

**The Use of Microphytobenthos (benthic
microalgae) as an Environmental Indicator
of Past and Present Environmental
Conditions: A comparative Study of the
Mnweni Catchment and the St. Lucia
Estuary**

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St.Lucia Estuary

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Submitted in Partial Fulfilment of the Requirement for the
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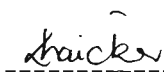
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DECLARATION

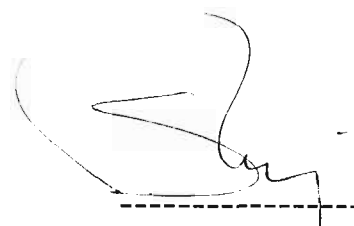
I, Deshni Naicker hereby declare that this dissertation titled:
The Use of Microphytobenthos (benthic microalgae) as an Environmental Indicator of Past and Present Environmental Conditions: A comparative Study of the Mnweni Catchment and the St. Lucia Estuary, is a result of my own investigation and research and has not been submitted in part or in full for any other degree or to any other institution or university.



Deshni Naicker

28 March 2007

Date



Dr S Pillay

28 MARCH 2007

Date

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ABSTRACT

Diatoms are a large and diverse group of algae which are distributed throughout the world in all types of aquatic systems and are one of the most important food resources in freshwater ecosystems. The need to sample diatoms arises from the necessity to know the history of the water quality. Diatom communities reflect the overall ecological integrity by integrating various stressors and as result provides a broad measure of their impacts. They are recognized as valuable organisms for bio assessment, due to their ease of identification, rapid life cycle and their visibility to the naked eye. Diatom communities provide interpretable indications of specific changes in the water quality

Environmental gradients and assemblage composition in the Mnweni River catchment revealed that water chemistry, habitat structure, flow type and land use were the most important environmental factors for diatom assemblage composition in the Mnweni River. The results indicates that the diatom diversity and assemblage composition in the Mnweni River shows that changes in the water chemistry as well as organic pollution, but also indicate changes in the habitat character which is related to the water flow, river bank character and catchment land use. The diatom assemblages that were found at the sites are typical of clean or mildly enriched water conditions.

The changing salinity in St. Lucia is the main physical factor that is driving the constant change in ecological conditions within the estuary. The changes in the diatom community structure in the estuarine system indicate the variable nature of the benthic communities under different salinity conditions. Contrary to the drought conditions that are being experienced, the diatom biomass did not show any significant changes or differences. This may be due to the rapid recovery ability that diatom communities exhibit in response to changes that are occurring in their environment.

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CHAPTER ONE

1. Introduction and Problem Contextualisation

1.1. Preamble

The rapid growth of the human population across the globe has placed severe stress on the supply of potable water. In many countries, consumption is already exceeding supply as extra demands for industrialization and agriculture grows apace. South Africa is broadly categorized as a semi-arid country and most natural fluvial systems have already been modified to varying degrees to ensure year round supply to meet agricultural, industrial, municipal and human needs. *Bate et al., (2004)* notes that the increased abstractions from natural systems tend to enhance despoliation of water further downstream due to increased total dissolved solids entering the systems as return flow from agricultural lands and from sewage treatment works. *This problem may be further exacerbated by poor catchment management practices and through increased erosion losses from farmlands.*

The water quality of semi- arid South Africa will consequently have to be managed efficiently if all the requirements of this developing country are to be met from limited resources. Coupled with efficient management is the necessary requirements of continuous monitoring and assessment of water quality of natural systems, particularly in their lowest reaches. These estuarine environments are well known as one of the most productive ecosystems in the world and serve a myriad of functions of numerous organisms that may live out part or all of their lives here (*Bate et al., 2004*).

Consequently, it is of utmost importance that the health status of estuarine systems, that are so dependant on water quality, be maintained at an optimum level.

During pristine times, large permanently open estuaries in Southern Africa had a regular flow of river water, interspersed at intervals by droughts and floods. The estuaries had an unblocked link to the river inland, a continuous or discontinuous connection to the sea and occasional floods. An estuary that is deprived of this fresh water input loses its fundamental physical characteristics and begins to function as an arm of the sea or as a marine embayment (Bate and Adams, 2000). Anthropogenic impacts within the catchments of fluvial systems have almost always impacted negatively on *estuarine health*.

Maintenance of the rich biodiversity of the estuarine environment is dependant *inter alia*, on the quantity and quality of freshwater input into the system. Whilst water quality may be monitored by abstracting and analysing point samples for a range of chemical parameters, it is recognized that this method merely provides a snapshot of the water quality status at the time of sampling. The temporal variation of most water quality variables is usually high in lotic environments and this is why biological indicators are advocated if they can accurately provide an indicator of the level of water quality (Bate *et al.*, 2004). This method becomes particularly useful and convenient if biotic communities can be used to accurately assess water quality with a lower degree of variability than can the snap-shot samples taken at different sites (Stevenson and Pan, 1999).

It has been internationally recognized that diatoms can be effectively used as water quality indicators. Research shows that assemblages of diatoms provide a valuable aid in determining water quality and more importantly, in undertaking environmental assessments of environmental change. Approaches, particularly those that examine the ecological associations of phyto- and epiplanktonic communities provide a powerful description of the chemical and biological cause and effect pathways. It has been shown that in many cases the use of diatoms for predictive assessments provide a more seasonally durable methodology when compared to similar approaches that

are based on invertebrate populations (info@dhec.co.za). Diatom material that is retained in sediment layers provides an historical perspective that can be retrieved through the sediment coring. Such analyses can provide a fingerprint of change that can extend back over centuries (info@dhec.co.za). Research has shown that the water quality of a system can be affected due to socio economic and environmental impacts on the system and as a result of this the microphytobenthos (benthic microalgae) community changes according to the environmental conditions.

1.2. Problem Contextualisation

The CSIR at Durban has a comprehensive diatom collection dating back several decades and covering virtually all parts of the country. The diatom collection for St. Lucia dates back four decades and has been made available for this study. This collection will be used as the basis for the comparative study of the microphytobenthos (benthic microalgae) community structures present in a pristine riverine environment and one that has recently seen a marked increase in human utilization over the past few decades. In the latter instance, and taking into consideration all the changes that are occurring at the *St. Lucia Estuary* (natural morphological changes; the increase in tourism and mouth manipulation by the Greater St. Lucia Wetlands Authority), there is a need to determine the extent to which benthic microalgae communities reflect the postulated changes that have occurred that could be correlated with the water quality changes over the same period. This would provide the basis for developing a water quality index based on biological parameters.

To further enhance the outcomes of this study, a parallel study will be conducted using a pristine riverine environment in the Drakensberg. The source of the Mweni River in the KwaZulu Natal escarpment is a pristine environment that has not yet been severely impacted upon by human intervention. Consequently, this area forms an ideal control of the study in that diatom assemblages would not have been impacted upon and may be truly reflective of pristine communities. Accordingly these communities could form

the basis for comparative study of the changes that have occurred at the St. Lucia Estuary due to anthropogenic and other factors.

Whilst the collection is fairly complete for St. Lucia, no existing data is available for Mnweni. The researcher will have to collect this material which will therefore represent modern day diatomic assemblages. This will be compared with similarly sampled modern day assemblages at St. Lucia.

1.3. Motivation for Study

The National Water Act (no. 36 of 98) recognizes rivers, groundwater, wetlands and estuaries as resources and requires that they are protected so that they will be sustained into the future. In order to achieve this there is an ongoing interaction between water engineers and biologists. This interaction is taking place in an effort to try and solve the difficult questions that arise from waterways and the ecological need to allow water to flow as naturally as possible.

The comparative analysis will reveal the extent to which changes have occurred at St. Lucia whilst the historical environmental changes that have occurred there will be inferred from a provenance study of the CSIR collection.

This is an interesting study in which diatoms are to be used as water quality indicators. The use of diatoms in South African water quality studies is virtually non-existent. The South African Diatom Collection is the product of many years of dedicated collection and taxonomic work, is a national resource that remains virtually unused. This study is an attempt to unlock the potential of this collection.

There are few researchers in this field in this country and most of them have been employed by the petroleum industry – therefore there is considerable opportunity to expand the study and contribute meaningfully to this field.

1.3.1. Further to the above are other considerations that include:

- This method of determining environmental quality and changes has been shown to be very accurate;
- Data and samples covering several decades is available;
- Other assistance – literature; microscope (describe) camera, computer gadgets – are readily available;
- Value of the study- nothing like this has been done at St Lucia and it may bring to light the impacts of natural / human induced changes on the estuary / lakes;
- The possibility of using this as a springboard to undertake similar research on a broader scale; and
- The St. Lucia Estuary is a very important heritage site in South Africa, and our understanding of what it was forty years ago as to the current situation will equip us with the knowledge to ensure a better management of the system in the future;

1.4. AIM

The aims of this study are twofold:

- To use the microphytobenthos (benthic microalgae) as an indicator tool of the current status of St Lucia and to compare this with the MPB communities of the pristine Mweni catchment in the KZN Drakensberg;
- To ascertain the environmental changes that have occurred at St. Lucia over a period of forty years using the diatom collection at CSIR

1.5. OBJECTIVES

- To determine how the microphytobenthos community has changed due to environmental changes that have occurred at the estuary over the

forty years and to review the socio- economic and environmental impact conditions under which it was found then.

- To make a comparative study of the microphytobenthos communities at Mnweni and the St. Lucia Estuary
- To use the information from the first objective to qualify the extent to which the St. Lucia Estuary may have degraded through natural and anthropogenic influences.
- To study the microphytobenthos communities that existed in the St. Lucia Estuary study area over the forty years.
- To ascertain the environmental conditions relevant to the microphytobenthos communities identified for different time periods over the forty years.
- To relate community structural changes to changing environmental conditions over the study period.

1.6. Structure of the Study

Literature forthcoming is to evaluate Quality and Biological monitoring is reviewed in chapter two of the investigation. This is followed by a description of the area in which the study was carried out. The fourth chapter highlights the methodology employed to affect the study while the penultimate chapter focuses on the results and discussion of the data obtained from the implementation of the techniques described in the previous chapter. The sixth and final chapter of this research presents the reader with recommendations and an overall conclusion to the investigation.

1.7. Conclusion

This chapter focused on the contextualisation of the problem, presented the motivation for the study, the aim and objectives and the overall structure of the thesis.

CHAPTER TWO

Literature Review

2. The use of microphytobenthos as an indicator in assessing Water Quality: A Theoretical Review

2.1. Introduction

As we enter into the 21st century we face the challenge of the growing demand for land and water to meet the needs of our growing populations. At the same time we face the increasing uncertainty of the impacts of climate change and loss of resource capacity due to loss of water quality. Our ability to secure adequate clean water is a function of our ability to manage both water and land and to manage our activities so that the water secured is clean *and used in the most efficient way. In addition to human need for water there is the need for water to support the ecosystem, to maintain natural functions, plant and animal diversity and productivity and the capacity of the environment to store and retain water and to cleanse it* (Vitousek *et al.*, 1997).

Humans have for a long time been using aquatic resources to provide a number of services and functions such as supplying drinking water, fishery and transportation and for recreational purposes. Human-induced impacts on ecosystems are at the same time increasing at an alarming rate resulting in a *substantial loss of biodiversity* (Vitousek *et al.*, 1997). There is also a growing concern that not only species diversity is lost, but also that ecosystem services may become lost due to anthropogenic stress (Daily *et al.*, 2000). There is a long tradition in ecology of considering the effect of abiotic factors (*ecosystem processes*) on biodiversity and for water management purposes the assessment of water quality has for a long time been based on physical, chemical or biological data (Loreau *et al.*, 2000). Many of the ecosystem functions such as species and habitat diversity, species interaction and

mineralization of organic matter are connected to water quality and therefore a more integrated approach to water management is needed (Knoben *et al.*, 1995).

The relationship between biodiversity, ecosystem and environmental processes has become a central issue and challenge in the last decade (Cardinale *et al.*, 2000, Loreau *et al.*, 2000, Loreau, 2001). These interests have spurred research to determine how species loss alters the rates of ecological processes that are vital to retain inherent ecosystem functioning. Although the contribution that species diversity has *per se* on ecosystem function is currently debated, several studies have, nonetheless illustrated that biological communities regulate important ecological processes such as productivity, decomposition and elemental cycling and that species loss can alter the structure and functioning of ecosystems (Naeem *et al.*, 1994, Loreau *et al.*, 2000, Petchey *et al.*, 2004). Some modelling studies indicate, however, that there may be no single, general relationship between diversity and production. The relationships between species diversity and ecosystem processes are seen to be complex (Cardinale *et al.*, 2000). Cohen *et al.*, (2003) and Petchey *et al.*, (2004) recently showed that interactions between species are likely to determine what effect species loss will have on rates of ecosystem functioning.

The challenge is to understand the natural system and their life sustaining roles in supporting humans, plants and animals. To this end there are a number of well-known functions of the water systems that we need to understand with greater depth. These include the water balance, the pathways of water movement and the pathways of the movement of containments, patterns and processes of water capture and use and the habitat functions, which water, provides. In addition we need to improve our understanding and management of water use and conservation at local, regional, national and international levels. This includes the legal, institutional, social and economic structures and processes, which form the basis of the management system (Daily *et al.*, 2000).

Sustainable development of watersheds and wetlands has become the objective to which improved science is being applied. Sustainable development requires that we consider not only the human use objectives of water management but also the maintenance and improvement of the ecosystem which supports the plant and animal populations with which we share the system. Sustainable development of watersheds also requires that we deal with the long-term well being of the system in a climate of change and uncertainty (Loreau *et al.*, 2000).

2.2. The use of Biological Material

Natural events and anthropogenic influences can affect the aquatic environment in many ways: synthetic substances may be added to the water, the hydrological regime may be altered or the physical or chemical nature of the water may be changed (Anderson, 1993). Most organisms living in a water body are sensitive to any changes in the environment, (such as chemical contamination or decreased dissolved oxygen arising from sewage inputs) (Mann & Droop, 1996). Different organisms respond in different ways. As stated by Mann & Droop, 1996: *The most extreme responses include death or migration to another habitat. Less obvious responses include reduced reproductive capacity and inhibition of certain enzyme systems necessary for normal metabolism.* Once the responses of particular aquatic organisms to any given changes have been identified, they may be used to determine the quality of water with respect to its suitability for aquatic life (Friedrich *et al.*, 1992).

Biological organisms can indicate the integrated effects of all impacts on the water body, and can be used to compare relative changes in water quality spatially and temporally (Reid *et al.*, 1995). Alternatively, aquatic organisms can be studied in the laboratory using standardised systems and methods, together with samples of water taken from a water body or effluent (Friedrich *et al.*, 1992). These tests, sometimes known as bio tests, can provide information on the intensity of adverse effects resulting from specific anthropogenic influences, such as the discharge of effluents into surface or

groundwater systems (Friedrich *et al.*, 1992). Most kinds of biological analysis can be used alone or as part of an integrated assessment system where data from biological methods are considered together with the data from chemical analyses and sediment studies (Bartman & Balance, 1996). A full appreciation of natural changes and anthropogenic influences in a water body can only be achieved by means of a combination of biotests and ecological methods. Sometimes these studies have to be carried out over a period of many years in order to determine the normal variation in biological variables as well as whether any changes (*natural or human induced*) have occurred or are still occurring. An example of a continuous programme of biological assessment using a variety of methods is that carried out in Lake Baikal, Russia (Kozhova and Beim, 1993). The programme is carried out every season in Lake Baikal to see the changes that are occurring. The information that is gathered from the data is then compared to the previous years and seasons to see how the water quality in the lake has changed or differed.

2.3. Biological Monitoring Techniques

Biological monitoring techniques have been introduced as part of routine monitoring programmes due to certain shortcoming in standard physical and chemical methods. They satisfy all the conditions to qualify as suitable indicators, in that they are simple, capable of quantifying changes in water quality, applicable over large geographic areas and can furnish data on background conditions and natural variability. Due to the difficulty and the cost to chemically analyse every potential pollutant in a sample of water and to interpret it in terms of impact severity, it makes sense to monitor aquatic biota. Biological monitoring are cost effective and the results can be obtained rapidly. The main advantage of using a biological approach is that it examines *organisms whose exposure to pollutants is continuous*. Thus species that are present in riverine ecosystems reflect both the present and past history of the water quality in the river, allowing the detection of disturbances that might otherwise be missed (Eekhout *et al.*, 1996).

Aquatic communities, both plants and animals integrate and reflect the effect of chemical and physical disturbances that occur over extended periods of time. These communities can provide a holistic and an integrated measure of the integrity or the health of the river as a whole (Chutter, 1998). Numerous methods have been developed for the bio assessment of the integrity of aquatic systems. Some of these are based on one or other aspect of a single species, but most are based on the attributes of whole assemblages of organisms such as fish, algae or invertebrates. Although methods have been available for many years, *bio monitoring has only recently become a routine tool in the management of South Africa's inland waters* (Hohls, 1996).

According to (Dickens & Graham, 2002) benthic macro- invertebrates are recognised as valuable organisms for bio-assessments, due largely to their visibility to the naked eye, ease of identification, their rapid life cycle that is often based on seasons and their largely sedentary habits. Currently, the backbone of the National River Health Programme is the South African Scoring System (SASS), which is a macro-invertebrate index that has been developed by Chutter in 1998. *The SASS system has undergone several refinements in order to suit all conditions; the most recent of these modifications refereed at as SASS 5 has been by Dickens and Graham in 2002.*

Round (1993) states numerous reasons why diatoms are regarded as useful tools of bio monitoring, amongst which the following bears special relevance to the South African situation; methods are cost effective, the data are comparable, techniques are scientifically recognized; results, with a higher degree of accuracy, can be obtained without lengthy delays and non-specialists, equipped with a reasonable knowledge of Biology together with illustrated guidelines can, very easily, embark on identification and counts.

However, concern has been expressed about the transfer and comparison between the Northern and Southern Hemisphere (Round, 1991). It is also well known that some species have the same Morphology or Biological Structure, but questions still remain concerning the range of ecological tolerances of these various species. This is of great concern when distance, climatic conditions and other environmental pressures are taken into account. However, Kelly *et al.*, (1998) stated that diatoms are 'sub cosmopolitan', i.e. they occur anywhere when certain environmental conditions are fulfilled. This concept suggests that the composition and geographic distribution of diatom communities is influenced by specific environmental conditions at any particular site and time

According to Kwadrans *et al.*, (1998) diatom indices may be used to overcome problems relating to the monitoring of rivers for the inorganic nutrients that cause eutrophication, organic loading, ionic composition and dissolved oxygen. Bate *et al.*, (2002) in their study of South African rivers conclude that benthic diatoms could be a useful addition to the national bio monitoring programme as they give a time integrated indication of the specific water components.

2.4. Factors that affect Biological Systems in an Aquatic Environment

2.4.1. Natural features of aquatic environments (As stated in the guide to The Use of Biota, Sediments and Water in Environmental Monitoring, 1992)

- The fauna and flora that is present in specific aquatic systems are a function of the combined effects of various chemical, physical and hydrological factors. Two of these factors specific to water bodies are:
 - The density of water, which allows organisms to live in suspension. Organisms, which exploit this, are called plankton and consist of

phytoplankton (photosynthetic algae), zooplankton (small animals) that feed on other planktonic organisms and some fish species, which feed on other fish or plankton.

- The development of rich planktonic community depends on the residence time of the organisms in the water body. Fast flowing water tends to carry away organisms before they have time to breed or establish populations, and therefore, planktonic communities are more usually associated with standing water such as lakes.
- *The abundance of dissolved and particulate nutrients in the water. The constant supply of these often allows diverse and rich communities of planktonic and benthic (those living in or on the bottom) organisms to develop. An abundance of dissolved nutrients in shallow, slow flowing or standing waters allows the growth of larger aquatic plants (macrophytes), which in turn provide food, shelter and breeding grounds for other organisms.*

2.5. Uses and Benefits of Biological Methods

2.5.1. Biological Effects used for the Assessment of aquatic Environments (As stated in The Guide to The Use of Biota, Sediments and Water in Environmental Monitoring, 1992)

The use of Biological Methods to Assess Natural and Anthropogenic effects on aquatic organisms

A variety of effects can be produced on aquatic organisms by the presence of harmful substances or natural substances in excess, the changes in the aquatic environment that result from them, or by physical alteration of the habitat. Some of the most common effects on aquatic organisms are:

- Changes in the species composition of aquatic communities

- Changes in the dominant groups of organisms in a habitat
- Impoverishment of species
- High mortality of sensitive life stages, e.g. eggs, larvae
- Changes in the behaviour of the organisms
- Histological changes and morphological deformities

2.5.2. Advantages of the use of Biological Methods for assessment of water quality

Biological assessment is often able to indicate whether there is an effect upon an ecosystem arising from a particular use of the water body. It can also help to determine the extent of ecological damage. Some kinds of damage may be clearly visible, such as an unusual colour in the water, increased turbidity (Balloch *et al.*, 1976). However, many forms of damage cannot be seen or detected without detailed examination of the aquatic biota.

Aquatic organisms integrate effects on their specific environment throughout their lifetime. Therefore, they can reflect earlier situations when conditions may have been worse (Bartram & Balance, 1996). This enables the biologist to give an assessment of the past state of the environment as well as the present state (Friedrich *et al.*, 1992). The length of past time that can be assessed depends on the lifetime of the organisms living in the water under investigation. Micro-organisms, such as ciliated protozoa, periphytic algae or bacteria, reflect the water quality of only one or two weeks prior to their sampling and analysis, whereas larvae, worms, snails, and other macro invertebrate organisms reflect more than a month and possibly several years of impacts on the water body (Friedrich *et al.*, 1992).

2.5.3. Further to the above biological methods can be useful for:

- Providing systematic information on water quality

- Managing fishery resources
- Defining clean waters by means of standardised methods or biological methods
- Providing an early warning mechanism, e.g. for detection of accidental pollution, and
- To assess water quality with respect to ecological, economic and political implications

2.6. Estuarine Microphytobenthos (benthic microalgae)

Estuarine microphytobenthos (benthic microalgae), which are regarded, as the jewels of the water due to their intrinsic shells largely inhabit mudflats, intertidal sand, vegetation beds and sediments of the coastal systems and are composed primarily of diatoms as well as cyanobacteria (blue green algae) and chlorophytes (green algae) (Sullivan and Moncreiff, 1988; Pinckney and Zingmark, 1993; MacIntyre *et al.*, 1996). The microphytobenthic community which are found in intertidal sediments has to adapt to a variety of stressful factors such as sediment transport, temperature, nutrient concentrations and also the steep gradients in light to ensure maximum productivity (Barranguet *et al.*, 1998). However, due to their intrinsic ability to recover or to adapt to physical changes in their habitat, certain species have the ability to be motile (which could migrate vertically up and down in the uppermost sediments in relation to environmental changes) than others, which are non-motile (bound to the sediment) (Wulff *et al.*, 1997; Round, 1965).

Although microphytobenthos are generally confined to the uppermost few millimetres of sediment due to their dependence on light for their growth, they have also been found in substantial numbers in depths of 10 cm or more (de Jonge and Colijn, 1994; MacIntyre *et al.*, 1996). Microphytobenthic algae at these depths have been shown to play a role in limited production because of the absence to light (Jorgensen and Des Marais, 1986; Pinckney and Zingmark, 1993), (Steele and Baird, 1968; Cadée and Hegeman, 1974; Delgado, 1989; Light and Beardall, 1998). Once these organisms have been exposed to light again they resume to their normal photosynthetic activities.

Recent research has shown that both temporal and spatial patterns in the composition and abundance of microphytobenthos are controlled largely by their interactions between light, grazing pressure and nutrient availability (Bianchi and Rice, 1988; Posey *et al.*, 1995; Bennett *et al.*, 2002). Changes in microphytobenthos abundances are likely to affect food webs in coastal systems because of their important role as a food resource for other benthic invertebrates (Miller *et al.*, 1996; and Bennett *et al.*, 1999). According to Bennett *et al.*, 1999: "for a comprehensive understanding of the complexity of parameters that influence the abundance and composition of microphytobenthos, further research is required on the interactive effects of irradiance, grazing pressure and nutrients."

