community of the 1989-1991 and 1999-2000 sampling periods. This study also provided insight into the nutrient load entering the estuary from the prawn farm. The findings are that the farm is discharging elevated concentrations of nutrients that have a potential of promoting rapid growth of algae (eutrophication) leading to poor water quality, which can impact negatively on the biotic community of the estuary. These findings have management implications for the Mlalazi estuary. In addition to the monitoring of the prawn farm effluent, regular routine monitoring of the benthic community and water quality of the estuary is imperative and would assist managers in detecting future changes in the system related to chemical and biological pollution as well as natural perturbations.

CHAPTER 6

6.1. General Discussion and Conclusion

Benthic macroinvertebrates have often been used to assess the general status and health of aquatic ecosystems (Rosenberg & Resh 1993). This is because benthic organisms are relatively long lived, allowing monitoring of spatial and temporal changes due to natural or human induced disturbance. Benthic macroinvertebrates are also relatively non-mobile, thus they can be regarded as representative of the location where they are found. Benthic organisms are also directly or indirectly involved in most physico-chemical processes ongoing in their environment (Mackay & Cyrus 1999). Biological surveys of benthic communities result in complex biotic and abiotic data from which patterns and relationships need to be extracted. This chapter aims at briefly summarizing the spatial and temporal changes in the composition and abundance of the macrobenthic community of the 1989-1991 and 1999-2000 sampling periods (Chapter 4) and the factors that caused the observed biotic patterns (Chapter 5). The chapter will also highlight management issues and implications arising from this study.

Physical analysis of the water during the two sampling periods indicated conditions that generally occur in a typical estuary. In contrast, the chemical water conditions during the two sampling periods revealed human influences on the estuarine environment. Effluents from industries and agricultural runoff are widely reported to cause changes in the estuarine environment which lead to poor water quality resulting in negative effects on aquatic biota (Scharler, *et. al* 1997). Such anthropogenic sources often lead to eutrophication problems caused by high levels of nutrients brought into the estuary. Agricultural runoff is the probable source of the high nutrient concentrations recorded in the estuary during the 1989-1991 sampling period. During this period, the prawn farm was not yet in operation and there were no major industries or rural settlements in the vicinity of the estuary. During the 1999-2000 sampling period, the high concentration of nutrients measured in the prawn farm effluent and at Site 3, which is adjacent to the prawn farm outlet, suggested the prawn farm as the source of high nutrient input into the estuary.

The benthic community structure of the two sampling periods was largely dominated by *Prionospio* and capitellid polychaetes, which are widely regarded as opportunistic or early colonizing species (Grassle & Grassle 1974, Harvey, *et. al* 1998). These two taxa are also regarded as indicators of disturbed or organically enriched areas. The presence of these highly abundant polychaete taxa in the Mlalazi estuary suggested that the biotic community was subjected to stressful conditions during both sampling periods.

Multivariate analysis of the benthic community provided information about spatial and temporal changes in the community structure of the two sampling periods. The influence of physico-chemical conditions was apparent in the biotic patterns whereby the benthos was distributed along the gradient of the certain physical and chemical parameters. The analysis revealed that the benthic community structure was forged predominantly by spatial variations in habitat conditions, rather than by temporal changes. As such, the variable most responsible for the observed biotic patterns during the 1989-1991 period was median particle size. During the 1999-2000 period, salinity appeared to be most responsible for structuring the biota. It was, however, found that a combination of a number of physico-chemical factors was important in structuring the benthic community. This confirmed a direct relationship between the benthic organisms and the physico-chemical conditions prevailing at that time.

Stresses that can cause changes in the estuarine system may result from natural hazards such as floods and droughts (Day & Grindley 1981). Stresses also come from human activities such as industrialisation, urbanisation, agriculture and other activities and these tend to be more deleterious than natural events (Day 1981). This study has revealed that although the variability in physico-chemical estuarine conditions was largely responsible for forging the distribution and abundance of benthic organisms, human induced effects also played a role in this regard. Even though the impact of the prawn farm effluent on the benthic fauna was not as apparent as the poor water quality of the effluent would have suggested, results indicated the prawn farm as a major anthropogenic source of nutrients.

6.2. Estuarine degradation and restoration

The degradation of South African estuaries is a cause for concern and increasing population growth and development activities mean that the threat to estuarine ecosystems is likely to increase (Wiseman & Sowman 1991). Recognition of the ecological importance of estuaries is crucial if we want ensure that estuarine functions and characteristics are restored and maintained. The features that determine the status of the estuarine system include water quality, biotic diversity and sedimentation. Adverse changes to one or more of these features can results in symptoms of degradation such as reduced biotic diversity.