2.7. The Role of Microphytobenthos

Microphytobenthos do not only constitute an important carbon source for local benthic food webs, but also provide an essential link as to organic matter and inorganic compounds which are available to top predators and higher trophic levels (Miller *et al.*, 1996; Mortazavi *et al.*, 2000). They are also an important food source in estuaries because they are available all year round and the periodic re-suspension of large numbers of cells that are present in the water column makes them easily accessible to filter and suspension feeders as well as benthic grazers (Adams and Bate *et al.*, 1999).

The microphytobenthic community forms a highly active bio film at the sediment water interface and as a result it has a significant impact on the exchange of dissolved and particulate matter between the water column and sediment, therefore influencing processes such as nutrient fluxes (Rizzo *et al.*, 1992; Reay *et al.*, 1995) and also the stabilization of the sediment (Yallop *et al.*, 1994; Wulff *et al.*, 1997).

Stevenson *et al.*, (1996) states that microphytobenthos can also act as a biological indicator of pollution and is also useful in the monitoring of water quality as these organisms are autotrophic and are found at the interface of biotic, chemical and physical components of the food web. Therefore, the importance of benthic micro algae present in shallow estuarine systems as food sources, oxygen producers and carbon fixers has been well documented and fully recognized (Heip *et al.*, 1995; Barranguet *et al.*, 1998). In South African estuaries and rivers microphytobenthos are being widely used for basic taxonomic studies (Archibald, 1983; Schoeman and Archibald, 1986), for the assessment of water quality (van Dam *et al.*, 1994; Watt, 1998) as well as for relating species composition to the physical characteristics of estuaries (Watt, 1998; Adams and Bate, 1999).

2.8. Microphytobenthic Biomass and their Primary Productivity

Microphytobenthic biomass distribution is a result of the interacting processes of which photosynthetic production is the most important and accounts directly for the accumulation of biomass in the sediment during the emersion periods (Guarini *et al.*, 1998). Microphytobenthos can contribute significantly to the total production in an estuarine environment or system (de Jonge and van Beusekom, 1995; Lucas *et al.*, 2000).

The primary production of benthic micro algae that live in the upper few millimetres of sediment can account for approximately 30 percent of an estuaries annual carbon budget (de Jonge and Colijn, 1994). This is of particular importance in the winter and early spring, when the production in the water column is still limited due to the high turbidity (Lucas and Holligan, 1999). Therefore, in turbid estuaries, production of microphytobenthos is restricted to the emersion period due to light limitation during a flood tide (Colijn, 1982; Barranguet *et al.*, 1998). According to MacIntyre *et al.*, (1996): "the total primary productivity of microphytobenthos is usually higher in sheltered muddy habitats than in that of sandy ones, which can be seen in the

biomass trends. However in a temperate estuary the microphytobenthic biomass is constant throughout the entire year.”

2.9. Microphytobenthic community structures

The microphytobenthos includes representatives of several algal classes (Baccillariophyceae, Chlorophyceae, Cyanobacteria, Dinophyceae). On sandy and muddy substrate, edaphic microalgae living on a variety of benthic surfaces are often dominated by diatoms (Colijn & De Jonge, 1984; Admiraal *et al.*, 1984; de Jonge & Colijn, 1994; Agatz *et al.*, 1999) whereas coccal and filamentous green algae and Cyanobacteria are usually known to occur at some seasonal stages (Yallop *et al.*, 1994; Taylor & Paterson, 1998; Hillebrand & Kahlert, 2001; Nozaki *et al.*, 2003). Microphytobenthic diatom populations are usually composed of pennate, prostrate forms, which are either epipsammic or epipellic (Daehnick *et al.*, 1992; Moncreiff *et al.*, 1992; Gaetje, 1992; Agatz *et al.*, 1999; Mitbavkar & Anil, 2002).

Epipsammic diatoms are monoraphidean, araphidean, biraphidean and centric species of small size that grow attached to sediment particles to which they are glued by mucilaginous pads or stalks. The epipellic forms which actively move through the sediment by means of the mucilaginous secretion of their raphes (Round, 1971). However, the difference between the two categories is not absolute as there are epipsammic diatoms that are capable of movement, though generally much slower than epipellic species (Harper, 1969) and furthermore, many diatom genera have representatives in either of these groups (e.g. *Nitzschia* sp., *Navicula* sp., *Amphora* sp.) (Wolff, 1979). In general, the relative exposure to wave action and currents is thought to favour the dominance of epipsammic biomass and to provide relative shelter to the dominance of epipellic forms. In contrast, to epiphyton or periphyton communities where a distinct three-dimensional layer is usually developed, these patterns are missing on microphytobenthic bio films and only few erect

forms are present. Thus, microphytobenthos is characterized as a distinctly flat, two dimensional community (Miller *et al.*, 1987).

Despite the typical characteristics of a microphytobenthic community, it has increasingly been seen that benthic algae may not be strictly edaphic and that plank tonic forms can temporarily dwell on sediments (Drebes, 1974, Gaetje, 1992; De Jong & De Jonge, 1995). The same algal classes are found in both the phytoplankton and the microphytobenthos and the basis of separating these types are mainly due to morphological and preferred habitat characteristics. However, the distinction between benthic and pelagic life styles can be fluid especially in shallow water systems since, the exchange of organisms between the sediment and the water column is common. Water currents can resuspend the microphytobenthos and wave action and thus they then dwell in the water column and contribute to the plank tonic community (Drebes, 1974; De Jonge & van Beusekom, 1995; De Jong & De Jonge, 1995). On the other hand, phytoplanktonic organisms can sink to the bottom under calm conditions and settle on the sediment in considerable amounts where they live on and photosynthesise and become incorporated into the microphytobenthos (Potter *et al.*, 1975; Gaetje, 1992; Blomqvist, 1996). Thus, these benthic-pelagic forms must be taken into consideration.

2.10. External factors influencing microphytobenthos growth

The main limiting factors of microphytobenthos are the availability of nutrients and light. Thus, microphytobenthic assemblages are found at the uppermost surface layers of the sediments right at the sediment-water interface. As the penetration of light is largely confined to the upper 0.2- 2 mm, the distribution of benthic micro algae is restricted to this relatively thin surface layer (Wolff, 1979; MacIntyre *et al.*, 1996). The layers in which light is good enough allowing the microphytobenthos to photosynthesise vary both with the granulometry of the sediment and its organic content. Many benthic micro algae are known for their mobility and they show diel rhythms of vertical

migration, moving to and away from the surface in response to a multitude of factors e.g. light, tide cycles, desiccation, predation or resuspension (Admiraal *et al.*, 1984; Pinckney & Zingmark, 1991; Paterson *et al.*, 1998).

Although the velocities at which the cells migrate vertically are low, ranging from 10 to 27 mm (Hopkins; 1963), the ability to move is important to the algae as the top layers of the sediment represent a region with strong physical and chemical gradients. Within a few millimetres of depth, the sediment properties go from fully oxygenated to anoxic conditions and pH, sulphide, irradiance, and nutrients are known to show strong vertical variability (Joergensen *et al.*, 1983; Wiltshire, 1992; Wiltshire, 1993). But despite their variability over vertical scales, there also appear to be considerable spatial fluctuations on a horizontal scale (centimetres to meters). Possible causes for the patchy distribution are variations in the texture and relief of the sediment surface (Joergensen & Revsbech, 1983; Jumars & Nowell, 1984; Gaetje, 1992) or microscale nutrient, irradiance, water content and salinity gradients (Wolff, 1979). Because of their location at the sediment surface, the microphytobenthos plays an important role in modulating nutrient fluxes at the sediment-water interface and this is particularly important with regard to Oxygen and nitrogen budgets of the sediments (Sundbaeck *et al.*, 1991; Wiltshire, 1993; Wiltshire *et al.*, 1996). In general it is assumed that growth of benthic micro algae is not limited by nutrients, since nutrient concentrations in the pore water are generally high (Cadée & Hegemann, 1974; Admiraal, 1984). However, in the thin layer of diatoms at the sediment surface biomass may be highly concentrated, and thus nutrients may temporarily become depleted (Admiraal, 1977). When abundant, the microphytobenthos can furthermore, stabilize the sediment surface against resuspension and erosion by secreting mucilaginous films and forming thin, brownish mats or carpets (Paterson *et al.*, 1990; Delgado *et al.*, 1991; De Brouwer & Stal, 2001). These bio films are mainly formed by diatoms excreting extra cellular Polymeric Substances (EPS) whereas the amount of excretion is directly related to the rate of primary production (Cadée & Hegemann, 1974).

2.11. Environmental Conditions that promote Microphytobenthos growth

Macintyre *et al* (1996) conducted a study about microphytobenthos and the ecological roles of bare, shallow water marine habitats. As light only penetrates the sediment to a depth of 0.2- 2mm, microphytobenthos could only photosynthesis to this depth. Microphytobenthos live, grow and are consumed in the top few millimetres of these shallow, bare ecosystems. Phytoplankton and microphytobenthos use light energy to fix CO₂ into organic matter. The depth distribution of microphytobenthos depends on the extent of the currents, sediment mixing by waves and tidal actions, and the abundance of benthic macro fauna. Microphytobenthos can be limited to the upper few mm of oxygenated sediments, due to the low energy organic rich environment. Micro algae can be found to a depth of 10cm in well-mixed sandy sediments within high-energy environments.

2.12. Microphytobenthic life cycles

The life cycles of benthic micro algae are complex and but past research does allow for some broad generalizations can be made. The biomass in sheltered, muddy habitats is higher than in exposed sandy habitats (Cadée & Hegemann, 1974; Colijn & Dijkema, 1981; Delgado, 1989; Sundbaeck *et al.*, 1991). Variations in the biomass of microphytobenthos in adjacent but distinct habitats can be as great as those over large geographic distances (Sullivan & Moncreiff, 1990; Pinckney & Zingmark, 1993; Moncreiff & Sullivan, 2001). In temperate regions microphytobenthic biomass, primary production and chlorophyll contents show a spring or summer maximum similar to biomass peaks of plank tonic algae (Kann, 1940; Admiraal & Peletier, 1980; Khondker & Dokulil, 1988; De Jonge & Colijn, 1994; Sundbaeck *et al.*, 2000; Nozaki *et al.*, 2003). Due to seasonal variations in light intensities in the northern hemisphere these blooms are of relatively short duration and with increasing

latitude they occur later in the year. Many studies on primary productivity and chlorophyll contents of microphytobenthic assemblages have been conducted over the last 30 years. However, their comparability is severely restricted by differences in methodologies (sediment volume, sampling techniques and measurement techniques), habitats and geographical distance. In response to these factors the chlorophyll contents observed ranged from less than 1 mg m⁻² (Golfe de Fos, France; Plante-Cuny & Bodoy, 1987) to 560 mg m⁻² (Ems-Dollard Estuary, The Netherlands; Colijn & De Jonge, 1984) as well as *primary production rates ranging from less than 1 mg Cm⁻²d⁻¹ at the same site in France (Golfe de Fos, France; Plante-Cuny & Bodoy, 1987) to 115 mg C m⁻² h⁻¹ in the Ems-Dollard Estuary (Colijn & De Jonge, 1984).*

The production variability can be explained by changes in irradiance and chlorophyll concentration ranges as well as by environmental factors (e.g. temperature, nutrient contents). Seasonal and temporal fluctuations not only occur in terms of total biomass but they also have been demonstrated on a taxonomic level. The dominance of particular algal groups at different times of the year have been shown by several authors and it was found that despite a general dominance of diatoms, green algae and Cyanobacteria are known to occur in abundance during the summer period (Kann, 1940; Kann, 1993; Khodker & Dokulil, 1988; Yallop *et al.*, 1994; Taylor & Paterson, 1998; Hillebrand & Kahlert, 2002; Nozaki *et al.*, 2003). Furthermore, seasonal succession has also been demonstrated on genera and species levels and thus, inter-annual taxonomic fluctuations are known to occur (De Jonge & Colijn, 1994; Khodker & Dokuli, 1988).

2.13. Food-web interactions and trophic significance

Sediment habitats represent complex aquatic ecosystems and, apart from the sediment micro flora, they are also inhabited by innumerable invertebrate species. The sediment dwellers can be classified according to their size ranges, as stated by Plante-Cuny & Plante (1984), into micro fauna (ciliates), meiofauna (harpacticoid copepods, nematodes, ostracods) and macro fauna

(mainly amphipods, isopods, gastropods, polychaetes, mussels). Most of these taxa can be defined as deposit- and suspension feeders that consume, obligatory or optionally, herbivorous food items. Furthermore, smaller benthic organisms may also play a governing role as trophic linkages to macro faunal consumers or other predators (e.g. demersal fish, birds). Apart from detritus and bacteria, the secondary production in shallow aquatic systems can be supported largely by the primary productivity of benthic micro algae (Daehnick et al., 1992; Moncreiff et al., 1992; Miller et al., 1996; Moncreiff & Sullivan, 2001). *Studies on grazer-micro algae interactions have stressed the relative importance of the microflora as a food source for benthic consumers* (Fenchel 1968; Fenchel, 1975 a; Sumner & McIntire, 1982; Plante-Cuny & Plante, 1984; Underwood & Thomas, 1990; Hillebrand et al., 2002; McCormick & Stevenson, 1991). *Consequently, there is now general consensus that the microphytobenthos can be considered as the major food source for herbivore invertebrates in the euphotic zone.* Additional support to these findings has been given by recent stable nitrogen, carbon and sulphur isotope studies which have demonstrated very convincingly that benthic micro algae are the basis for secondary production at the bottom of shallow freshwater and marine aquatic systems (Sullivan & Moncreiff, 1990; Hecky & Hesslein, 1995; Moncreiff & Sullivan, 2001; Herman et al., 2000; James et al., 2000b).

Due to its abundance and productivity, the microphytobenthos is considered to be a reliable and a highly nutritious food source (Fry & Sherr, 1984; Kitting et al., 1984; Plante-Cuny & Plante, 1984; Decho & Fleeger, 1988; Jernakoff et al., 1996; James et al., 2000b; Moncreiff & Sullivan, 2001). Thus, there is sufficient evidence that *the relative importance of labile fractions derived from the renewable pool of microphytobenthos by far outweigh the significance of refractory detritus material as a food source for benthic organisms.*

2.14. Importance of benthic microalgae for primary production

Algae play an important part in primary production and are a major food source for many organisms. When dividing production and respiration among benthic size groups, bacteria and microalgae are the most productive within tropical and temperate intertidal sediments (Blanchard *et al.*, 2001). On tidal flats benthic macro-organisms receive their nutritional needs by feeding on benthic micro algae and labile organic matter settling out of the overlying water (Alongi, 1998). When the high tide covers the mud flat, filter-feeding benthos such as bivalves and polychaetes may consume up to 25% of the phytoplankton production of $82\text{g Cm}^{-2}\text{ yr}^{-1}$. Some remaining phytoplankton carbon is exported by the tide and deposited on the sediment surface (Alongi, 1998).

As stated by Alongi, (1998) Macroalgae, benthic microalgae, phytoplankton, epiphytes and neuston may, in total, contribute to more than half of total net production in some systems. The relative contributions of different autotrophs to annual net primary production ($\text{gCm}^{-2}\text{ yr}^{-1}$) in some salt marshes and Australian mangrove forest are: marsh/mangrove-2969, microalgae-104, phytoplankton-150, epiphytes & neuston-260 (Alongi, 1998) to 25% of the phytoplankton production of $82\text{g Cm}^{-2}\text{ yr}^{-1}$. Some remaining phytoplankton carbon is exported by the tide and deposited on the sediment surface (Alongi, 1998).

Coultas and Hsieh (1997) in their study of the ecology and management of tidal marshes found that one of the main primary producers in this system was microalgae. Consumers included zooplankton, post larval fish and invertebrates, microbes, meiofauna and larger animals such as shrimps, crabs, fish and birds. Soil micro algae contributed approximately 10% of the total marsh primary production and were a major food source for secondary

producers. Benthic and plank tonic micro algae form the basis of the food chain for the fish and invertebrate fauna of the *Spartina* marsh habitats.

Microphytobenthos [benthic] microalgae provide a major energy source to the higher trophic levels in marine littoral ecosystems, especially food webs connected to intertidal mudflats. There is a net increase of microphytobenthos biomass in the top layers of the sediment during daytime exposures due to photosynthesis. Microphytobenthic communities contribute to the intertidal biological and physical processes (Blanchard *et al.*, 2001).

Guarini *et al.*, (2002) found that the two key communities that contributed to primary production in tidal estuaries were phytoplankton and microphytobenthos. Motile benthic diatoms (mainly pinnate forms) migrated upward during the day and downward at night. Microphytobenthic primary production occurs during the day on the surface of the intertidal mudflats although primary production is limited by the high turbidity in shallow areas. At the beginning of the daytime emersion period the surface of the mud reaches a saturation value, with microphytobenthos production dynamics mostly governed by the biomass specific productivity of benthic micro algae and changes in light exposure.

A study, which was conducted by De Brouwer and Stal (2001), on short-term dynamics in microphytobenthos distribution and associated extra cellular carbohydrates in surface sediments of an intertidal mudflat, showed that benthic epipellic diatoms were due to the to increased micro algal most important group of primary producers in intertidal mudflats and that these diatoms have the ability to produce copious amounts of extra cellular polymeric substances (EPS), mainly consisting of carbohydrates. The presence of diatom bio films increases the stability of the sediment surface, which can have a major affect on the morphodynamics of mudflats. Through the excretion of EPS, diatoms are responsible for the input of high-quality organic carbon into the sediment and this maybe used as a food source for heterotrophic consumers.

According to Nozais *et al.*, (2001) primary producers such as phytoplankton and the sediment which are associated with microphytobenthos have crucial ecological functions in providing links between inorganic compounds and organic matter to make these available to higher trophic levels and top predators such as fish. All fauna derive their nutritional requirements (energy and nutrients) from plants. In “bare” substrates micro algae are an important food source for juvenile fishes. Microalgae also support diverse communities of small benthic invertebrates, for example polychaetes, nematode worms, cumaceans, copepods and soldier crabs (Hollaway and Tibbets, 1995). Bacteria and diatoms are common within muddy shores and provide a primary food source for associated larger fauna (Underwood and Chapman, 1995).

Previous studies that involve grazing experiments have shown that a variety of benthic infauna and epifauna are able to consume large quantities of algal matter, with localized “bare” patches being the result of grazing on the sediment surface. The effects of grazing by large densities of bivalves can inhibit the development of microalgal communities in some marshes (Alongi, 1998). Microalgae are a major carbon source for higher trophic levels such as benthic macro faunal communities. They are known to respond to increases in food resources with population increases. When food is limited these macro faunal communities would also be affected (Stocks and Grassle, 2001).

A study using multiple isotopes has clarified the importance of micro algae, phytoplankton and mangrove material in the diet of penaid prawns. Juvenile *Penaeus merguensis* feed on mangrove creeks, while adults offshore feed on phytoplankton and benthic micro algal material with a lower intake of mangrove detritus. The majority of isotope studies have indicated that the algae and detritus are equally important food sources for macro consumers (Alongi, 1998).

The use of stable isotope analysis in Port Curtis Queensland, to investigate the contribution of plants from different estuarine habitats to support fisheries

species caught over “bare” mudflats, found the plant species most important to fish and crustaceans were sea grass and its associated microalgae, salt marsh grass and microalgae on the mudflats (Alongi, 1998).

Benthic micro algae provide an important link within the food chain as one of the mixed diet of benthic microalgae and mangrove detritus in major primary producers. In some systems macroalgae, benthic microalgae, phytoplankton, epiphytes and neuston may, in total, contribute more than 50% of total net production (Alongi, 1998). Microalgae directly support diverse communities of small benthic invertebrates such as polychaetes, nematode worms, cumaceans, copepods and soldier crabs. Microalgae lives interstitially within the sediment and form part of the local and regional fish production cycle (Hollaway and Tibbets, 1995).

2.15. Activities and subsequent impacts on Benthic Micro algae

- Extractive dredging in the coastal marine environment, impacts on marine plants, and local tidal fluctuations all influence the movement and distribution of sediments and extent of turbidity plumes throughout the water column. Increased turbidity, caused by dredging will vary the marine environment impacts depending on the prevailing tidal and current regimes,
- Potential impacts of dredging activities in coastal areas include direct loss of fish habitats, direct smothering of marine fish habitats such as *sea grass and coral and of other benthic organisms*, reduction of light from increased turbidity resulting in stress and/or mortality of photosynthetic organisms, and remobilization of heavy metals and pesticides/herbicides. The removal of relatively shallow intertidal and sub tidal fish habitats from dredging cause a disruption to the local food chain, resulting in changes to fish catches (Hopkins & White, 1998).

- The direct and indirect effects of agricultural and industrialization practices
- Pollution from point and non point sources (e.g. urban runoff and town sewage)
- Channelisation (to regulate flows and reduce flooding)
- Construction works (e.g. jetties)
- Department of Primary Industries and Fisheries Report (DPI&F) recognises that impacts from dredging and other extractive industries may cause either temporary or long-term changes to benthic microalgae production. Department of Primary Industries and Fisheries Report (DPI&F) has a key management role in mitigating these long-term impacts where adverse affects are caused to fisheries production.

To date, most studies dealing with microphytobenthic assemblages in temperate regions focused on intertidal areas in marine and estuarine habitats (Admiraal *et al.*, 1984; Pinckney & Zingmark, 1991; Paterson *et al.*, 1998; Pinckney & Zingmark, 1993; Colijn & De Jonge, 1984; De Jonge & Colijn, 1994; Herman *et al.*, 2000). The restriction of microphytobenthic research to intertidal mud and sand flats is most likely related to their great significance in terms of spatial distribution and ecological relevance. In addition, accessibility and sampling are facilitated in those regions due to low tides and thus intense research can easily be conducted. In contrast, the potential importance of the microphytobenthos in littoral zones and especially in freshwater lakes has received little attention. Thus, its role in freshwater ecosystems still remains poorly investigated (Lowe, 1996).

2.16. Ecological Significance of Benthic Microalgae

Benthic microalgae are ecologically significant in coastal marine environments from coral reefs to estuaries (Luxford, 2002). They are a major food source for benthic feeders such as prawns and other crustaceans. Suspension feeders,

such as oysters may also graze on them when they are resuspended into the water column due to current or tides (Luxford, 2002). Benthic microalgae excrete polysaccharides, which bind the sediment and minimise the influence of overlying water movements. This results in an increase in sediment stability reducing the potential for sediment erosion and resuspension. BMA communities also modify nutrient exchange, between the water column and sediments and hence may play an important role in regulating water quality of an estuary or river (Department of Primary Industries and Fisheries (DPI&F), Notes, 2003).