The efforts to restore estuaries have largely focused on reducing pollution loads, re-establishment of wetlands, reducing nutrient loading and eutrophication (Allen & Hull 1987, Collett & Leatherland 1985, Goldsmith & Hildyard 1988). The rehabilitation of the Siyaya and Sipingo estuaries involved re-establishment of riparian vegetation and restoration of wetland systems (Wiseman & Sowman 1991, Mackay 1996). Restoration of Sezela estuary included efforts to recycle industrial wastewater and to adopt improved effluent treatment methods (Wiseman & Sowman 1991). However these are not simple processes because a number of scientific, technological, economic and political factors are involved. Despite these complexities, implementations of restoration strategies in estuaries are sure to enhance the health status of the estuary.

6.3. Estuarine Management and Implications

KwaZulu-Natal contains a large number of estuaries, most of which are small temporarily open/closed systems such as the Mlalazi estuary. Estuaries generally are sensitive environments that support diverse animal and plant communities (Begg 1978). These systems also serve as breeding grounds for invertebrates, fishes and birds and as such are important as nursery grounds. Recognition of the value and importance of estuarine systems has resulted in efforts to conserve these systems. Ongoing estuarine degradation and the mounting demand on these ecosystems has led to establishment of management actions along the coast of KwaZulu-Natal. The establishment of marine protected areas (MPAs) in KwaZulu-Natal is one such management action. However, out of the 74 estuaries in KwaZulu-Natal, only ten fall within official protected areas. This is totally inadequate because many more estuaries on the KwaZulu-Natal coast need conservation protection (Mann, *et. al* 1996). The Mlalazi Nature Reserve, under which the Mlalazi estuary is situated, was proclaimed a MPA in KwaZulu-Natal under the auspices of KwaZulu-Natal Wildlife (formerly the Natal Parks Board). The estuary has a flourishing mangrove community (Mann, *et. al* 1996). It is also a popular tourist venue with camping, fishing and boating as the primary activities.

In order to implement a management strategy it is important to identify possible harmful effects, their source, and to establish whether the effects are detrimental to the estuary in question. Once it is established that the estuary has been adversely impacted, steps must be taken to minimize or control the harmful effects. Three main approaches on how to achieve this have been identified (Walmsley, *et. al* 1999). They are:

- Source-based management i.e. controlling waste at the source to minimize pollution,
- Impact management i.e. managing the impact on the water environment, and
- Remedial action i.e. taking remedial action where unacceptable quality exist in the water environment

This study has revealed that the prawn farm effluent is probably the primary anthropogenic cause of the poor water quality in the estuary. If not controlled, this will potentially lead to eutrophication problems and a concomitant decline in estuarine function. In view of the effect prawn farm effluent will have on the future water quality and biotic diversity of the estuary, it is imperative that a proper management strategy involving all stakeholders, be developed and implemented. Based on the information obtained during this study, the following recommendations with regard to a biomonitoring program and a management strategy are made:

- Monitoring of the water quality and zoobenthic community of the Mlalazi estuary and also the prawn farm effluent should continue with the present study being used as a baseline study. This will enable detection of improvement in the water quality in the estuary and to determine the extent to which natural events or anthropogenic (prawn farm) activities are responsible for causing observed changes.

- Monitoring should also include continued assessment of the prawn farm effluent, as there is at present no control measures or monitoring program in place to assess the quality of the prawn farm effluent being dumped into the estuary. The South African Coastal Zone Management Act of 1986 requires careful environmental assessment of any new development within 1 km of the high water mark. Since the prawn farm falls within this margin from the Mlalazi estuary such an assessment should have been undertaken. The potential pollution problem should have been identified and mitigating measures should have been addressed and implemented.

- Nutrient concentrations in the prawn farm effluent should be reduced at all cost prior to its discharge into the estuary so as to avoid nutrient enrichment of estuarine waters. The New Water Act of 1998 clearly states that: "*the owner of the land or a person in control of the land on which any activity or process is or was performed which causes, has caused or is likely to cause pollution of a water resource, must take all reasonable measures to prevent any such pollution from occurring, continuing, or recurring*". Measures may include ceasing, modifying or controlling any acts or processes causing pollution.

- All the stakeholders involved in utilization and control of the Mlalazi estuary, which include KwaZulu-Natal Wildlife and management of the prawn farm should participate in the development and implementation of a management plan for the estuary. Such a management plan should be nested within the proper authority (KwaZulu-Natal Wildlife), who should take responsibility for implementing and controlling the management plan.

In conclusion, there is increasing awareness of the effect of land-based sources of pollution on the coastal environment (DWAF 1995). It is also recognized that control of land-based sources of pollution is difficult and issues pertaining to it are not adequately addressed (Karau 1992). However, in order to control pollution there is a need for estuary users and managers to work together so as to produce management plans. The Thames Management Plan launched in 1996 is one such example, taking into account all aspects of the estuary including, sustainable development, agriculture, nature conservation, recreation, water management and monitoring (Thames estuary Project 1996, in Walmsley, *et. al* 1999). Such a management plan, as a long-term strategy, is therefore imperative in the Mlalazi estuary so as to improve water quality and the general ecological condition of the estuary.

CHAPTER 7

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