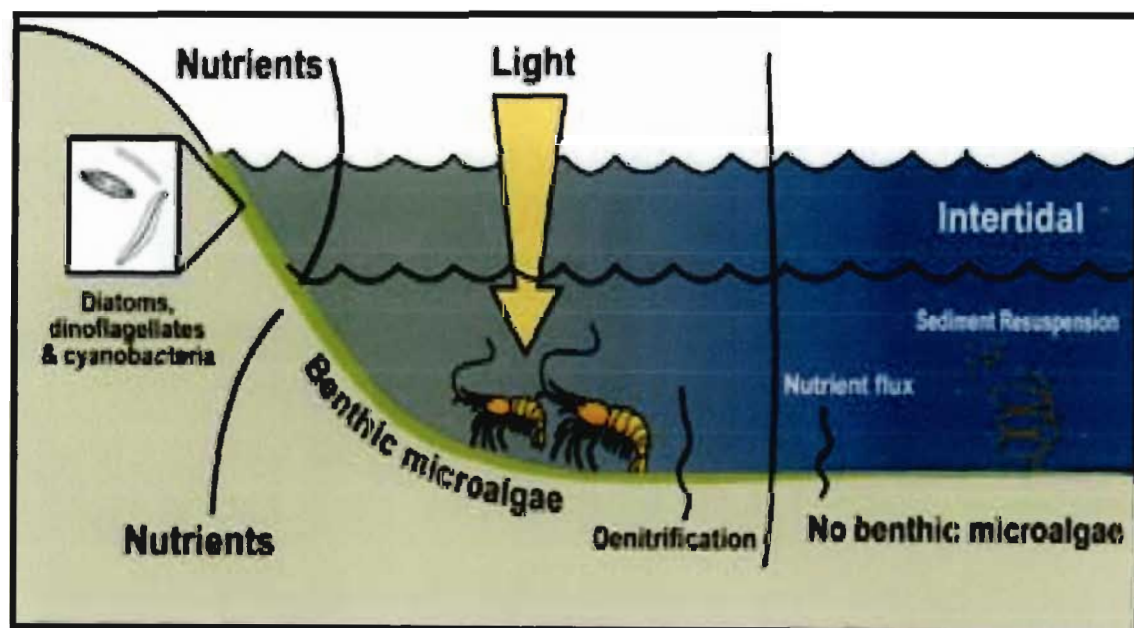


Figure 2.1 Conceptual model depicting the ecological role of benthic microalgae. (Department of Primary Industries and Fisheries (DPI&F), Notes, 2003).

2.17. Conclusion

No single group of organism is always best suited for detecting the diversity of environmental perturbations associated with human activities. If the maintenance of ecosystem integrity is the aim of riverine and estuarine

management, then there is a need to monitor the status of different taxonomic groups. Diatoms provide interpretable indications of specific changes in water quality. Diatoms comprise a highly successful and distinctive group of unicellular algae and as autotrophs, they contribute significantly to the functioning and the productivity of ecosystems, frequently forming the base of aquatic food chains (de le Ray *et al.*, 2004).

CHAPTER THREE

3. STUDY AREA

3.1. Introduction

The study area forms an important part of the thesis as it provides the reader with an in-depth description of where the research was undertaken. In this chapter the location of the study area is discussed. This chapter describes the environmental, biological and social characteristics of both the St. Lucia estuarine system and the Mweni Riverine system.

3.2. St. Lucia Estuary

St. Lucia is the largest estuarine system in Africa and without any doubt the most important in KwaZulu-Natal, as it comprises $\pm 80\%$ of the total estuarine area of the region (Begg, 1978). The St. Lucia system is situated on the coastal margin of the Zululand coastal plain on the south-eastern coast of Africa in the province of KwaZulu Natal, occupying a position between 27°52'S to 28°24'S and 32°21'E to 32°34'E (Begg, 1978, Figure 3.1.). The Greater St. Lucia Wetland Park is one of the jewels of the South African coastline and stretches from Kozi Bay in the North to Cape St. Lucia in the South. The park was the first to be declared a world heritage site by UNESCO in, 1999.

The St. Lucia system comprises the lake, which is a large coastal lagoon and the "Narrows", a tidal channel that links the lake to the sea (Natal Parks Board, 1991). Peripheral to these but important and integral to the whole system are the catchments that feed freshwater into the lake, which are the wetlands of the eastern shores, the Mkuze swamps, Mfolozi flats and the marine environment of the mouth (Natal Parks Board, 1991). The lake is separated from the Indian Ocean, by the world's highest forested sand dunes (South African. Info, 2004).

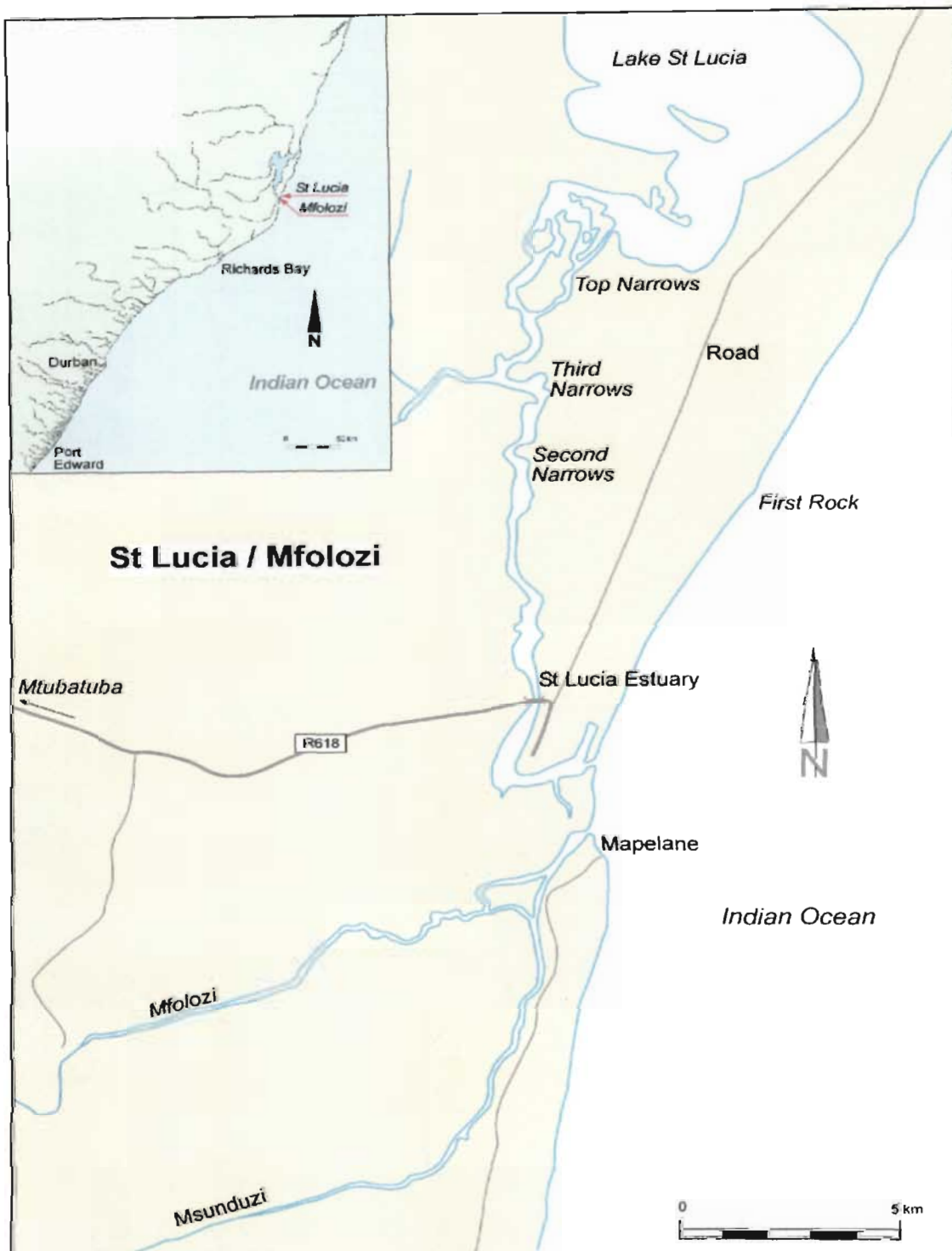


Figure 3.1 Locality map of St. Lucia Estuary

3.2.1. Environmental characteristics of the St. Lucia system

a) Physical & Chemical Characteristics

St. Lucia is 'H' – shaped with the long axis having a North South orientation (Natal Parks Board, 1991). The lake has a water surface area of about 350km² and a water volume of 330m³ × 10⁶ (Hutchison, 1974). These two values vary considerably due to the changing lake values. St. Lucia is a large and shallow estuary and has a surface area of 35 000ha and an average depth of only 90 cm. When the estuary mouth is open, water flows from the estuary to the sea whenever the water level is above that of the sea, and from the sea into the estuary when the level is below sea level (Taylor, 2003). The main rivers that flow into the St. Lucia are the Hluhluwe, Mzinene, Mkuze, Nyalazi and Mpate Rivers and together their combined catchments area is about 25 times as large as the surface of the St. Lucia estuary (Taylor, 2003).

The rivers contribute most of the water received by the estuary. Until the early 1950's the Umfolozi was partially connected to the estuary at the mouth, and at times some of the freshwater from this large river would enter the estuary (African Wildlife, 1993). This led to it being separated from the St.Lucia system when large volumes of sediment that were carried by the river, clogged up the mouth of the St.Lucia system. This is a result of the breakdown of the filtering properties of the Umfolozi swamp ecosystem due to sugar planting and canalisation (African Wildlife, 1993).

Due to the large surface area to the volume ratio, relatively large amounts of the total water volume can be lost to evaporation (Hutchison and Pittman, 1973). The falls in the lake level are compensated by the inflow of water from the sea so that evaporation results in increasing salt content and a rise in salinity levels (Natal Parks Board, 1991). Under normal rainfall conditions the salinity gradient ranges from fresh water at the river mouths in the northern

reaches of the system to seawater at the estuary mouth. During extended dry periods, the river ceases to flow and as a result the salinity levels in the system rises. The salinity gradient then reverses so that the lowest salinity is that of the seawater at the mouth and the northern reaches become hyper saline with salt levels in excess of three times seawater on occasions (Taylor, 1987).

Due to the shallowness of the system, the total water column is stirred by wind action, which prevents the formation of thermocline or halocline. The wind and wave action stirs up sediments which result in the system being highly turbid and due to the shallowness of the estuary, wide temperature fluctuations occur (Taylor, 1987).

The mouth of the St. Lucia was closed naturally in 2002 (Plates 3.4 and 3.5).

It would have probably been opened within a few weeks, but in an effort to protect it from an oil spill, a sand wall was pushed up by bulldozers to prevent the natural breaching of the mouth (KZN Wildlife, 2003). Since then the rainfall in the area has been below average and which has resulted in evaporation, which exceeded fresh water gains. Due to this, there has been a drop in the water level. The water levels are now about as those that were experienced in the 1993 drought. As the water evaporates, the salt becomes concentrated. *The mixing of water, however, has resulted in a significant reduction of salinity levels.*

Due to the mouth closure only a minor gradient may be formed, as there is more evaporation relative to water volume in the shallower North Lake and False Bay area, than in the South Lake (Plate 3.1). It would be of concern if the salinity in the whole lake was high, and the gradient did not exist, as there would then be a few places in which the estuarine organisms that cannot tolerate the high salinity levels (Taylor, 2003).



Plate 3.1. Evaporation is relative to water volume in the False Bay area



Plate 3.2. Wildlife photo showing mouth closure



Plate 3.3 Closure of estuary mouth from the sea and the diversion of the Umfolozi River into the sea.

St. Lucia is currently in a drought period, which has led to increased sediment load within the estuary. As a result, the accumulated sediment may only be removed by dredging which is currently been done at the St.Lucia estuary. However, the process of dredging is very disruptive as it impacts on the area that is being dredged. Under normal conditions, sediments accumulate slowly and as a result organisms are able to accommodate to the changes that are occurring within the estuarine habitat.

Dredging also affects the chemical and physical properties of an environment and recovery after dredging varies. As stated by Breen & Mckenize, (2001): dredging can be quick but also very slow with some sites at St.Lucia showing very little recovery after 20 years.

This is partly attributed to a change in the physical structure of an environment. Where coarse material is removed, the fine unconsolidated material that remains behind forms an unsuitable environment for benthic organisms. Dredging attempts to establish and maintain a condition that is inconsistent with the processes shaping an estuary structure and functioning,

particularly when these conditions have been modified by human activities (Breen & Mckenzie, 2001).

The National Institute of Water Research (NIWR) superficially studied the chemistry of the St.Lucia estuary in the 1960's. According to Begg (1978) a thorough understanding of nutrient relationships within the system has yet to be acquired. However, some determinations of nitrogen and phosphorous levels in False Bay in 1975 have been made. Between January and July, phosphorous was shown to be the limiting nutrient, while between August and December both nitrogen and phosphorous were equally limiting.

b) BIOLOGICAL CHARACTERISTICS

St. Lucia contains the most southerly populations of several plants and animal communities characteristic of tropical Africa. The area is considered critical to the survival of a large number of animals species, including South Africa' largest populations of hippopotamus (*Hippopotamus amphibius*), crocodile (*Crocodylus niloticus*), white-backed pelican (*Pelecanus onocrotalus*), and pink-backed pelican (*P. rufescens*). More than 530 species of birds use the wetland and other areas of the lake region. These waters also are graced by 20,000 greater flamingos (*Phoenicopterus ruber*), 40,000 lesser flamingos (*P. minor*), as well as thousands of ducks. With 36 species, this area has the highest diversity of amphibians in South Africa. Two species of sea turtles lay their eggs on the ocean shores of the eastern peninsula. Here, and nowhere else in the world, can one-find hippopotamuses, crocodiles, and sharks sharing the same waters (The Wilderness Foundation, 2004).



Plate 3.4. Rising salinity levels as the lake evaporates creates a new eco system, which attracts pelicans and flamingos in large numbers...

The biota in St. Lucia is influenced by the physical characteristics of the system and with the most important of these factors being salinity and turbidity. Other factors, which are likely to affect the biota in the area, are the changes in the temperature levels, lake levels, nutrient status and the state of the link between the lake and the sea and submerged macrophytes (Fielding *et al.*, 1991).

St. Lucia like all shallow coastal lagoons is likely to be detritus based. The main sources of detritus are the mangroves, the fringing reed beds, and input from the rivers during floods and input from the sea during low lake levels (Taylor, 1982a).

The primary producers in the system are:

(a) Algae – the lake system is turbulent and generally turbid, however the turbidity varies in response to wind events (short term) and response to salinity, which enhances flocculation (long term). Therefore, the rates of

primary production of both the phytoplankton and benthic algae are expected to be significantly influenced by three factors: turbidity, salinity and turbulence.

(b) Submerged vascular macrophytes – the dominant species are *P. pectinatus*, *Z. capensis* and *R. Cirrhosa*, which all have different salinity tolerance levels. These plants all form dense beds in which turbidity is reduced and they form the substrate for extensive colonisation by epiphyton. As the salinity rises or falls, the plants die and produce a pulse of detritus into the system. Turbulence then causes large amounts of the material to be washed up onto the shore.

(c) Vegetation – the vegetation on the shoreline consists of mangroves *Avicennia marina* and *Bruguiera gymnorhiza* are found particularly in the intertidal areas of the Narrows and a wide spread of reeds and grasses are found around the lake. Of these, the reed beds are considered the most important. It is thought that the leaf litter accumulates in the reed beds, and is then introduced into the estuarine system as a pulse of detritus when the lake levels rise (Taylor, 1987).

Much of the fascination of St. Lucia as an ecosystem is a result of its dynamic character. It is always changing, and much of its biological change is related to its changes in salinity levels. Each species of plant and animal that lives in St. Lucia has a range of salinity in which it can live. For some species, this “window” is wide, and for others it is narrow. However, above or below its particular window, the species cannot survive. Thus, there is a continual changing of species as salinity concentrations in St. Lucia rise and fall (Taylor, 2003).

Many of the estuarine organisms, especially at the medium salinities, do have life cycles that necessitate movement between the sea and estuary. This movement, however, can be restricted during mouth closure (African Wildlife, 1993). The loss of connectivity to the sea means that the species that spawn at the sea are trapped in the estuary and will grow larger without breeding.

Also affected is the recruitment of larvae and juveniles of these species into the estuary and as a result, this will affect the next generation. There is also connectivity to freshwater ecosystems (Cyrus & Viviers, 2006).

As the rivers dry up, the low salinity species cannot move upstream. However, if the overall condition of the ecosystem is maintained, then the connections to the other wetlands of Zululand and further a field are maintained. The pelicans will continue to breed and the ecosystem will be used by ducks. Under these conditions, the permanent residents of the system, such as the crocodiles and hippos and many smaller species, will also survive (Taylor, 2003).

The lake itself has now been reduced by some 30% of its surface area, which has resulted in the gradual reduction of the surface area and has led to the exposure of wide tracts of the lake bottom as the water level drops (Plate 3.5). Evaporation has now concentrated the salt content to 150ppt in the north and 55ppt in the south. Although the water level of the lake is some 90cm below the sea level, grasses irrigated by freshwater seepage from the eastern shores (Plate 3.6), have grown well along the newly exposed and dried out wetlands, providing supplementary grazing grounds for hippos (KZN Wildlife, 2003).

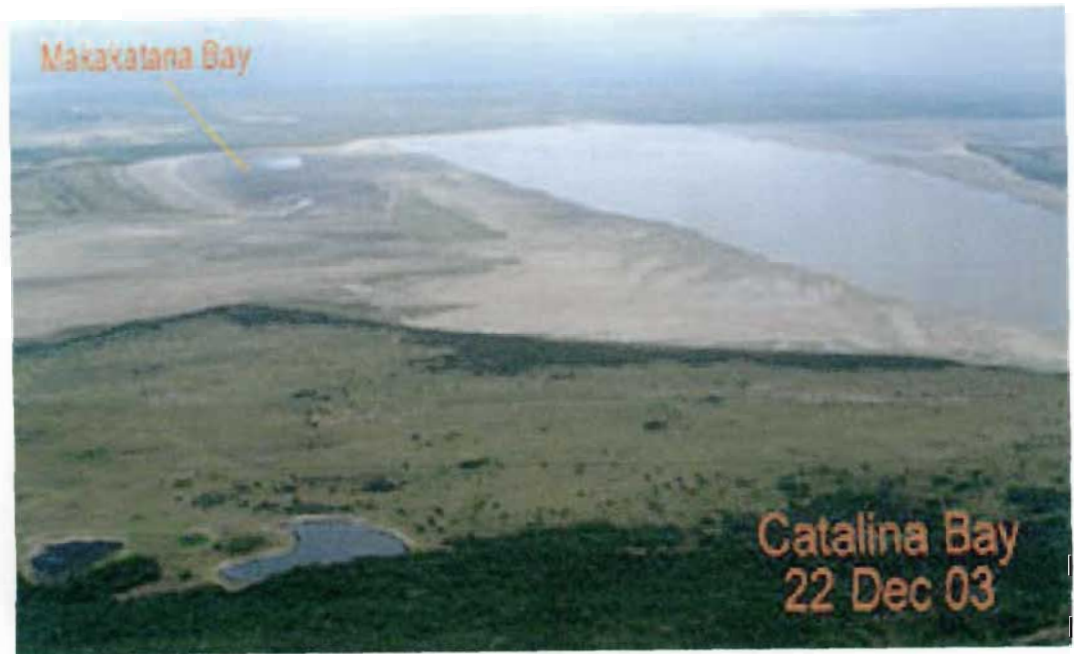


Plate 3.5. Photo of Catalina Bay



Plate 3.6. Seepage of freshwater from the eastern shores

3.2.2 Social and Economical characteristics of the St. Lucia system

a) Social characteristics and impacts on the system

Due to an increase of human interference on the system in the late 1930's a chain of events started, that has irreparably altered the estuary's once pristine condition. To a large extent can be attributed to sugarcane farmers who drained and canalised the Mfolozi Flats. Due to the canalisation and poor catchment management, it resulted in large amounts of mud being dumped into the estuary. The estuary remained blocked from 1951 until 1955, when dredging reopened it. To ensure that the Mfolozi River mud was no longer deposited into the estuary, a separate mouth for the Mfolozi was cut to the south (African Wildlife, 1993).

During the 1970's and early 1980's, a groyne was constructed to stabilise the St. Lucia estuary mouth. By constricting the tidal flow, it was hoped that the tidal scour would be enough to keep the mouth open. This was only partly successful as the dredging programme still had to be maintained (Taylor, 2003). However, in 1984 Cyclone Demoina destroyed the groyne complex, enabling the system to once again have a naturally dynamic mouth.

b) Economical characteristics and impacts on the system

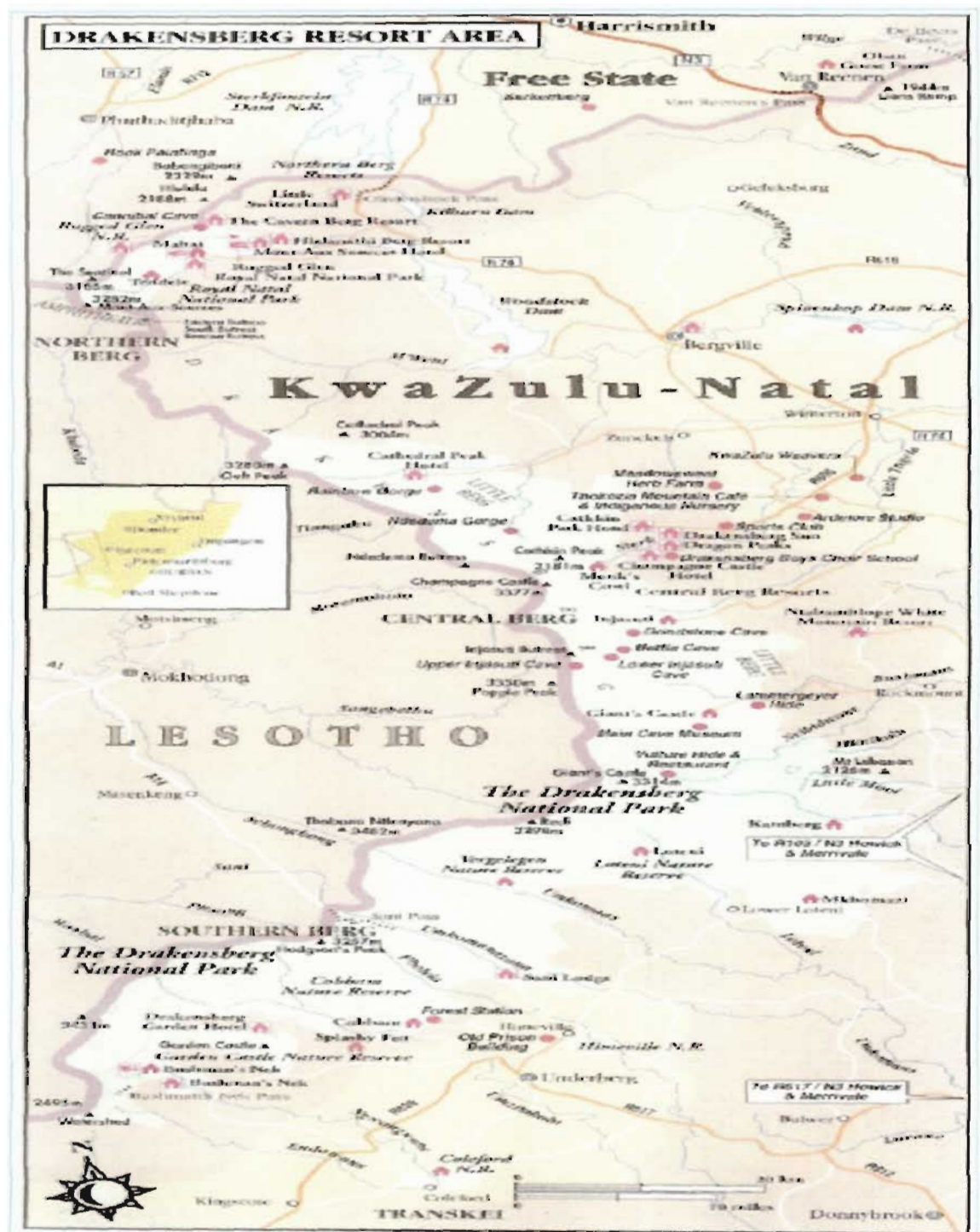
The St. Lucia estuarine system is a valuable ecosystem in terms of regional conservation, recreation, tourism and the economy. Despite its size, the system is highly sensitive and easily stressed. Problems which arise in the catchments may manifest themselves in the estuary as changing patterns of water quality, fresh water input and sedimentation which impacts upon the biotic and biotic elements of the system. In addition, the consequences of large-scale economic exploitation of the heavy mineral rich sand dunes on the eastern shores by dredge mining could be disastrous (Taylor, 2003).

The primary concern is related to the interconnectedness of the Mfolozi and St. Lucia estuary systems has been the massive input of the riverine fine sediment brought down by the Mfolozi River. These mud's will impact on both the Mfolozi and the St. Lucia systems in such as increased turbulence, modification of flow through density and sedimentation effects, the transport of pollutants through adsorption, and the ecology of these estuarine systems through a number of mechanisms, for example, light extinction, reduction of dissolved oxygen (DO) levels and once deposited on the bed of the estuary, covering and thereby damaging or destroying the habitats of a variety of estuarine organisms (50/50 website, 2004).

3.3. MNWENI STUDY AREA

The Mnweni River situated between the Royal Natal National Park (RNNP) and the Cathedral Peak, is the wildest and most pristine part of the Northern Drakensberg. The Mnweni area is the largest tribal tract in the Drakensberg and it falls under amaNgwane Tribal Authority, the chief is Inkosi Maswazi Hlongwane. Mnweni is a place of streams and passes, and natural attractions. The people who live in Mnweni are the Amangwane and Amazizi clans (Bristow, 2003). Mnweni is the typical rural area, sparsely populated and the community depends mainly on farming and natural resources for their survival. As a result, this pure untouched site has led to it being an area for ecotourism development (figure 3.3 and figure 3.4)

The Drakensberg mountain range is situated along southeast coast of South Africa. These spectacular Drakensberg Mountains are a challenge for adventure seekers and a heaven for wildlife lovers. Extending from northeast to southwest for around 1125 km, the Drakensberg is part of the Great Escarpment and the main watershed of South Africa. The local Zulu name for Drakensberg is Ukhahlamba or 'The Barrier of Spears', an accurate description for Drakensberg that raises over 3000 metres in height. It is an important region for adventure activities like mountaineering, camping, bird watching, river crossing, trout fishing, and many more such activities (Derwent, 2000; Bristow, 2003).



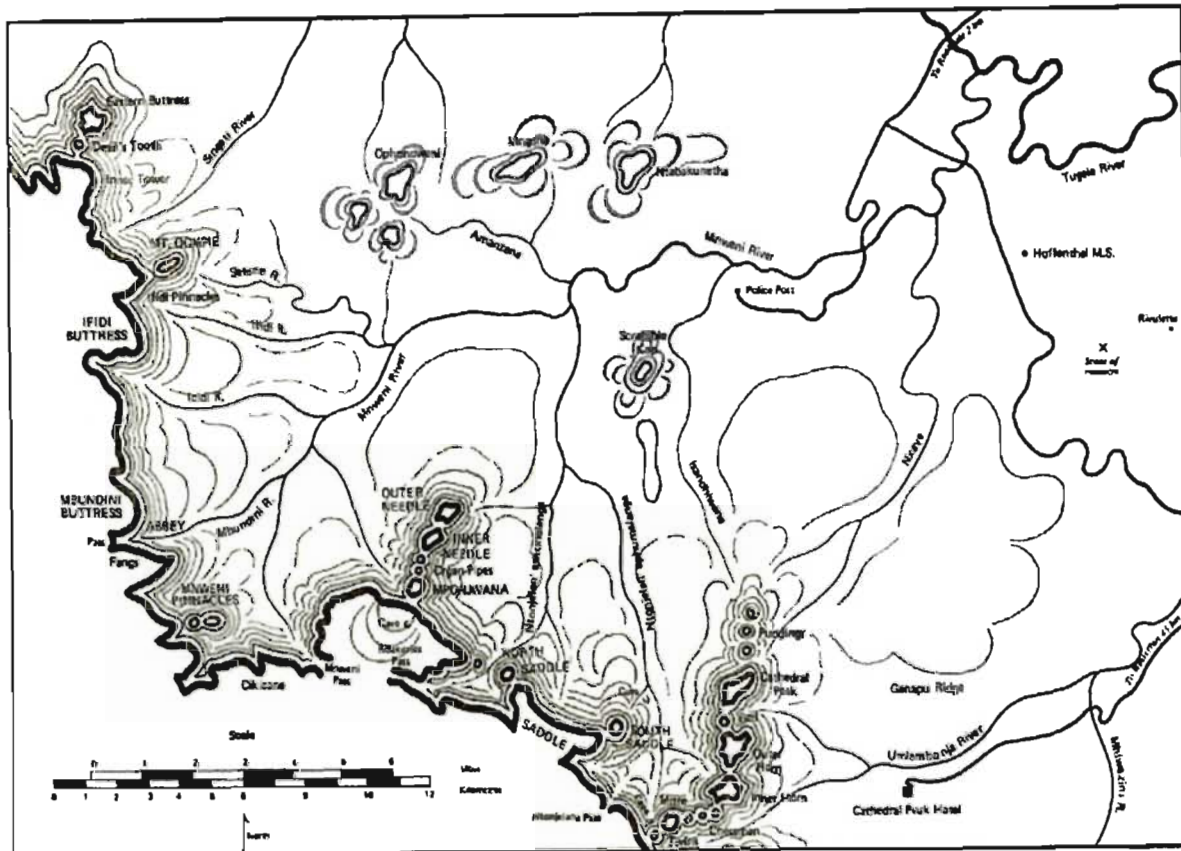


Figure 3.4: Map of the Mnweni Area (Bristow, 2003)

3.3.1. ENVIRONMENTAL CHARACTERISTICS OF THE MNWENI SYSTEM

a) Physical, chemical and biological characteristics of the area

The unique Mnweni Valley between the Ukhahlamba Drakensberg landmarks of the Amphitheatre and Cathedral peak is one of the most important sources of water in South Africa, that contributes to the delivery of clean water to rural, agricultural, urban and industrial consumers in both Gauteng and KwaZulu-Natal. The scenic, cultural and archaeological resources, enhance the area's potential for tourism development.

The physical biodiversity of the Mnweni area manifests itself through a rich mix of plants and animal species, inter-connected systems of wetlands that range from open water bodies such as mountain tarns and a variety of

marshes, to intricate networks of stream and river courses. These wetlands are distributed in a complex mosaic throughout the altitude gradient of the mountains and occupy a variety of positions in the landscape, which range from small hanging systems high on valley sides to valley bottoms marshes and extensive watercourses.

3.3.2 SOCIAL AND ECONOMICAL CHARACTERISTICS OF THE MNWENI SYSTEM

a) Social and economical characteristics and impacts of the area

Scattered huts along the foothills of the majestic Drakensberg are home to several thousand people who make a living from raising livestock and subsistence farming. However, years of overgrazing and inappropriate land management practices have degraded parts of the land (Water Wheel, 2005).

Everson, (2005) who is involved in land rehabilitation projects in the Mnweni and Okhombe, explains that when livestock such as cattle and goats are allowed to eat too much of vegetation in the area the water does not infiltrate into the ground when it rains. This leads to increase water runoff and soil erosion. Huge loads of silt also land up in the rivers of the catchments and are washed into the dams that make up the Tugela-Vaal water transfer scheme. This silt not only reduces the capacity of the storage reservoirs, but also is expensive to remove (Water Wheel, 2005).

3.3.3. MNWENI TOURIST ATTRACTIONS

3.3.4. The Mnweni Cultural Centre



Plate 3.7. Cultural Centre

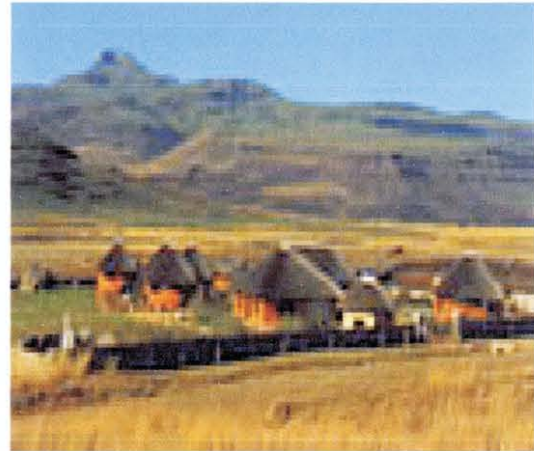


Plate 3.8. Cultural Centre

Protecting the environment, Building our community. Ukuvikela imvelo, Ukwakha umphakathi kwethu (Rand Water Community Work, Article 113, 2005)

The Rand Water Mnweni Trust was launched in 1999 and was the result of a joint effort between Rand Water, the Wildlife and Environment Society of South Africa (WESSA), Bergwatch and the communities of amaNgwane in the Upper Mnweni Valley of the Drakensberg. The Trust was established with an initial capital investment of R2 million from Rand Water (Rand Water Community Work, Article 113, 2005).

The Mnweni River Catchments area is one of the most important sources of water in South Africa. It supplies water to five provinces, namely, KwaZulu Natal, the Free State, Gauteng, North West Province and the Northern Cape.

The value and spectacular beauty of the Mnweni Valley offers a unique blend of opportunities to the local communities. The rich and vibrant culture of the

community and their intimate knowledge of the area have never been properly utilised in the development of the tourism centre (Tourism KwaZulu-Natal News, 2005). The opportunities are being explored through the Mnweni Trust, utilising the landscape's inherent beauty, its people and history. All of this is set against the backdrop of environmental and cultural sensitivity.

3.3.5. Economical characteristics

The Department of Environmental Affairs and Tourism appointed the Rand Water Mnweni Trust as the appropriate implementing agent to develop the Mnweni Valley Cultural and Hiking Centre. The grant of R2.2 million will be used for the construction of a cultural centre, access roads and water supply, walking trails and camps. There will also be rehabilitation of areas that have been exposed to soil erosion (Rand Water, Community Work, Article 113, 2005).

The Mnweni Trust is committed to the strengthening of its community trustees and the communities that they represent. A central focus is training in financial management, fund raising, management and leadership principles, intercultural group dynamics, project management and evaluation, natural resource management and strategic planning (Rand Water, Community Work, Article 113, 2005).

The Cultural Centre provides a place through which local craft skills are promoted and can serve as an outlet for marketable products. The following functions can be performed or organised through the facilities of the Cultural Centre:

3.3.6. Hiking Trails

Mnweni offers the most exciting hiking experience in the Drakensberg. There are numerous hiking trails that can be experienced, namely the Mnweni Pass, Rockeries Pass, Waterfall Cave and Ntonjelana Pass, Fangs Pass, Icdi Pass and the Ifidi Pass.



Plate 3.9 Hiking Trails

3.3.7. The Mnweni Pools

The Mnweni baths is found one kilometre upstream at the confluence of the flowing Mnweni and the Ifidi Rivers. The baths are a series of water flowing for about a hundred metres down a channel of sandstone, forming lovely pools at each level along the way.



Plate 3.10 Mnweni Pools

3.3.8. Bushmen Paintings

The small indigenous people of southern Africa, known as the San or Bushmen, once inhabited the wilderness which is now called the Drakensberg, and evidence of their stay are found in the thousands of rock paintings they left behind (Plate 3.11). They are mainly of people and their equipment, of animals and often of what appears to be hunting and spiritual activities. However, when the researcher visited the area of the paintings, it was clear that the paintings were fading and some were being destroyed by people who tried to chip the rocks.



Plate 3.11. Bushmen paintings

There are many other tourism activities or attractions available in Mnweni, which include. horse riding, Camping, Porter services, canoeing, and the arts and crafts which are made by the local people. The people in the Mnweni area make a living by selling their crafts to tourists that visit the area (Plate 3.15).



Plate 3.12 Horse riding (Sycholt, 2003)



Plate 3.13 Canoeing (Sycholt, 2003)



Plate 3.14 Camping site (Bristow, 2003)



Plate 3.15 Arts and Crafts (Sycholt, 2003)

3.3.9. Conclusion

This chapter provides the reader with an understanding and also a description of both the study areas. The study areas are completely different from each other as one is an estuary and the other is a river system. Diatoms are used to reflect the socio-economic impacts, biological and environmental impacts that are occurring in both the systems.

CHAPTER FOUR

4. MATERIALS AND METHODS

4.1. Introduction

The need to sample diatoms in estuaries and rivers arises from the necessity to know the water quality in terms of the South African National Water Act 36 (1998). When benthic diatoms are collected from riverine sediments and from estuaries, the selection of the precise site is either specific or random (Bate *et al.*, 2004). This chapter provides the reader with an overview of the methodology that was used to carry out this research.

4.2. DIATOM COLLECTION AND PROCESSING

Sampling of diatoms involved obtaining a segment of the upper 10 mm of the estuary and riverbed. A simple specially devised hand held and operated corer was used. This consisted of a 20mm diameter Perspex pipe of 2m in length. Built into the pipe was a simple plunger, which aided extraction of the sediment. One end of the pipe is inserted into the sediment to a depth of a few centimetres. By cupping ones palm over the other end suction is created within the pipe, enabling it to be removed with the sediment in place. The pipe is then lifted with the sediment core held in place by suction. Upon extraction, the stopper is removed and the sediment core is removed by gently depressing the plunger (Bate *et al.*, 2004).

Upon removal of the sediment a knife was used to cut out the top 10mm of the sediment. The sediment is then placed into a sterilized plastic bottle; water from the sampling site is then added to cover the sample. This procedure was replicated thrice in different positions at the same site in order to get a sample that is representative of the different micro- habitats at the

site. On each bottle a label is placed, so that the site number, sample number and the GPS readings can be written on.

4.2.1. Laboratory sample processing

The sediment samples were then taken to the sampling camp station where they were carefully placed in 90mm diameter plastic petri dishes and allowed to settle overnight. The following morning most of the supernatant was drawn off and 8 new cover slips (covering 90% of the sediment surface) were placed on top of the wet sediment. Two hours later the cover slips were carefully removed with as little sediment as possible. In this way only living cells that were attached to the cover slips were sampled (Bate *et al.*, 2004). The 5 cover slips from each sample were placed in glass bottles and transported to the laboratory at C.S.I.R.

The cover slips were kept in glass bottles to which 2 ml of KMnO_4 (saturated) and 2 ml of HCl (10 M) was added. All acid cleaned samples were washed with distilled water using 5 consecutive spins (2000 rpm for 10 minutes). Permanent light microscopy slides were made with 2 drops of diatom “digest”, placed onto an acid-washed, ethanol stored cover slip and allowed to air dry (Bate *et al.*, 2004).

Cover slips treated and stored in this manner allow the drop of sample to spread more evenly (Bate *et al.*, 2004). When completely dry, a small amount of DPX mounting medium (Saarchem-Holpro Analytical Supplies, S.A.) was dotted onto a glass microscopy slide and the cover slip placed over it. The DPX mountant was allowed to dry for 2 days and each slide was eventually sealed around the edge of the cover slip with Bioseal^R to prevent ageing of the mountant. The slides were logged and stored in the Choinoky Collection Slide Library, C.S.I.R., to form a permanent record.

4.2.2. Diatom identification and enumeration

Diatom frustules were examined under an Olympus Bx41TF light microscope with Differential Interference Contrast (DIC) optics and attached with a Sony Hyper HAD Digital video camera (SSC-DC18P). Images of the species enumerated were visualised and captured using the AnalySIS image analysis programme (© 2001, Soft Imaging System GmbH). From each sample 200 diatom valves were counted using 1000X magnification). Identifications were facilitated largely by taxonomic work done by Bate *et al.*, (2004) and a variety of diatom classification nomenclatures and literature including Schoeman and Archibald (1976); Hustedt, (1976); Archibald, (1983); Krammer & Lange-Bertalot, (1986-1991) and Round *et al.*, (1990).

4.2.3. Statistical data analysis

A variety of formal statistical analyses were used to investigate patterns in community structure. These comprised both univariate and multivariate techniques. According to Clarke and Warwick (1994), diversity indices, which effectively collapses full sets of species count in samples into single coefficients, can be validly treated using univariate statistical methods. To compare the diversity indices amongst different reaches or seasons, the Kruskal-Wallis procedure was used to conduct analysis of variance by ranks. When the null hypothesis of no difference was rejected at a probability $P < 0.05$, differences of ranks were compared using pair-wise multiple comparisons procedure (Tukey's test). The following diversity indices were considered:

- S = total number of species recorded per sample
- N = total density of diatoms recorded per sample
- d = Margalef's species richness
 $d = (S-1)/\log N$
- H' = Shannon-Wiener diversity

$H' = -\sum_i p_i(\log p_i)$ where p_i is the proportion of the total sample density arising from the i th species

➤ $J' = \text{Pielou's evenness index}$

$J' = H'_{\text{observed}} / H'_{\text{max}}$ where H'_{max} is the maximum possible diversity which would be achieved if all species were equally abundant

Multivariate analyses were conducted using the computer software package PRIMER v5 (developed at Plymouth Marine Laboratory, UK) and included techniques of clustering (hierarchical group average linkage), ordination (non-metric Multi-Dimensional Scaling [MDS] and k-dominance curves (ranked species abundance plots). The adequacy of an MDS in representing similarities amongst samples in a low dimension ordination plot can be measured by the extent to which the ordination preserves the rank order of dissimilarities. A stress value can be determined and gauged against benchmarks (Clark and Warwick, 1994), as indicated below:

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

Tests for differences in structure and composition of assemblages were performed using analysis of similarities (2-way crossed ANOSIM). The ANOSIM test statistic (R) is defined (Clarke and Warwick) so that:

- $-1 \leq R \leq 1$
- $R = 1$ only if all replicates within sites are more similar to one another than any replicates from different sites.

R approaches zero if the null hypothesis is true, so that similarities between and within sites will be the same on average

CHAPTER FIVE

5. RESULTS AND DISCUSSION

5.1. Introduction

The chemistry of unpolluted freshwater systems such as rivers is primarily controlled by the lithology of the drainage basin (Meybeck, 1987; Dupre *et al.*, 2003). Changing land use patterns in southern Africa together with climate change will almost certainly impact on the chemical weathering processes and the erosion rates, which will result in important implications for river, suspended and dissolved loads (Legesse *et al.*, 2003). As stated by Keulder (1979), southern Africa and the high relief and erodable sedimentary layers of the Drakensberg Escarpment are extremely susceptible to such change. Its impacts on the quality of freshwater, our most valuable resource, can only be assessed if the fundamental controls of the river system are understood (Day & King, 1995; Schaler and Baird, 2003). River water runoff provides most of the freshwater that is used for human, agricultural and also industrial utilisation in KwaZulu- Natal, and the results are therefore also of direct relevance to the assessment of water quality for these purposes (DWAF, 1996; Lin *et al.*, 2004).

There is a great need for good quality water in South Africa. In many areas as the population grows, consumption is beginning to exceed supply, with industrial and agricultural requirements increasing in proportion. Most of the rivers in South Africa have been modified, primarily by dams and weirs, in order to increase their year round ability to supply for agriculture, industry, municipal and human purposes (Bates *et al.*, 2004). As a result many modifications have led to a reduction of the water quality within the rivers because of the return flow from irrigated agricultural lands (Bates *et al.*, 2004). Due to the agricultural activities, erosion and high sediment yields has become a problem and has increased the already naturally high turbidity of many rivers.

5.1.1. The use of diatoms as an indicator of water quality of the Mnweni river catchment

Diatoms are a large and diverse group of single celled algae, and are distributed throughout the world in nearly all types of aquatic systems and are one of the most important food resources in freshwater ecosystems (Stoermer & Smol, 1999). Some of the factors which are often found to be important for the distribution of benthic river diatoms are water chemistry, substrata, current velocity, light and grazing (Patrick & Reimer, 1966; Round, 1981; Stevenson *et al.*, 1996). Most of these factors depend strongly on the climate, geology, landuse and other landscape characteristics, and therefore are similar within ecological regions defined by these characteristics (Stevenson & Pan, 1999).

The use of benthic diatoms (microphytobenthos) in South African river systems has been studied extensively since the early 1950's (e.g. Cholonky, 1953; Cholonky, 1960; Cholonky, 1968; Archibald, 1983; van der Molen, 2000; Bate *et al.*, 2004). Efforts have been made to relate diatom associations to water quality (e.g. Archibald, 1983; Schoeman, 1979, Schoeman and Archibald, 1986 as above comment.). The distribution of microphytobenthos in a river system is a result of the complex series of interactions between the water quality, hydrological and biotic interactions. Short term differences in the community composition are driven by the immigration of cells, the differences in the growth rate between population and loss processes such as death, grazing and emigration (Bates *et al.*, 2004). Benthic microalgae become abundant where the water systems are impacted by anthropogenic influences.

This study will attempt to serve as a tool in order to understand the changes that are occurring in the microphytobenthos diatom community in the Mnweni river system. In this chapter we look at the river system, by using bioindicators as a water quality indicator. The diatom community will enable us or provide us with information about the different impacts, which are impacting, on the

Mnweni river system and the levels of the impacts at each site that was sampled.

5.1.2. Results

In this study a total of 17 diatom species, which belong within the 7 genera, were found in the Mnweni River system (refer to Figure 5.1.). Species belonging to the genera *Cymbella* (namely *Cymbella turgidula*) was found to be present at all sites [it was the most abundant in site 1(105 diatoms), site 2 (44 diatoms), site 3 (41 diatoms) and was followed by the Genera *Navicula* (namely species *dehissa* which was found in abundance at sites 3(102 diatoms) and 4 (68 diatoms). Species *Navicula decussis* which was abundant in sites 3 and 4 [comprising of 55 diatoms and 25 diatoms]. The diatom *Navicula salinarum* was found to be only present in site 4 with an abundance of 45 diatoms. Species belonging to the genera *Hantzschia*, *Fragilaria* and *Diploneis* were also found at the sampling sites but in smaller percentages as compared to the other genera.

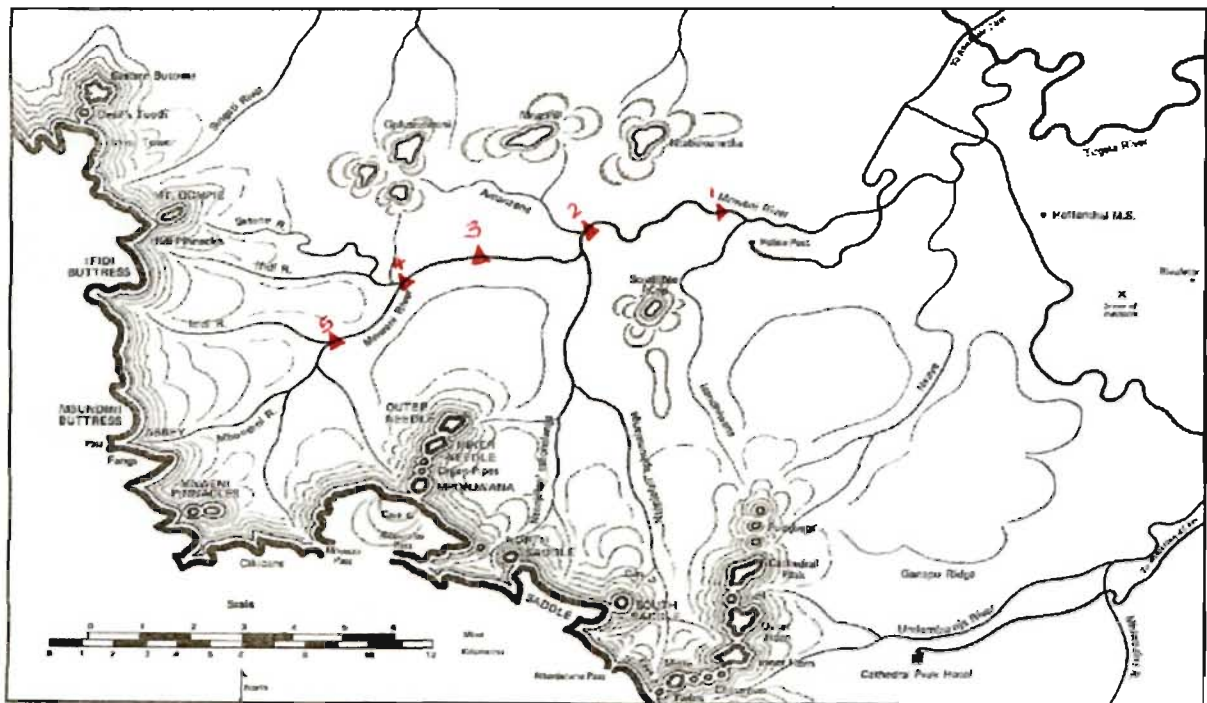


Figure 5.1 Map of Mnweni River system depicting sampling points 1- 5

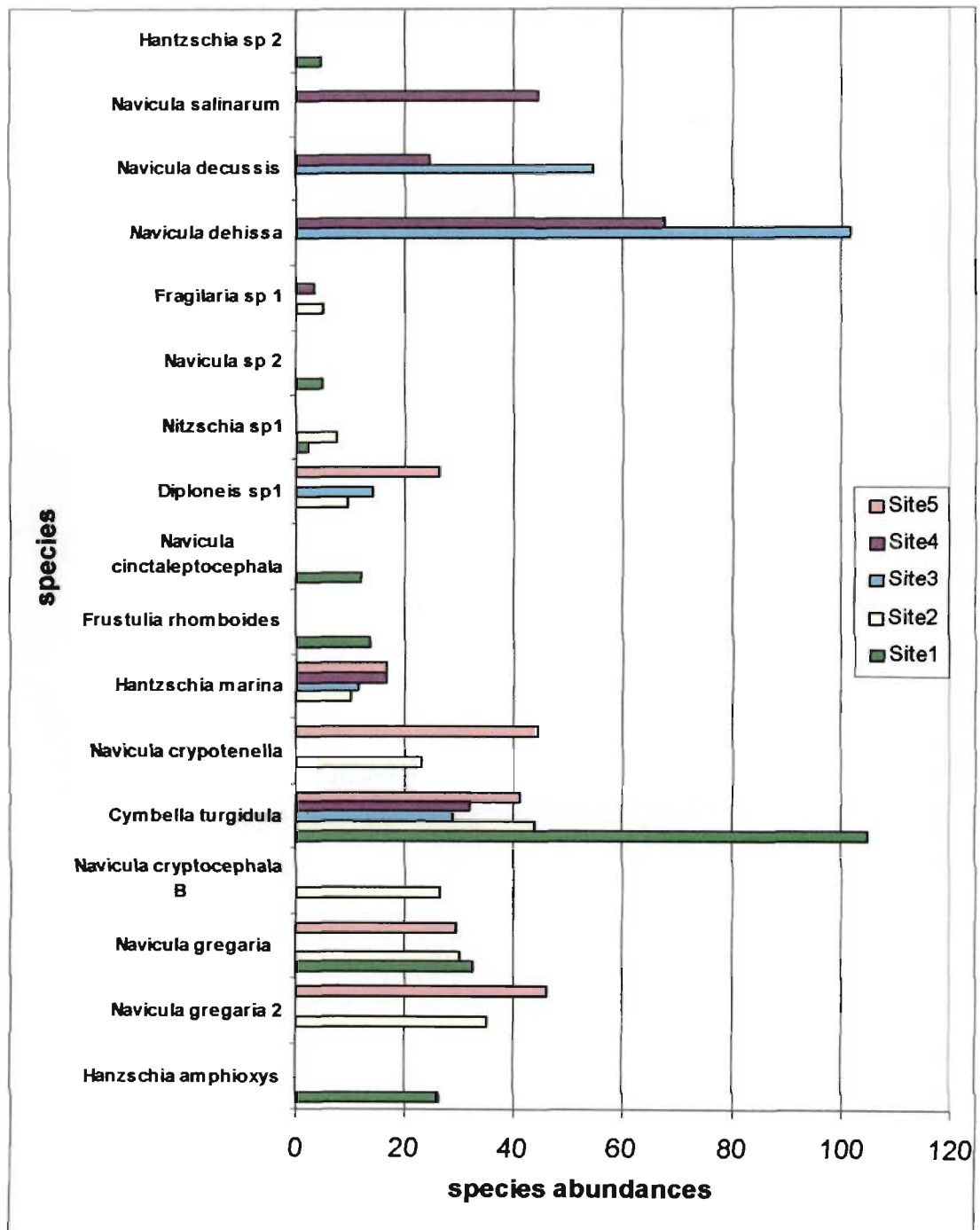


Figure 5.2 Graph depicting species abundance numbers among five sampling sites

The Multi Dimensional Scaling (MDS) (3D stress = 0.04), refer to Figure 5.3. Which shows that the similarity that can be seen from the assemblage data displays a good indication of the dissimilarity of species that are in the river. A SIMPER analysis of the data showed an average dissimilarity between the sites. In sites 1 and 2, the dissimilarity was 65%. At site 1 *Cymbella turgidula* was found to have the greatest abundance of 105 diatoms in the river system, which was followed by *Navicula gregaria* 33 diatoms and *Hantzschia amphioxys* 26 diatoms. Species belonging to the Genera *Frustulia*, *Navicula* and *Nitzschia* were found in small abundances at this site. In sample site 2 *Cymbella turgidula* has an abundance of 44 diatoms as compared to the abundance found at site 1. The Genera *Navicula* was found in abundance [comprising of species *gregaria* 2 (35 diatoms), *gregaria* (30 diatoms), *cryptocephala* B (27 diatoms) and *cryptotenella* (23 diatoms). The Genera *Hantzschia*, *Diploneis*, *Nitzschia* and *Fragilaria* were also found at site 2 but in smaller abundances.

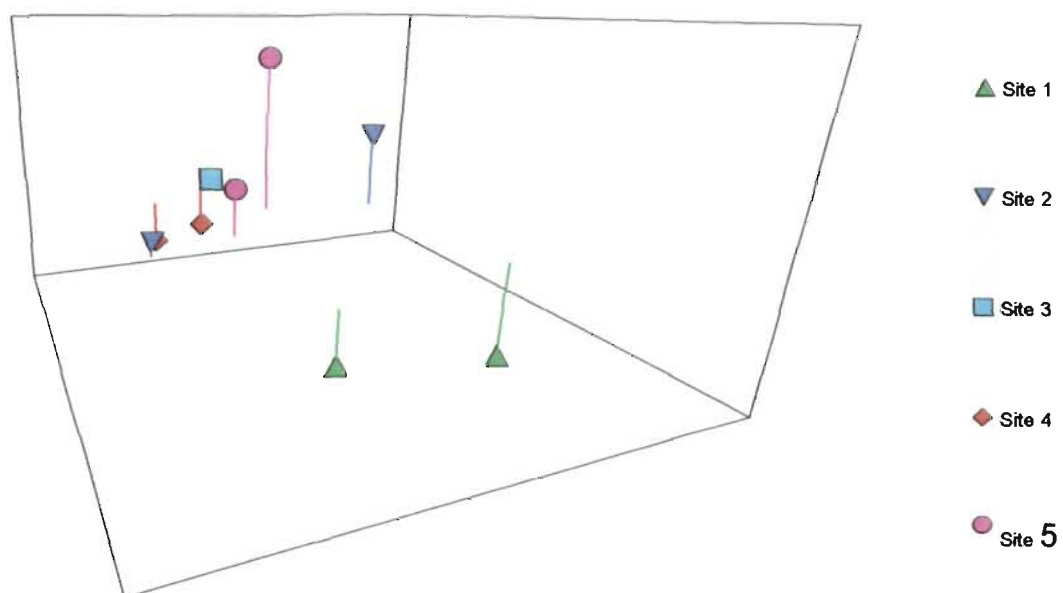


Figure 5.3 MDS Plot of Mnweni River Catchment System.

The average dissimilarity between sites 1 and 3 is 77%. In site 3, *Navicula dehissa* shows the greatest abundance of 102 diatoms, which is then followed by *Navicula decussis* with an abundance of 55 diatoms and *Cymbella turgidula* with an abundance of 29 diatoms. The Genera *Hantzschia* and *Diploneis* comprise of an abundance of 14 diatoms and 12 diatoms at the site. Site 4 shows an average dissimilarity of 76%. At a site the dominant specie is *Navicula dehissa*, which comprises of 68 diatoms, *Navicula salinarum* has an abundance of 45 diatoms. The specie *Cymbella turgidula* has an abundance of 32 diatoms and *Navicula decussis* has an abundance of 25 diatoms. The Genera *Hantzschia* and *Fragilaria* are present at this sampling site but in small abundances [*Hantzschia marina* (17 diatoms) and *Fragilaria* SP 1. (4 diatoms)].

Site 5, which was the last site that was sampled, has an average dissimilarity of 71%. The specie *Navicula gregaria* 2 shows the greatest abundance at this site of 46 diatoms and *Navicula cryptotenella* also indicates a great abundance at this site with 45 diatoms. The specie *Cymbella turgidula* was also found in abundance of 41 diatoms at this site. Species belonging to the Genera *Navicula*, *Diploneis* and *Hantzschia* were also found at this site [comprising of *Navicula gregaria* 30 diatoms, *Diploneis* 27 diatoms and *Hantzschia marina* 17 diatoms].

The species *Cymbella turgidula* also indicates a great abundance of 41 diatoms at this group. *Cymbella turgidula* which was found throughout the sites is a clear indication that the Mnweni River system is a fresh water system and that the system (Bill Harding, pers. comm., 2006)

5.1.3. Physical and Chemical Parameters

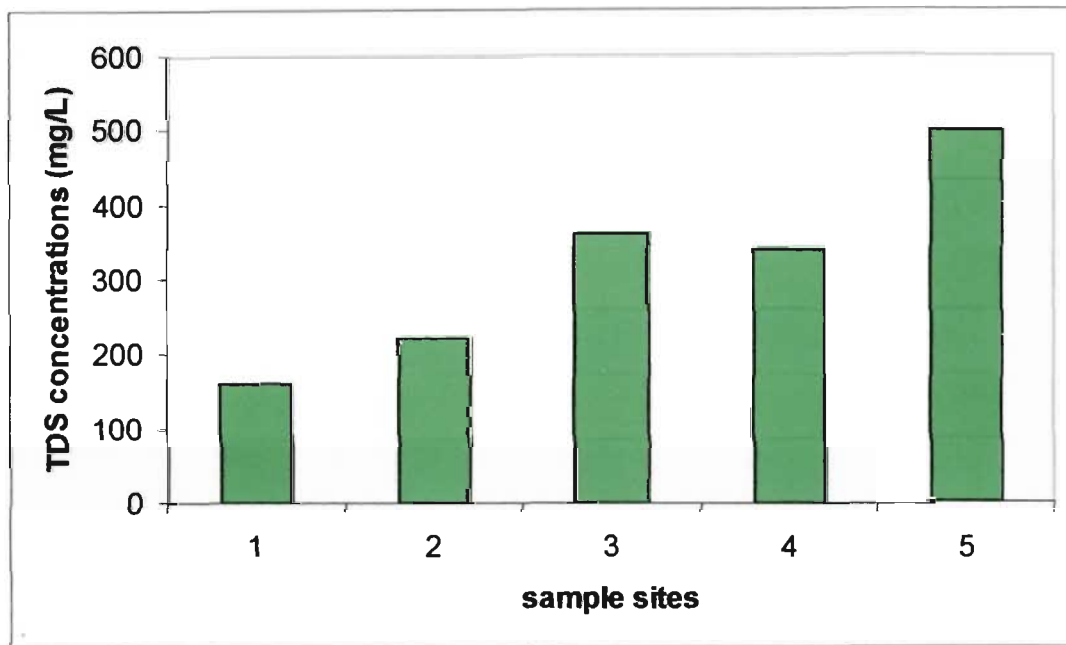


Figure 5.4 TDS concentrations at sample sites

As seen in figure 5.4, the TDS concentrations at sites 2 and 4 display the highest values [comprising of 500mg/L at site 2 and 450mg/L at site 4]. Site 5 has the lowest TDS concentration of 200mg/L as compared to all the sites that were sampled. Site 1 has a concentration value of 360mg/L and site 3 has a value of 340mg/L. As seen in the graph, the TDS value increases at site 2, and then decreases at site 3, then at site 4 the value increases and then falls once again at site 5. There is a general progressive pattern in the TDS concentration in the downstream direction.

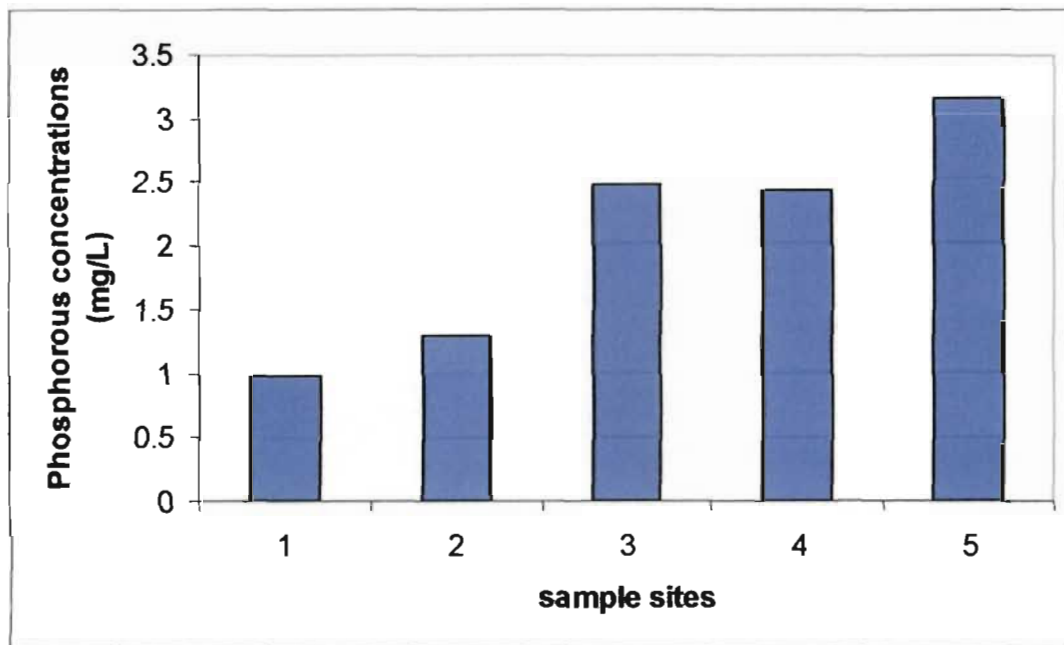


Figure 5.5 Phosphorus levels among sites

It can be seen that sample sites 3, 4 and 5 show relatively high phosphorus concentrations as compared to sample sites 1 and 2. The graph, shows that sample sites 1 (0.979mg/L) and 2 (1.2975mg/L) has the lowest phosphorous level respectively, which is then followed by site 4 with a level of 2.438mg/L. Sampling sites 3 and 5 has the highest phosphorous values comprising of 2.475mg/L at site 3 and 3.145mg/L at site 5 as compared to the other three sites that were sampled. Once again, it is clear that phosphorous concentrations increase in the downstream direction as impacts due to increased landuse become apparent.

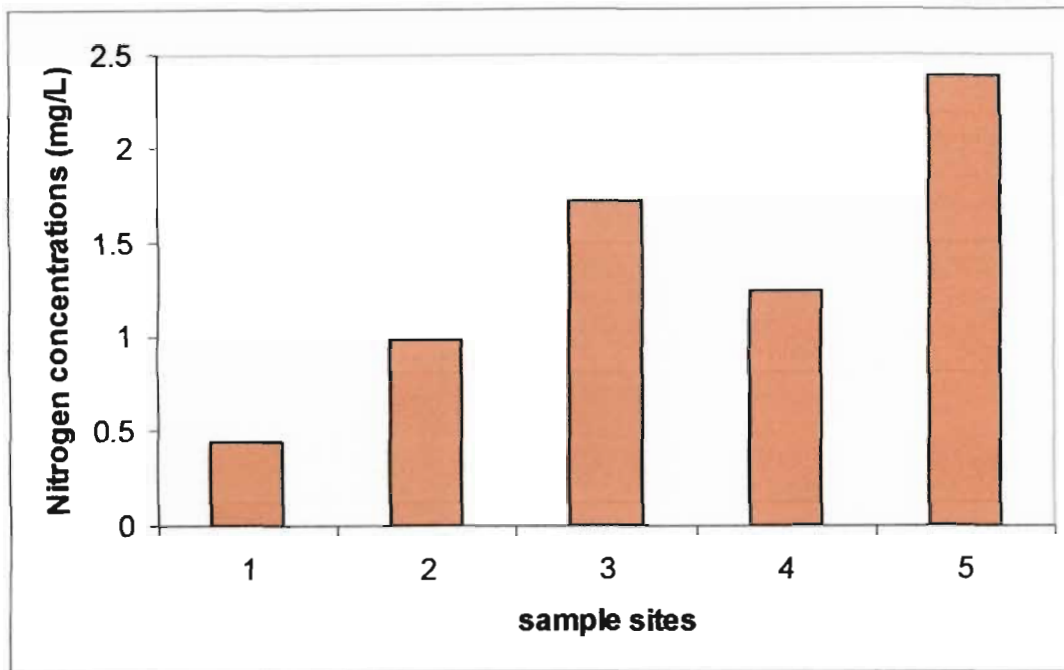


Figure 5.6: Nitrogen concentrations along sites

In the following graph sample sites 3 and 5 depicts the highest nitrogen levels and site 1 has the lowest value. It can be seen from the graph that site five has the highest value of 2.375mg/L, which is then followed by sample site 3 with a value of 1.721mg/L. Sample site 4 displays a value of 1.238mg/L and site 2 and 1 has the lowest values [comprising of site 2 (0.983mg/L) and site 1 (0.438mg/L)].

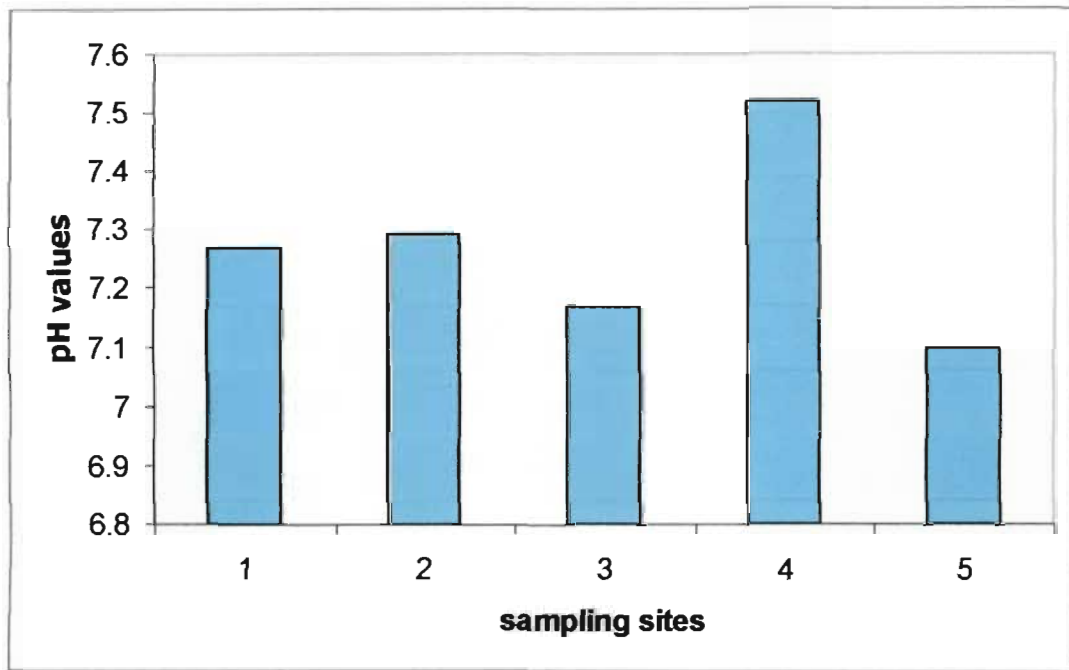


Figure 5.7 Graph depicting pH values at the various sites sampled

In the following graph sampling sites 1, 2, 3 and 4, displays a high pH value as compared to site 5, which has the lowest. From the graph it can be seen that site 1 has a pH value of 7.27, which then increases to 7.29 at site 2. Sampling site 3 then shows a decrease in the pH value at this site with 7.17, and then increases at site 4 with a pH value of 7.52. Sample site 5 has the lowest pH value of 7.1. Sites with higher TDS, Phosphorous and Nitrogen tend to have slightly suppressed pH values although all the Ph values are above pH 7.

5.1.4. Biological Parameters- Diatom diversity (H') and evenness (E) and richness (D) species indices

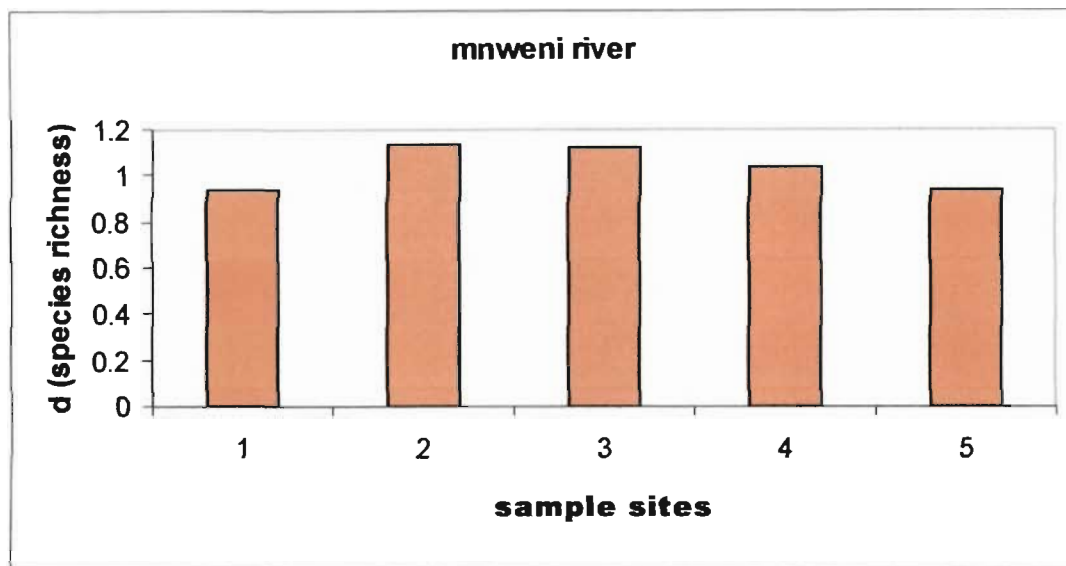


Figure 5.8 Graph indicating the differences of species richness between the five sampling sites

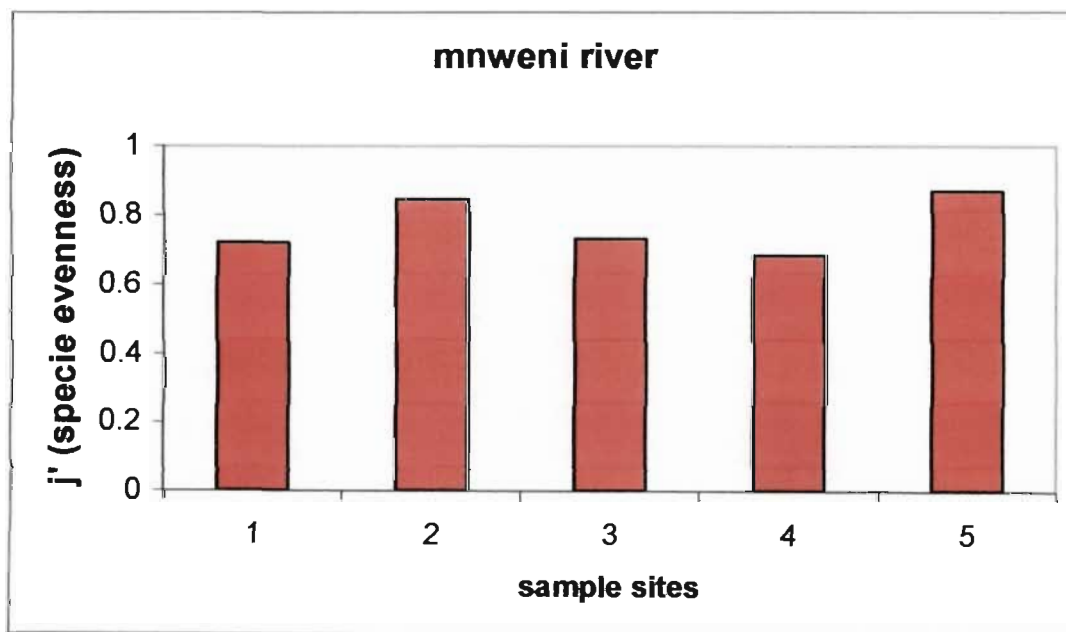


Figure 5.9 Species evenness graph indicating the diatom differences between the five sites that were sampled

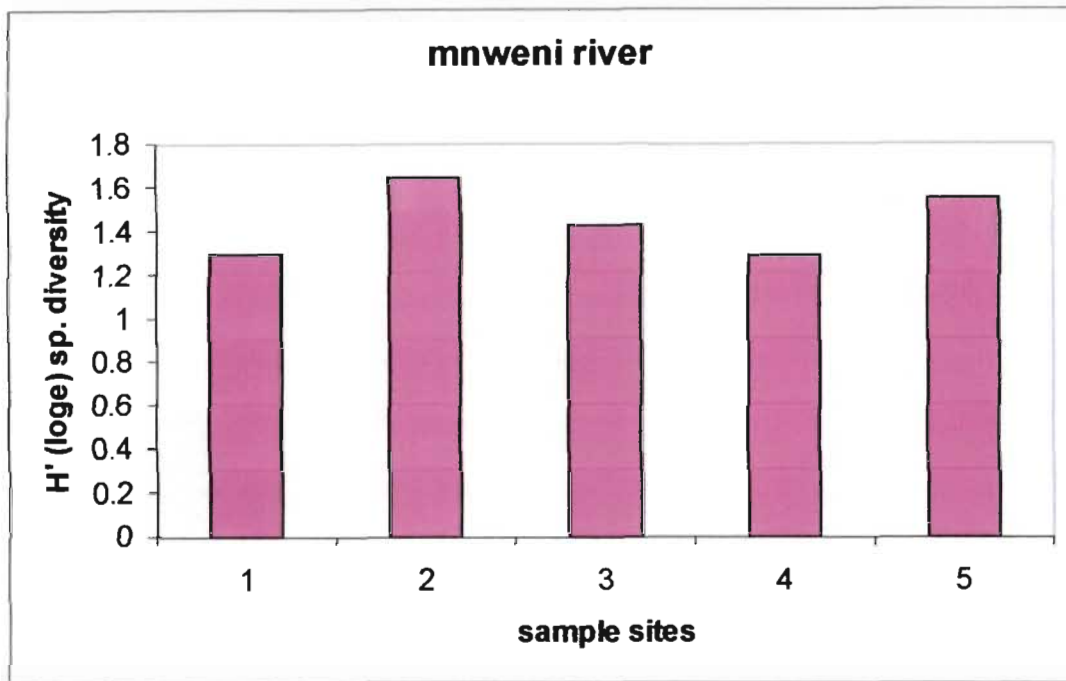


Figure 5.10 Species diversity graph indicating the diversity of diatom species along the sites

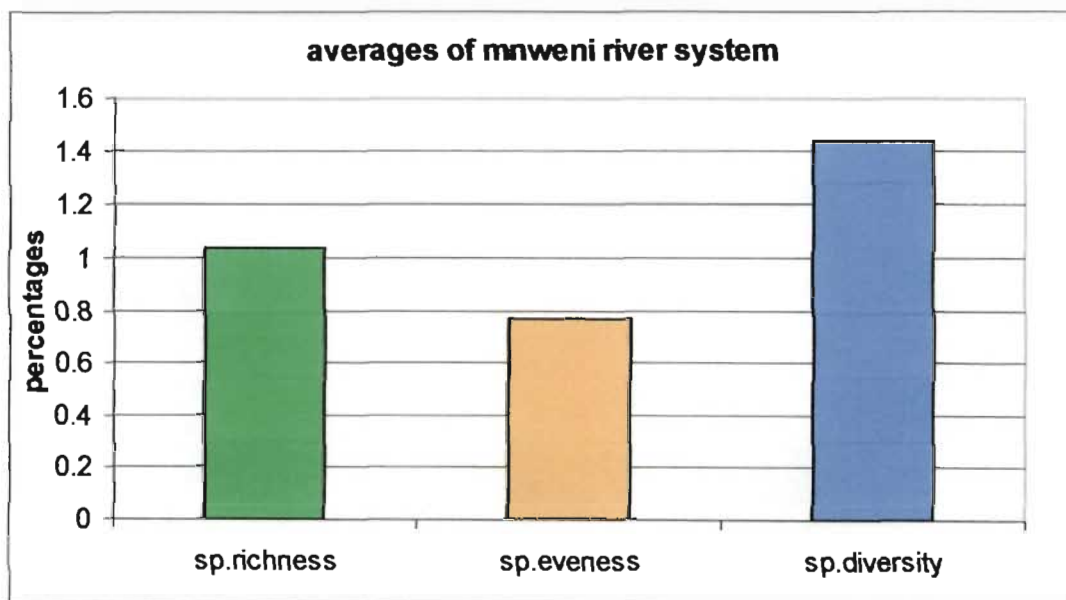


Figure 5.11 Graph depicting the differences in percentages between species richness, species evenness and species diversity

The species diversity between the sampling sites was relatively high and varied from 1.2 – 1.5 respectively (refer to Figure 5.11.). Sample site 2 showed the highest species diversity as compared to the other four sites. The species richness at the sites 2, 3 and 4 shows an increase in richness as at these sites whereas sites 1 and 5 showed a low level of species richness (refer to Figure 5.8.). Species evenness between the sites were low and varied between 0.6 – 0.8 (refer to Figure 5.9.). There were marked variations in the richness and diversity between the sites that reflected variations in the water chemistry and the habitat character. In particular, richness and diversity declined at the sites with slower water flow, pebble, gravel and silt substrata.

5.1.5.DISCUSSION

Total dissolved solids (TDS) are a common measurement of freshwaters. When TDS is determined by summing the results of separate analyses for all major ions, it is analogous to salinity (Day, 1990). When TDS is measured gravimetrically (by weight), it can be greater or less than salinity, depending on whether loss of bicarbonate (H_2CO_3) in the gravimetric analysis is more than offset by the presence and, consequently, measurement of dissolved organic carbon (Day, 1990).

Human activities have severely increased the TDS concentrations of inland waters worldwide (Hart *et al.*, 1991). As seen in Figure 5.4, sites 3, 4 and 5 shows an increase in TDS levels as compared to sites 1 and 2. The increase in the TDS levels may be caused by irrigation and greater turbidity caused by humans and animals. Irrigation causes salinization of rivers in two ways. Firstly, although the irrigation water may be low in TDS, the water itself is taken up by crops or even evaporates, resulting in solutes being left behind. Ion exchange processes in the soil, results in the accumulation of NaCl, which is washed out of the soil into rivers. Secondly, irrigation may result in a rise in the water table and subsequent evaporation from the surface of the now wet soil (Hart *et al.*, 1991). The reuse of water, whether it is recycled for immediate water consumption or returned to a stream and subsequently

reused further downstream, will result in partial evaporation during the cleansing process and will consequently lead to increased TDS levels as stated by (du Plessis & van Veelen, 1991).

In general, it seems that many species are able to survive and even flourish at relatively high salinities. It has been concluded by (Hart *et al.*, 1991) that many freshwater blue greens and bacteria can adapt readily to TDS values. As shown by Prinsloo & Pieterse, (1994) that an increase in the TDS concentrations in the middle of the Vaal River has been accompanied by decreases in turbidity. They have also shown that production in green algae, *Monoraphidium circinale*, increased in TDS values between 500 and 2500mg l⁻¹, while that of the diatom *Cyclotella meneghiniana* and the blue green *Microcystis aeruginosa* were inhibited at these concentrations.

The high concentrations of phosphorus are due to leaching or runoff from agricultural land. As seen in Figure 5.5, sampling sites 3, 4 and 5 displays the highest phosphorous values as compared to the rest of the sites that were sampled. Due to the increase of phosphorous levels at these sites, there is an increase of diatom biomass.

Phosphorous plays a major role in the structure of nucleic acids and in molecules, which are involved in the use, and storage of energy in cells (Addiscott *et al.*, 1991). The role of the processes and mechanisms that control the supply of bioavailable phosphate is essential in the management of catchments, rivers and lakes in order to avoid eutrophication (Webster *et al.*, 2001).

Nitrogen occurs abundantly in nature and is essential in the constituent of proteins, which include enzymes that catalyse all biochemical processes, and are therefore a major component of all living organisms. The relatively high concentrations of nitrogen at sample sites 3 and 5 results in increases in electrical conductivity (Figure 5.6). The high levels of these nutrients also add

to the high TDS levels at certain sampling sites. This nutrient enrichment is restrictive on the healthy growth of all biota in the system.

PH, which is determined largely by the concentrations of ions (H^+) and alkalinity by the concentrations of hydroxyl, bicarbonate and carbonate ions in water. As stated by Dallas & Day (2004), that in very pure waters the pH can change rapidly because the rate of change is determined by the buffering capacity, which in turn is usually determined by the concentration of bicarbonate and carbonate ions in the water. The pH of the natural water is determined by geological and atmospheric influences (Dallas & Day, 2004). As seen in the graph depicting the pH ranges along the sites (refer to Figure 5.7.), sample sites 1 – 4 shows a high pH value as compared to sample site 5, which has the lowest pH value.

The river chemistry and habitat character differed between sites, which had shown agricultural land use, and human use of the river system. The concentrations of nitrogen at the sampling sites 3 and 5 was relatively high, probably because of the use of fertilizers, as well as higher erosion rates and increased weathering and evaporation (Jenkins, Sloan & Cosby, 1995; Collins & Jenkins, 1996). It was observed during sampling that due to an increase of agriculture in the Mnweni area, it had also led to the degradation or removal of vegetation along the riverbanks.

Environmental gradients and assemblage composition in the Mnweni River catchment revealed that water chemistry, habitat structure, flow type and land use were the most important environmental factors for diatom assemblage composition in the Mnweni River. The chemical components reflected changes because of enrichment by agriculture as well as changes in the geology along the watercourse as at the top of the river system volcanic rocks were present and towards the end of the sampling points sedimentary rocks and boulders were present. Diatom assemblages differed between sites in agricultural catchments and sites that may have been affected by sewage or human waste from the settlements. In a study on benthic algae in an

agriculturally dominated landscape, Munn, Black & Gruber (2002) identified that conductivity, nutrient concentrations, catchment land use and water flow as key environmental variables, but had failed to detect and effects on the substratum type on assemblage composition. Water chemistry gradients was an important component in this study, but changes in the flow character, anthropogenic modification or influences on the river bank and catchment land use was also significant with the changes in the assemblages composition between the sites.

5.1.6 Conclusion

The results indicates that the diatom diversity and assemblage composition in the Mnweni River shows that changes in the water chemistry as well as organic pollution, but also indicate changes in the habitat character which is related to the water flow, river bank character and catchment land use. The diatom assemblages that were found at the sites are typical of clean or mildly enriched water conditions.

Demands on the water resources and freshwater habitat changes in the Mnweni River catchment are likely to increase in the future as the population grows, which will result in the expansion of settlements and also an increase of fertilizers for the crops and livestock.

Types of diatoms found in the Mnweni river system

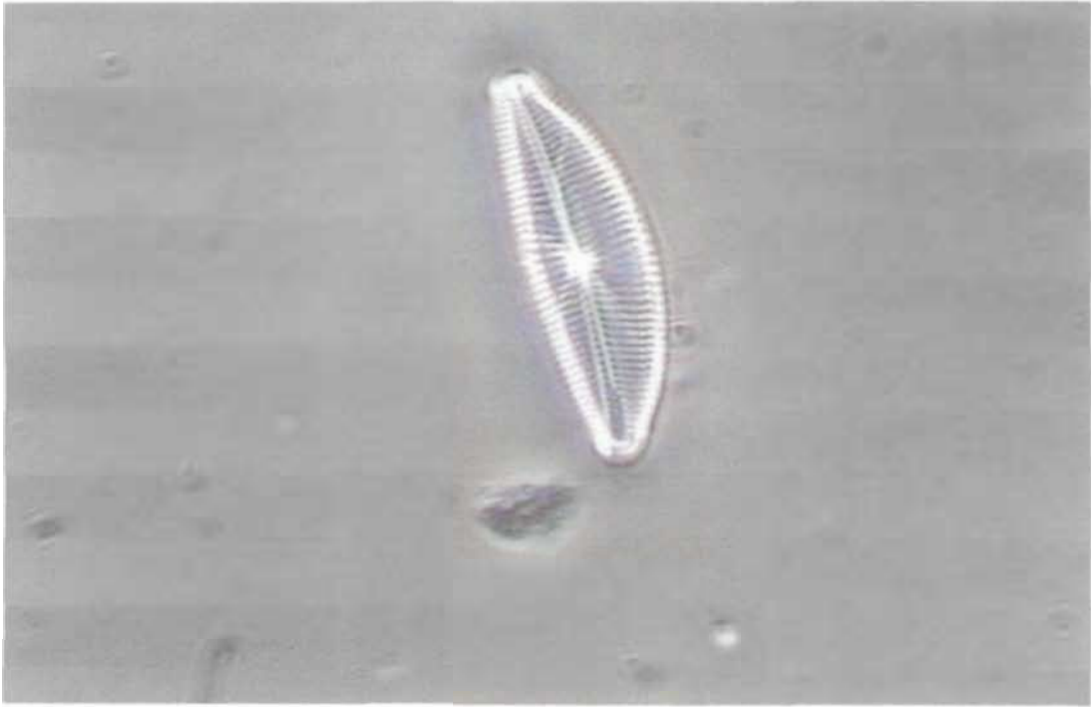


Plate 5.1: ***Cymbella turgidulla***



Plate 5.2: ***Cymbella turgidulla*** and ***Navicula salinarum***



Plate 5.3: *Navicula decussis*

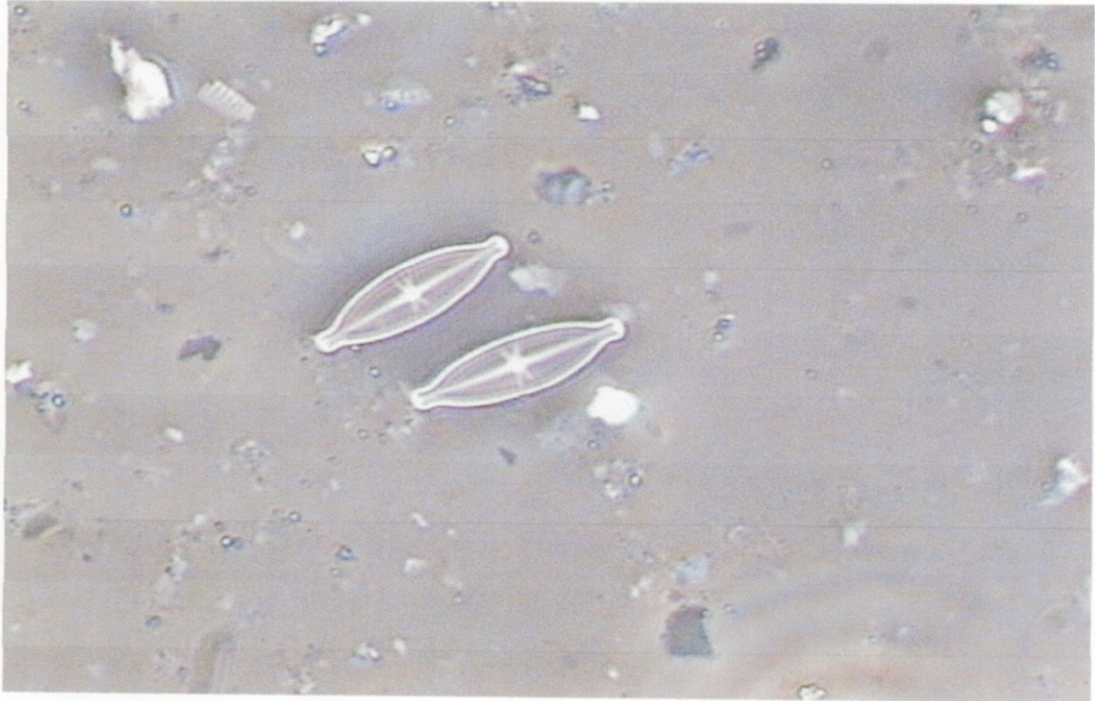


Plate 5.4: *Navicula gregaria* 2



Plate 5.5: ***Navicula dehissa***



Plate 5.6: ***Hantzschia marina***

5.2. A survey on the St. Lucia diatom community composition

5.2.1. Introduction

There are approximately 250 functional estuaries in South Africa (Whitfield, 2000), which together makes up our countries most productive habitat and many of which are considered unique in terms of their biodiversity and physical characteristics. These estuaries perform several important ecological and economic functions, not least of which is their aesthetic and recreational value (Water Wheel, May/June 2005). Estuaries are well known for their biodiversity, productive fish and invertebrate fisheries and the important nursery functions that they perform, such as providing nursery areas for marine fish and staging and feeding sites for significant migratory birds (Skelton, 1993; Turpie, 1995). Estuaries also support a number of endemic species, many of which depend on these habitats for their survival. However, estuaries constitute one of the most threatened habitats in the country (Turpie *et al.*, 2000). Despite existing laws that govern activities in and around estuaries, the overall protection of these valuable assets remains low. In a recently published assessment on South Africa's national biodiversity, it shows that only 28% of the countries estuaries are still in an excellent condition. A further 31% are in good condition, 25% in fair condition, with the remaining 15% of the countries estuaries being in poor condition as they suffer from major ecological degradation (Water Wheel, 2005). In many cases freshwater inflows, which are vital to the maintenance of salinity profiles, nutrient supply and sediment scouring have been polluted or siphoned off. Due to a result of all these pressures, many South African estuaries have become functionally degraded, and have led to the loss of certain species or a reduction in population (Love, 2000; Wooldridge, 1999).

The need to sample diatoms in estuaries arises from the necessity to know the history of water quality in terms of the South African National Water Act

36 (1998), whereby adequate freshwater must be supplied to estuaries in order to maintain their ecological integrity (Bate *et al.*, 2005). Biological communities reflect the overall ecological integrity by integrating various stressors, thus providing a broad measure of their synergistic impacts (Taylor, 2004). Aquatic communities, both animals and plants, reflect and integrate the effects of physical and chemical disturbances that occur or have occurred over a period of time (Chutter, 1998). These communities can provide an integrated and holistic measure of integrity or health of a system. Biomonitoring has only as recently as 1996 become a routine tool in the management of South Africa's water systems, although methods have been available for many years (Hohls, 1996). Benthic macro- invertebrates are recognized as valuable organisms for bioassessments, due largely to their visibility to the naked eye, ease of identification, rapid life cycle which is often based on seasons and their large sedentary habitats (Dickens & Graham, 2002). Diatom communities provide interpretable indications of specific changes in water quality (Kwandrans *et al.*, 1998).

Biological monitoring techniques have been introduced as part of routine monitoring programmes due to certain shortcomings in standard physical and chemical methods (Rey *et al.*, 2004). Due to the difficulty and the cost to chemically analyse every potential pollutant in a sample of water and to interpret it in terms of impact severity, it makes sense to monitor aquatic biota (Rey *et al.*, 2004). Biological monitoring are cost effective and the results can be obtained rapidly. The main advantage of using a biological approach is that it examines organisms whose exposure to pollutants is continuous (Eekhout *et al.*, 1996).

Benthic fauna are considered important indicators of water quality and are used in a variety of monitoring programs to assess overall estuarine health and to follow long trend in estuarine communities, especially related to anthropogenic impacts (Boesch *et al.*, 1976, Aschan & Skullerod 1990, Simboura *et al.*, 1995, Hyland *et al.*, 1996). Diatoms offer three positive features from a monitoring perspective. These include: 1) occupy an

important intermediate trophic level, 2) are relatively long-lived and sedentary, and 3) respond differentially to varying environmental conditions. After settlement, most diatoms remain within a relatively constrained area, for their entire adult lives (Posey & Alphin, 2000). These organisms unlike many other biotic or chemical measures, are able to adapt to conditions at a specific location. By examining shifts in the diatom community over time, one can gain an understanding of the major environmental processes affecting the local biota (Hyland *et al.*, 1996).

Estuaries represent one of the few sheltered environments along a rugged coastline, which is dominated by high wave energy. Therefore, estuaries have naturally become the focus of coastal development for either commercial or industrial use such as harbours, ports or as recreational or residential areas (Morant & Quinn, 1999). The rapid growth and increase of the human population, urbanisation and industrial activities, will lead to an increase in the pressure to develop estuaries. However, as stated by Day & Grindley, 1981; the protection of the estuarine environment in the long term is essential if the natural resources that are provided by estuaries and the quality of the life offered by them are to be maintained.

The pristine scenery associated with natural estuaries, attracts many people and these people are not only visitors to the area but also the local residents who enjoy the aesthetic appeal of the natural area close to home. However, when an area has or displays such appeal people flock to it and abuse or over use it, thereby reducing the aesthetic qualities that had attracted them in the first place (Reimold *et al.*, 1980). Estuaries offer a number of socio- cultural values other than economic or ecological ones and these socio- cultural values are often those attributes, which contribute to the "quality of life" (Reimold *et al.*, 1980). Estuaries also serve as an important educational function to the general public, as the education of citizens is essential in order to develop an awareness of estuaries and to also ensure their conservation and long-term sustainability.

Portman & Wood, (1985) state that the aesthetic aspects of estuarine quality are an important factor and the appearance of an estuary contributes to its perceived environmental health, particularly in terms of its utilisation by humans.

This study will attempt to provide an understanding of the changes occurring in St. Lucia estuarine diatom community with regard to the microphytobenthos community. In this chapter we look at the estuarine system, by using diatoms which will serve as indicators in assessing the health status of the estuarine and in the same process the salinity gradient will be determined.

5.2.2. Results

The methodology that was implemented for this study is very similar to the one described on page 54. In this study, a total of 17 diatom species, which fall within 7 genera, were found in the St. Lucia estuarine system (refer to Figure 5.12 and Figure 5.13). Species belonging to the Genus *Nitzschia* (namely *Nitzschia pellucida*) was found to be abundant at site 3 (149 diatoms), site 4 (72 diatoms), site 5 (58 diatoms) and was followed by the species *Nitzschia scalpelloides* which belongs to the Genera *Nitzschia*. *Nitzschia scalpelloides* was found in abundances at site 4 (77 diatoms) and site 5 (48 diatoms). Species belonging to the Genera *Nitzschia* was also found in abundance [namely *sippen* (68 diatoms), *clausii* (60 diatoms), *scalpelliformis* (56 diatoms) and *aræmonica* (48 diatoms)]. The Genera *Frustulia* (namely *rhomboids*) was found to be present at sample site 2 (26 diatoms), site 3 (15 diatoms), site 4 (16 diatoms) and site 5 (20 diatoms). Species belonging to the Genera *Cymbella*, *Navicula*, *Hantzschia*, *Amphora* and *Aulacoseira*, were also found at the sampling sites but in smaller percentages as compared to the Genera *Nitzschia* and *Frustulia*.

diatoms)]. Species belonging to the Genera *Amphora* and *Navicula* were both found to present at sites 3, 4, 5 but in relatively small abundances. *Navicula aequorea* [site 3 (29 diatoms), site 4 (8 diatoms), site 5 (35 diatoms)] and *Amphora veneta* [site 3 (9 diatoms), site 4 (11 diatoms), site 5 (35 diatoms)]. Species belonging to the Genera [*Navicula* (namely *diserta* (21 diatoms), *Amphora* (namely *cfmontana* (15 diatoms) and *Cymbella* (namely *turgidula* (8 diatoms)]. The species *Amphora arcus* (11 diatoms) and *Aulacoseira* sp. 1 (7 diatoms) was only found at sample site 4 and species belonging to the Genera *Navicula* [namely *pseudohalophila* (16 diatoms) and sp. 1 (3 diatoms) was present at site 5 only.

Figure 5.15, this graph shows the dissimilarity between sampling sites 3 – 5. In this graph the species *pellucida* belonging to the Genera *Nitzschia* was found to be abundant at sites 3, 4, 5. *Nitzschia pellucida* was found in greatest abundance of 149 diatoms (site 3), 72 diatoms (site 4) and 58 diatoms (site 5). Species that were also found to be present at all 3 sites belong to the genera *Amphora* (namely *veneta*) and *Navicula* (namely *aequorea*). The species *Amphora veneta* [site 3 (9 diatoms), site 4 (11 diatoms), site 5 (35 diatoms)] and specie *Navicula aequorea* [site 3 (29 diatoms), site 4 (8 diatoms), site 5 (34 diatoms)]. The specie *scalpelloides* that belongs to the genera *Nitzschia* was found to be abundant at site 4 (77 diatoms) and site 5 (48 diatoms). Species which belong to the genera *Diploneis*, *Navicula* (*pseudohalophila*), *Navicula* sp 1, *Aulacoseira* sp 1, *Amphora arcus* and an unknown diatom was found in the estuarine system but in relatively small numbers.

In figure 5.16, the graph depicts the average dissimilarity between site 4 and site 5. At sites 4 and 5, the Genera *Nitzschia* (namely *Nitzschia scalpelloides* and *pellucida*) was found to be the most dominant diatom. *Nitzschia pellucida* has an abundance of 72 diatoms at site 4 and an abundance of 58 diatoms at site 5. *Nitzschia scalpelloides* has an abundance of 77 diatoms at site 4 and 48 diatoms at site 5. The Genera *Navicula* and *Amphora* was also found to be relatively abundant at the sites.

Navicula aequorea 8 diatoms at site 4, 34 diatoms at site 5 and *Amphora veneta* 11 diatoms at site 4 and 35 diatoms at site 5. Species belonging to the genera *Navicula*, *Amphora*, *Aulacoseira*, *Surirella* was also found at these sampling sites but in small abundances.

The Multi Dimensional Stress (MDS) (2D stress = 0.01), which shows that the similarity that can be seen from the assemblage data, displays a good indication of the dissimilarity of species that are found in the St. Lucia estuarine system (refer to Figure 5.17). Statistically it has been shown that there is a significant difference between the five sites that were sampled within the estuarine system as $R = 0.62$ (this is supported by Anosim). The MDS plot graph also shows that site 1 and 2 are different as $p \leq 0.01$, whereas sites 3, 4 and 5 on the graph shows no significant differences (refer to figure 5.17). Variation in species relative abundance was also examined by using the graphical representations of species cumulative frequency distributions (k-dominance curves, Lamshead et al., 1983; refer to figure 5.18.)

A SIMPER Analysis of the data showed an average dissimilarity between the sites. In sites 1 & 2, the average dissimilarity was 93%. At site 1 *Nitzschia clausii* was found to have the greatest abundance of (60 diatoms) in the estuarine system, which was followed, by *Nitzschia aremonica* (48 diatoms) and *Cymbella turgidula* (42 diatoms). Species, which belong to the genera *Hantzschia*, *Navicula* and *Amphora*, was found in small abundances at this site. In sample site 2, *Nitzschia sippen* has the greatest abundance of (68 diatoms), which was then followed by *Nitzschia scalpelliformis* with an abundance of (56 diatoms) and *Frustulia rhomboides* shows an abundance of (26 diatoms). Species belonging to the genera *Navicula*, *Cymbella* and *Amphora* was also found at this site but in small abundances.

The average dissimilarity between site 1 & 3 is 98%. In site 3, *Nitzschia pellucida* shows the greatest abundance of 149 diatoms, which is then followed by *Navicula aequorea* with an abundance of 29 diatoms and *Frustulia rhomboides* with an abundance of 15 diatoms. Species, which belong to, the genera *Amphora* and *Navicula* was also present at this site

but in small abundances. Sample site 4 shows an average dissimilarity of 100 diatoms. At this site the dominant specie is *Nitzschia scalpelloides*, which comprises of 77 diatoms and *Nitzschia pellucida* has an abundance of 72 diatoms. The specie *Frustulia rhomboides* has an abundance of 16 diatoms and species belonging to the genera *Amphora* (namely *veneta* and *arcus*) has the same abundance of 11 diatoms at this site. *Navicula aequorea* was also present at this site but in a small abundance of only 8 diatoms

Sample site 5, which was the last site that was sampled in the estuarine system, has an average dissimilarity of 96%. The specie *Nitzschia pellucida* shows the greatest abundance at this site with 58 diatoms and *Nitzschia scalpelloides* also indicates a great abundance of 48 diatoms at the sample site. *Amphora veneta* shows an abundance of 35 diatoms and the specie *Navicula aequorea* with an abundance of 34 diatoms. Species belonging to the Genera *Frustulia* and *Navicula* was also present at this sampling site [which comprised of *Frustulia rhomboides* (20 diatoms) and *Navicula pseudohalophila* (16 diatoms)].

The average dissimilarity between sites 2 & 3 is 85%. At site 2, *Nitzschia sippen* was found to have the largest abundance of 68 diatoms, *Nitzschia scalpelliformis* 56 diatoms and *Frustulia rhomboides* 26 diatoms. Species, which belong to the genera *Navicula*, *Amphora* and *Cymbella*, was also found at the site [comprising of *Navicula diserta* 21 diatoms, *Amphora cfmontana* 15 diatoms and *Cymbella turgidula* 8 diatoms. In sample site 3 *Nitzschia pellucida* shows the greatest abundance of 149 diatoms, which is then followed by *Navicula aequorea* 29 diatoms, *Frustulia rhomboides* 15 diatoms and *Amphora veneta* 9 diatoms.

The average dissimilarity between sites 2 & 4 is 84%. At this site the dominant specie is *Nitzschia scalpelloides*, which comprises of 77 diatoms, *Nitzschia pellucida* has an abundance of 72 diatoms. Species belonging to the genera *Frustulia*, *Aulacoseira*, *Amphora* and *Navicula* was also found at

this site [comprising of *Frustulia rhomboides* 16 diatoms, *Amphora veneta* 11 diatoms, *Amphora arcus* 11 diatoms, *Navicula aequorea* 8 diatoms and *Aulacoseira* sp1 . 7 diatoms]. The average dissimilarity between sites 2 & 5 is 87%. *Nitzschia pellucida* is the species that is dominant at site 5 with an abundance of 58 diatoms and *Nitzschia scalpelloides* shows an abundance of 48 diatoms. The specie *Amphora veneta* has an abundance of 35 diatoms, which is followed by *Navicula aequorea* 34 diatoms, *Frustulia rhomboides* 20 diatoms, *Navicula pseudohalophila* 16 diatoms and *Navicula* sp. 1. Has the smallest abundance of 3 diatoms at this site.

Between sites 3 & 4, the dissimilarity between the 2 sites is 48%. The dominant specie at site 3 is *Nitzschia pellucida* with an abundance of 149 diatoms. Species belonging to the Genera *Navicula* and *Amphora* was also found at this site [comprising of *Navicula aequorea* 29 diatoms, *Amphora veneta* 9 diatoms and *Aulacoseira* sp1 and *Navicula pseudohalophila* share the same abundance of 3 diatoms]. At site 4, the Genera *Nitzschia* has the largest abundance [comprising of *Nitzschia scalpelloides* 77 diatoms and *Nitzschia pellucida* 72 diatoms]. The genera *Navicula*, *Amphora* and *Aulacoseira* was present at this site but in relatively small abundances [*Amphora veneta* 11 diatoms, *Amphora arcus* 11 diatoms, *Navicula aequorea* 8 diatoms, *Aulacoseira* sp1. 7 diatoms and *Navicula* sp 1. has the smallest abundance of 4 diatoms].

The average dissimilarity between sites 3 & 5 is 40%. The specie *Nitzschia pellucida* shows the greatest abundance of 58 diatoms and *Nitzschia scalpelloides* also shows a great abundance of 48 diatoms at this site. The specie *Amphora veneta* shows an abundance of 35 diatoms, which is followed by *Navicula aequorea* 34 diatoms and *Navicula pseudohalophila* 16 diatoms. Species belonging to the genera *Aulacoseira*, *Navicula*, *Diploneis* and unknown diatom specie was also found at the sampling site but in small abundances [comprising of *Navicula* sp. 1. (3 diatoms), *Diploneis* sp. 1. (2 diatoms), *Unknown* sp1. (2 diatoms) and *Aulacosira* sp 1. (1 diatoms)].

The average dissimilarity between the sites 4 & 5 in the estuarine system is 53%. In site 4, the genera *Nitzschia* seems to be the dominant specie. *Nitzschia scalpelloides* has the greatest abundance of 77 diatoms and *Nitzschia pellucida* shows an abundance of 72 diatoms. Species, which fall within the genera *Amphora* [namely *Amphora veneta* 11 diatoms and *Amphora arcus* 11 diatoms], which is followed by *Navicula aequorea* with an abundance of 8 diatoms. Species belonging to the genera *Aulacoseira*, *Navicula* and *Surirella* were also found during sampling but in small abundances [comprising of *Aulacoseira* sp1. (7 diatoms), *Navicula* sp. 1. (4 diatoms) and *Surirella* sp.1. (2 diatoms). At sample site 5, the dominant species are *Nitzschia pellucida* and *Nitzschia scalpelloides*. *Nitzschia pellucida* has an abundance of 58 diatoms and *Nitzschia scalpelloides* with an abundance of 48 diatoms as compared to the diatom abundance that is found at site 4. The species *Navicula aequorea* has an abundance of 34 diatoms and *Amphora veneta* has an abundance of 35 diatoms. *Navicula aequorea* and *amphora veneta* displays a greater abundance at site 5 as compared to site 4. Species belonging to the genera *Navicula* and *Aulacoseira* were also found at this site [comprising of *Navicula pseudohalophila* 16 diatoms, *Navicula* sp1. 3 diatoms and *Aulocoseira* sp. 1. has the smallest abundance of 1 diatom].

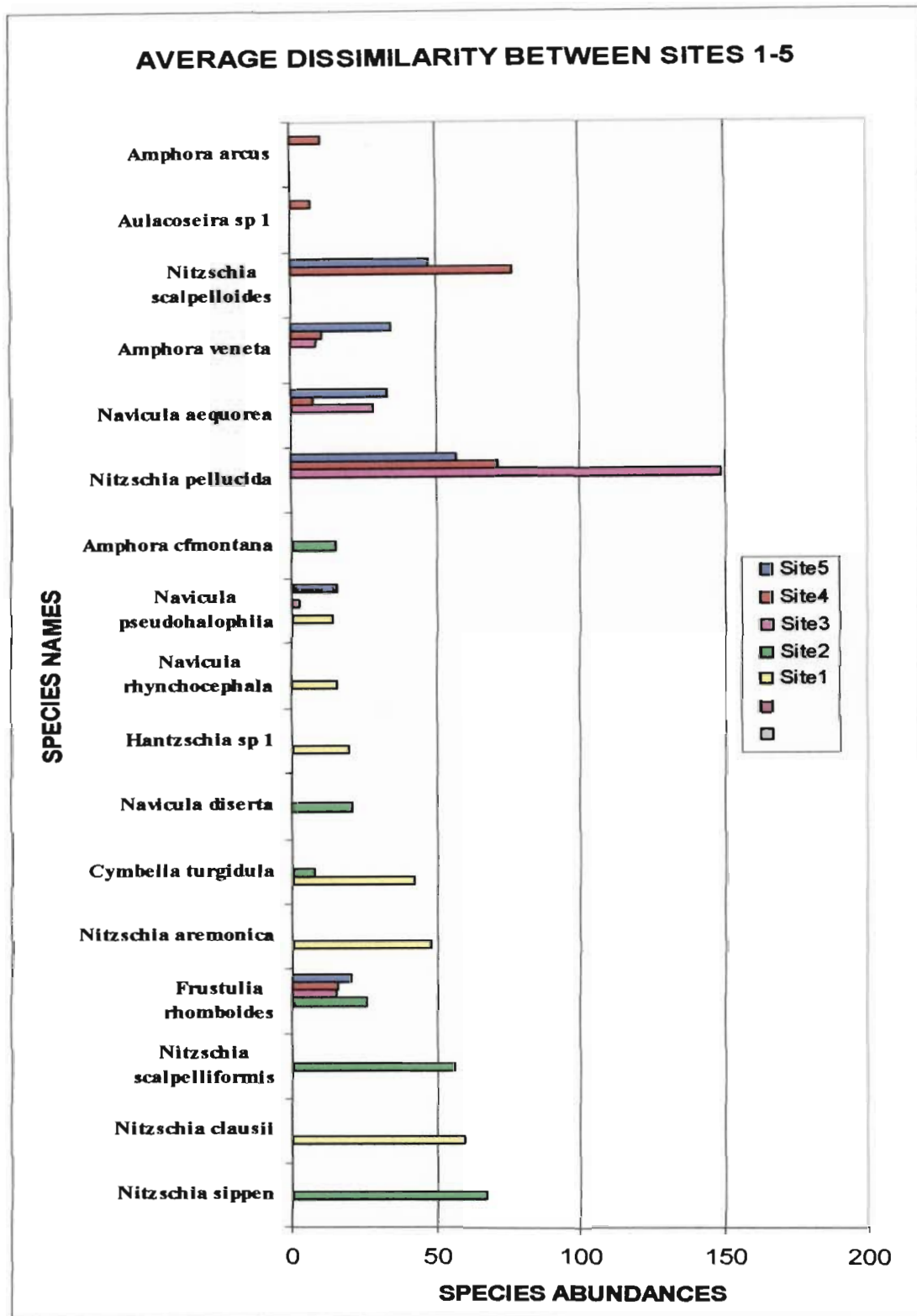


Figure 5.13 Average dissimilarity between sites 1-5

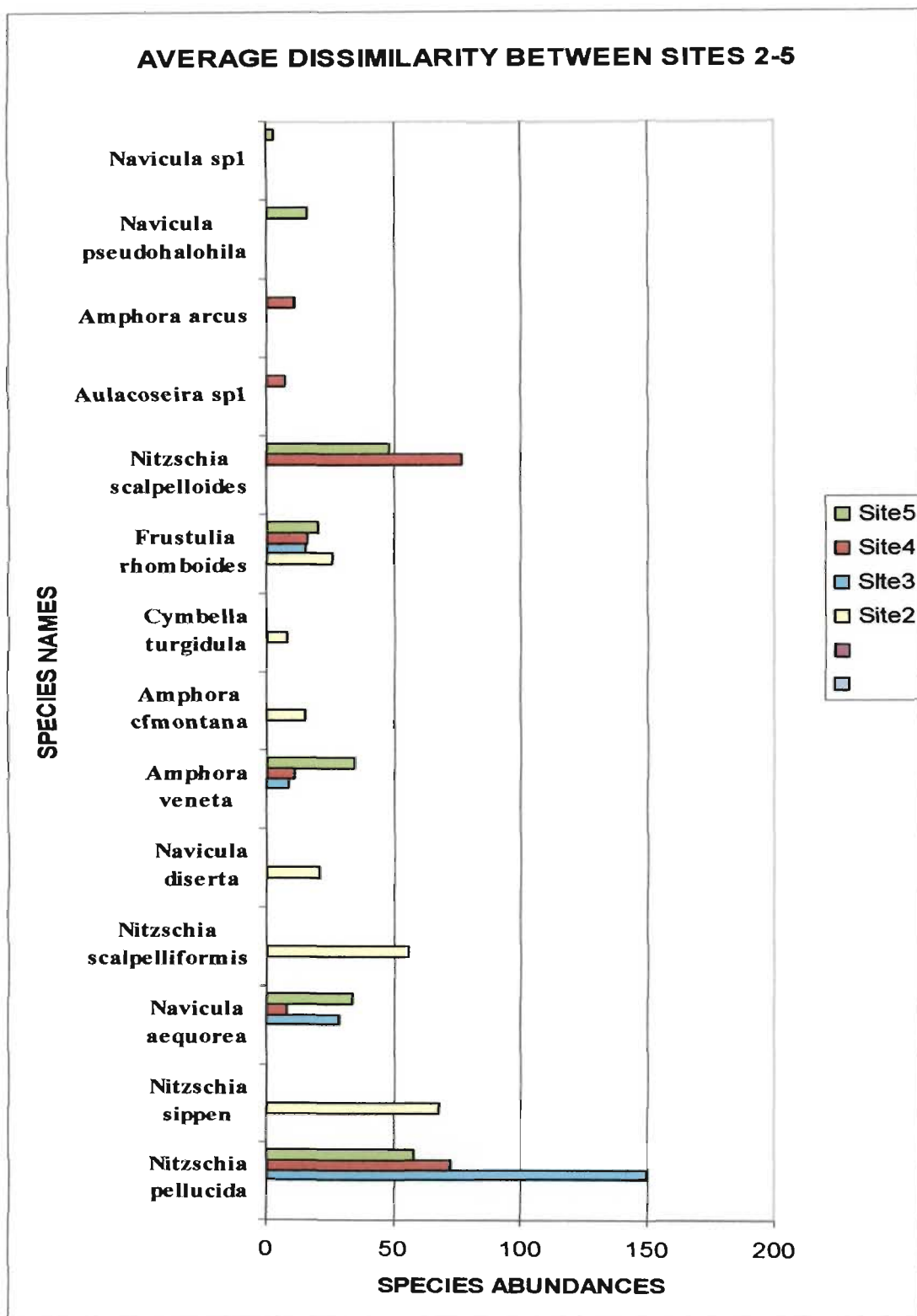


Figure 5.14 Average dissimilarity between sites 2-5

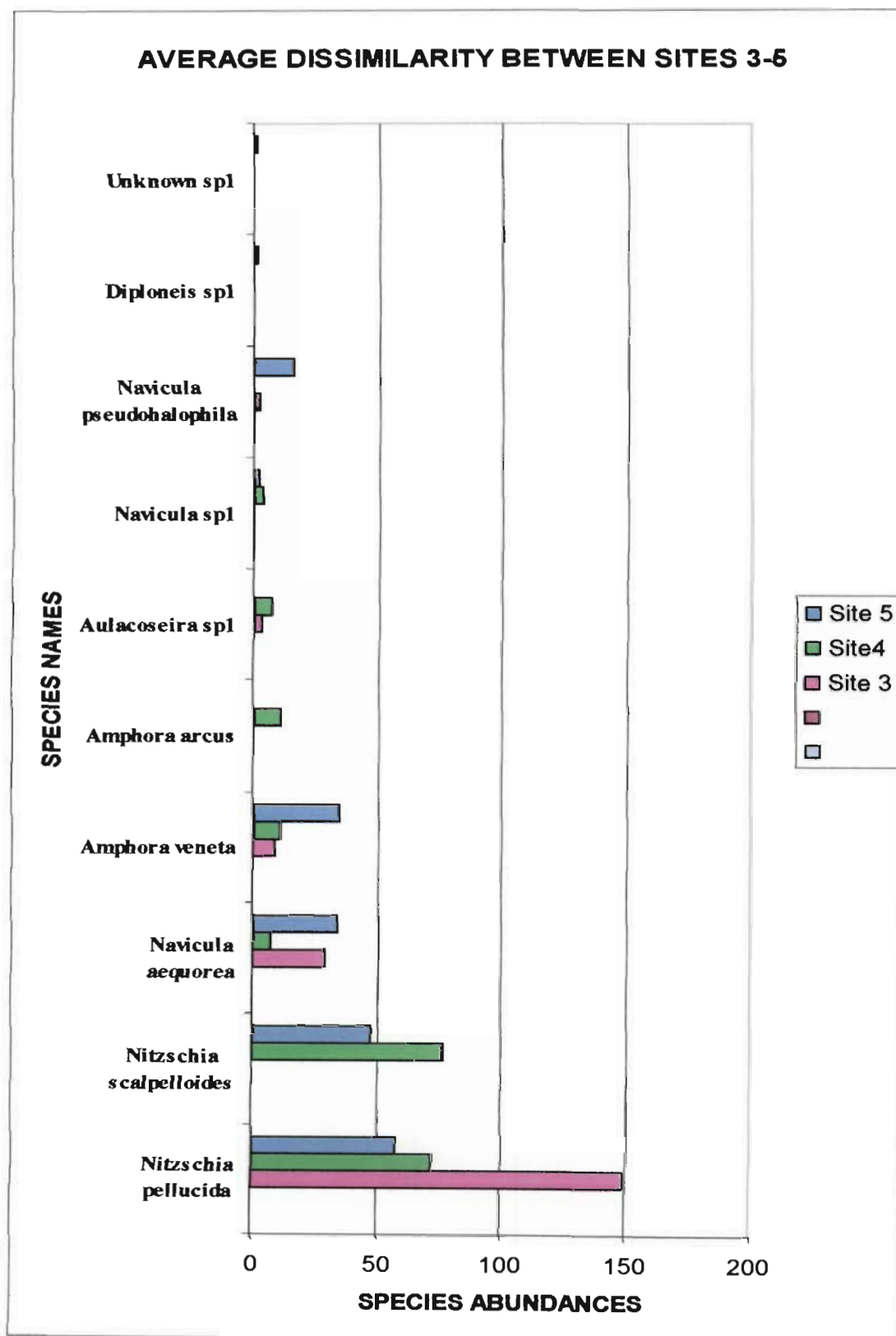


Figure 5.15 Average dissimilarity between sites 3-5

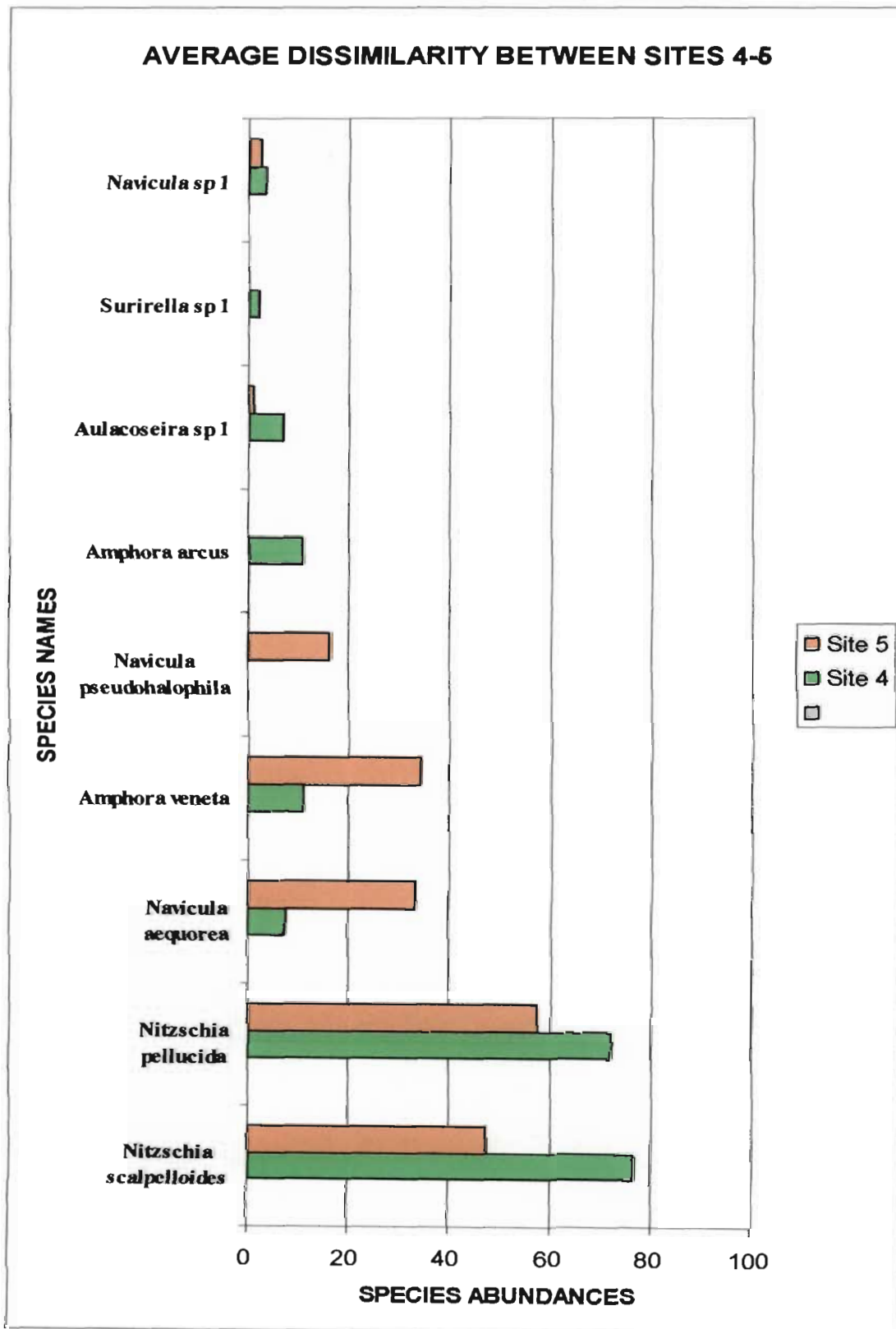


Figure 5.16 Average dissimilarity between sites 4 and 5

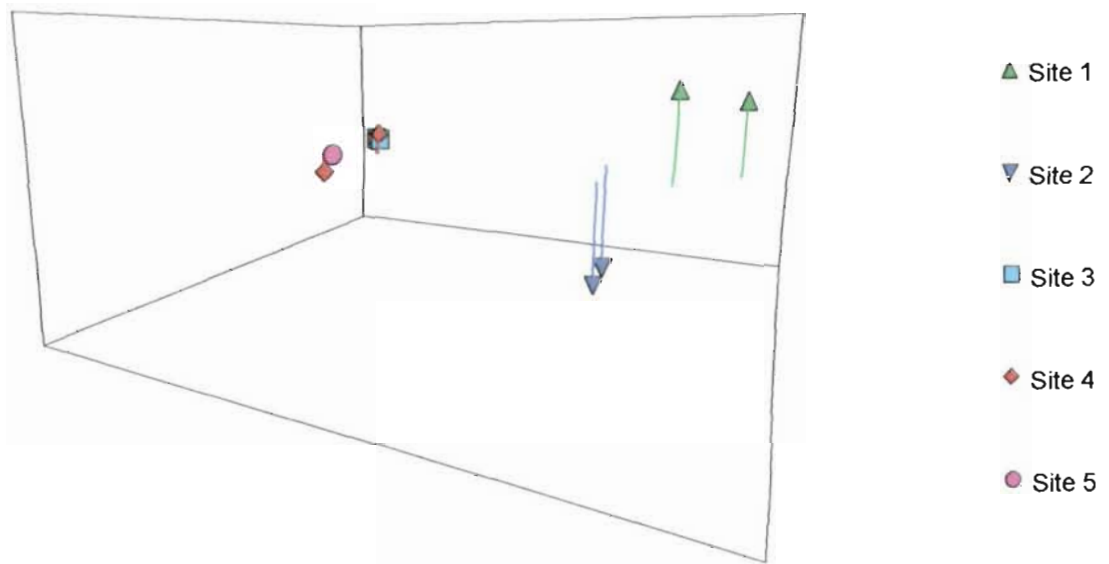


Figure 5.17 MDS Plot of St.Lucia Estuarine System

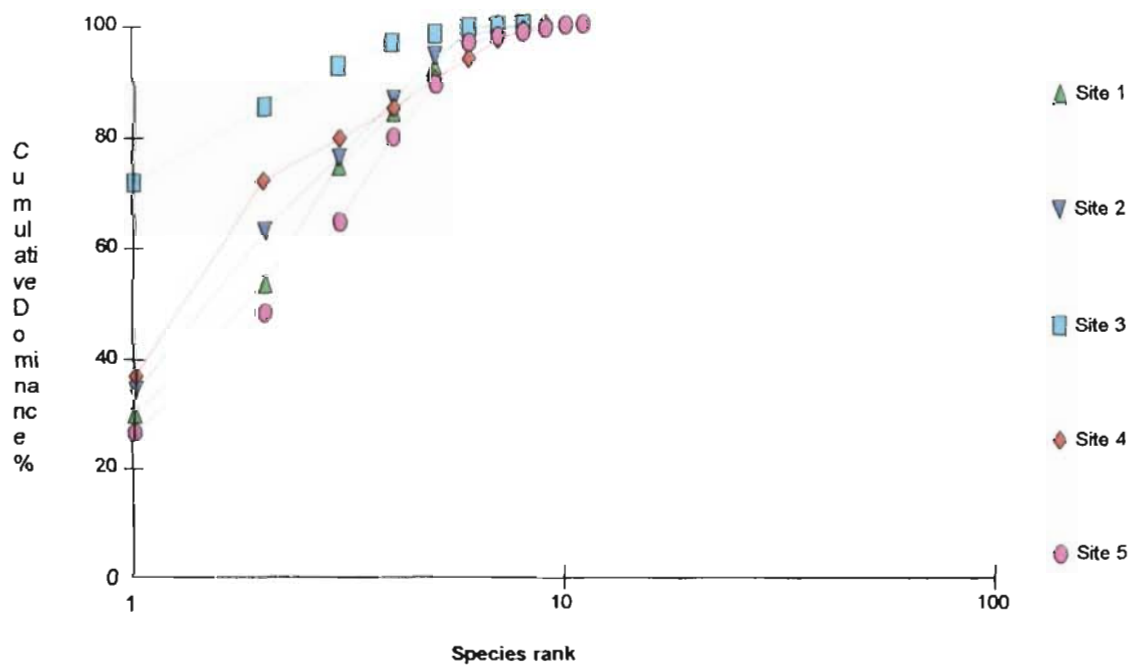


Figure 5.18 K-dominance curve of St.Lucia Estuarine system.

5.2.3. Physical and Chemical Parameters

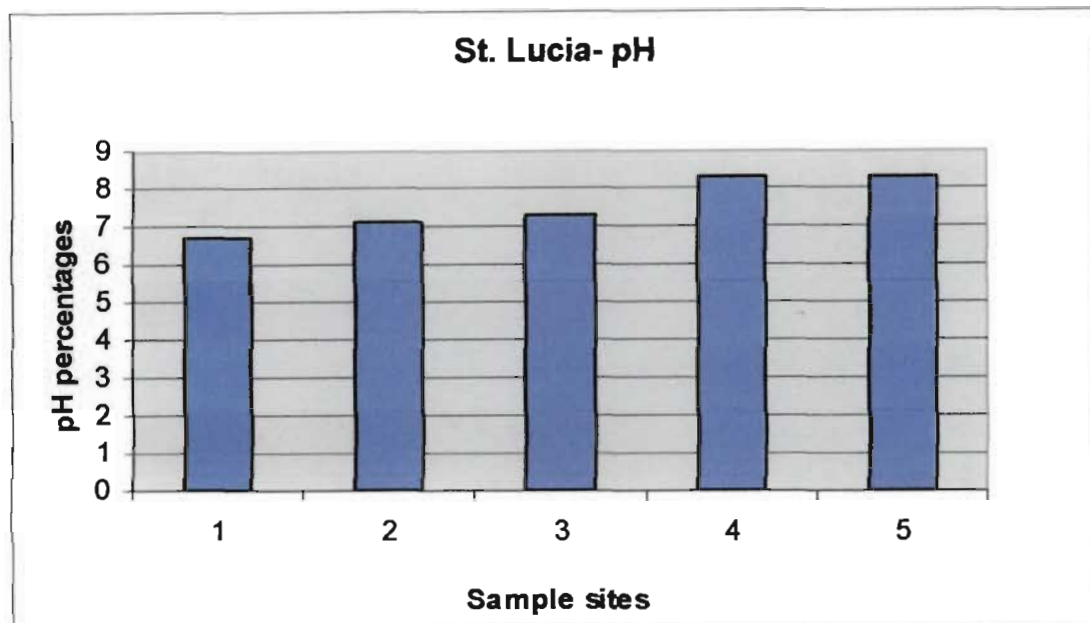


Figure 5.19 Graph depicting pH values at the various sampling sites

As seen in Figure 5.19, sampling sites 2-5, shows a high pH value as compared to site 1, which has the lowest pH value. From the graph it can be seen that site 1 has a pH value of 6.51, which then increases to 7.1 at site 2. Sampling site 3 then increases to 7.27, then the pH value increases again to 8.28 at site 4. Sample site 5, which was the last site to be sampled in the estuarine system, displays the pH value of 8.28.

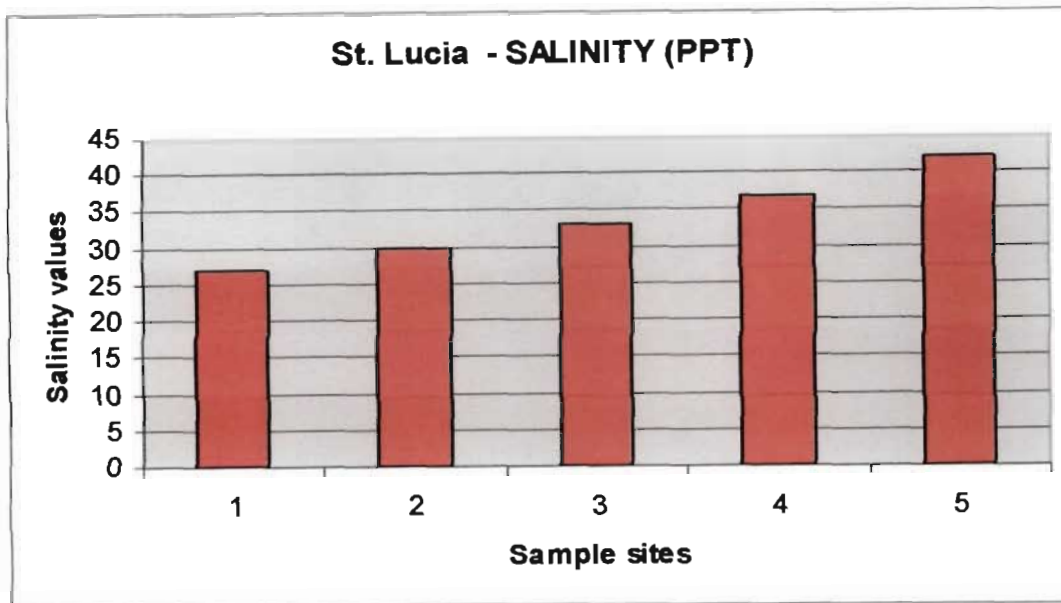


Figure 5.20 Salinity values among the five sampling sites

As seen in Figure 5.20, the salinity values at site 4 and site 5 display the highest value [comprising of 42ppt site 5 and 37ppt at site 3]. Sample site 1 has the lowest salinity value of 27ppt as compared to all the other sites that were sampled in the estuarine system. Site 2 has a salinity value of 30ppt and site 4 has a salinity value of 33ppt. As seen in the graph, the salinity levels increases at site 2 and site 3, then decreases at site 4, then at site 5 the value increases once again.

5.2.4. Biological Parameters- Diatom Richness (D), Evenness (J) and Diversity (H) species indices

The species Diversity (H') between the sampling sites varied from 0.9 – 1.4 % respectively (refer to Figure 5.21). Sample site 5 showed the highest species diversity as compared to the other 4 sites. Sample site 5 displays the highest specie richness as compared to the other sites (refer to Figure 5.22). Species evenness between the 5 sites was low and varied between 0.5 to 0.7 % (refer to Figure 5.21). There were marked variations in the richness, evenness and diversity between the sampling sites, which reflects the variations in the water chemistry as well as the estuarine habitat. The pairwise k- dominance curve

(refer to Figure 5.18) comparisons between groups was significant in the estuarine system, while the pairwise R values, which give an absolute measure of how separated the five groups are, which gives an absolute measure of how separated the groups are, i.e. the $R = 0.62$.

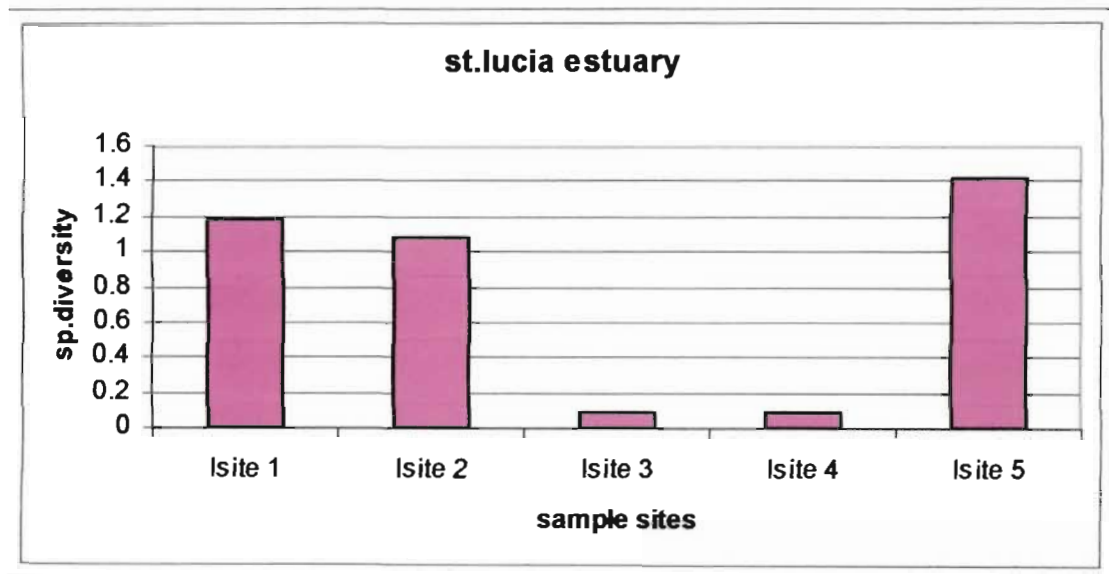


Figure 5.21 Species diversity graph indicating the diversity of diatoms along the sites.

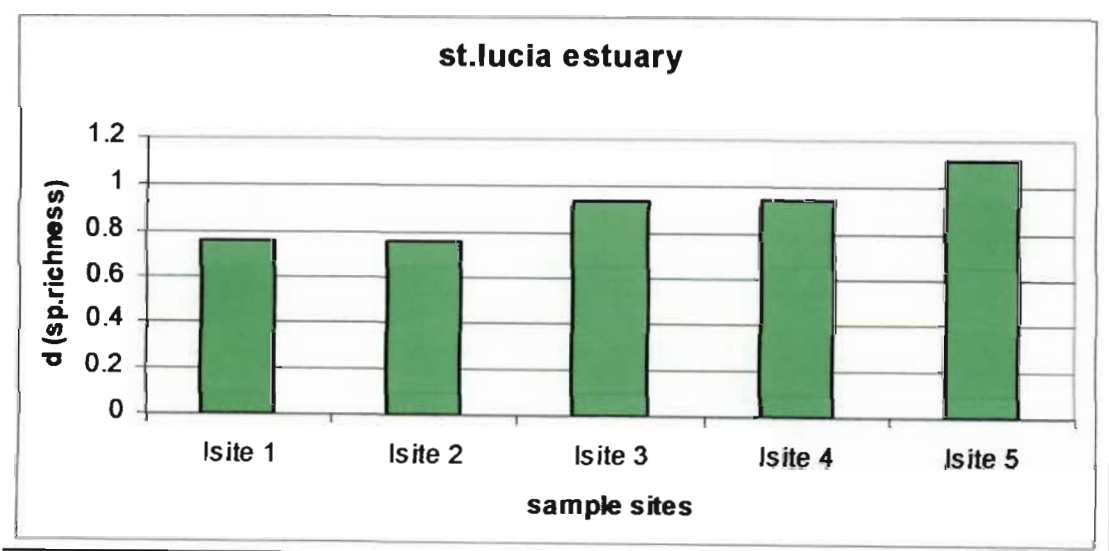


Figure 5.22 Graph indicating the differences of species richness between the five sampling sites

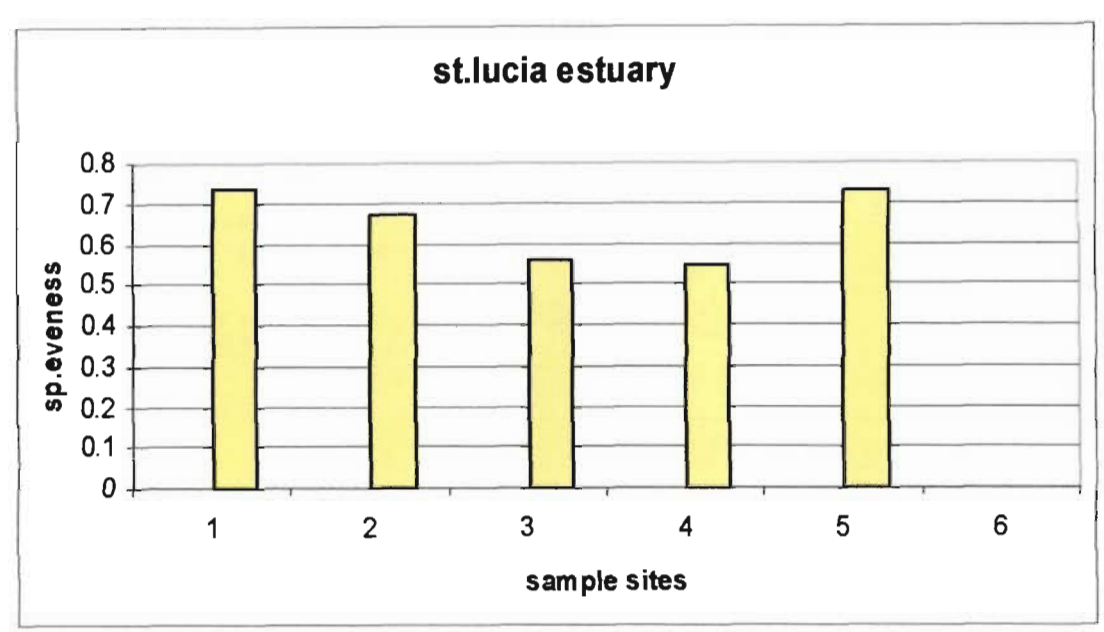


Figure 5.23 Species evenness graph indicating the diatom differences between the five sites

5.2.5. DISCUSSION

The salinity fluctuations in the St. Lucia estuary are probably the most extreme of any estuarine system on the southeast coast of Africa (Owen & Forbes, 1997). The unusual combination of tributary rivers at both the northern reaches of the lake and at the mouth together with the lake being lower than the mean sea level, leads to reversed net flow patterns and salinity gradients in the system during dry periods (Owen & Forbes, 1997). The resultant inflow of sea water to the lakes, combined with the high evaporation rates contributes to the shallow nature of the system, leads to hypersaline conditions that occur in the middle and the northern reaches, where salinity values can reach an excess of 50 parts per thousand have been recorded several times, even exceeding 100 in 1970 and 1983 (Forbes & Cyrus, 1993).

Salinity is one of the main physical variables in estuaries, with daily seasonal fluctuations which are due to rainfall and tides or stressful events such as droughts (which is being experienced now) or floods. It is widely accepted that salinity is regarded as one of the most important factors that affects estuarine benthic communities, with many authors that have reported changes in benthic community structures in response to droughts and floods (McLachan & Grindley, 1974; Boltt, 1975, Cyrus, 1988, Hanekom, 1989; Read & Whitfield, 1989). As seen in the graph (refer to figure 5.20.) sample site 5 displays the highest salinity value (42 ppt), which is then followed by sample site 4 with 37ppt. Site 4 has a salinity value of 33ppt, site 2 a salinity value of 301ppt. Sample site 1 has the lowest salinity value of 27ppt.

The susceptibility of diatoms to the physico- chemical parameters defining their environment was first demonstrated by Hustedt (1939) in relation to pH of lake waters. The changes in pH have wide ranging effects on water chemistry, and therefore on aquatic ecosystems. The pH of a sample of water determines the particular chemical species in which many elements are found in that sample. As seen in graph which displays the pH ranges along the sampling sites (refer to Figure 5.19), sample site 4 and 5, displays the same pH value of 8.28, sample site 3 has a pH value of 7.27 and sample site 2 displays a value of 7.1. Sample site 1 has the lowest pH value of 6.71 compared to all the sites sampled.

The overriding effect of water chemistry, especially salinity and pH, on species composition of diatom assemblages has been recognized since the beginning of ecological studies of this algal group (Potapova & Charles, 2002). Many studies that have employed multivariate analyses to analyse diatom communities showed that they varied conspicuously along gradients of ionic concentrations and pH (Johansson, 1982; Leclercq & Depiereux, 1987; Pan *et al.*, 1996). Factors that govern water mineral content become more evident and these include landuse, bedrock type and watershed size (Potapova & Charles, 2002). As pH is partly regulated by the water buffering capacity or the amount of alkaline metals, it also depends on all the above-

mentioned factors, but also on the extent of wetlands, which are related to local topographical features (Potapova & Charles, 2002).

The changing salinity in St. Lucia is the main physical factor that is driving the constant change in ecological conditions within the estuary. It is this factor that makes St. Lucia such an interesting system (R. Taylor, 2003). The mouth of the St. Lucia estuary closed naturally in 2002. It probably would have opened within a few weeks, but in an effort to protect it from an oil spill from the stranded Jolly Rubino, a sand wall was pushed up by bulldozers to prevent natural breaching of the mouth. Since then the rainfall has been low and evaporation has exceeded fresh water gains. As a result there has been a drop in the water levels. Lastly, as stated by R. Taylor, (2003): "we would have to be concerned if the salinity throughout the estuary was very high, as there would be a few places in which estuarine organisms that cannot tolerate the high salinities wouldn't be able to survive."

5.2.6.Past results for St. Lucia

The diatom associations of the St. Lucia lagoon system were composed of marine littoral species. Only those that were able to osmoregulate over a wide salinity range were common and wide spread in the lagoon. The periodic changes in osmotic pressure occur in the lagoon, the amplitude of which increases with distance from the sea and is greatest at False Bay. The number species consequently drops from the coast to False Bay since fewer and fewer species are able to tolerate wider and wider amplitudes of salinity changes. The predominance of littoral species and the absence or scarcity of the fresh and brackish water diatoms of the tributaries is in agreement with the observed direction of water flow in the lagoon. This demonstrates that the sediments in the lagoon are chiefly of marine origin. The diatoms in the sediments of the St. Lucia are derived mostly from the flora of the marine littoral. Fresh water species were found only in traces in the deposits. It

seems highly probable that at the time of deposition the influence of the rivers on the diatom flora was unimportant as it is today (Cholnoky, 1968).

5.2.7 Conclusion

This study shows that diatoms, when classified by statistical techniques according to species and corresponding physico- chemical parameters, can be grouped into assemblages, which are indicative of specific ranges of water quality conditions. This study has shown that diatom populations can adapt to their community structure to changes in their habitat specifically and quickly in response to changes in the water chemistry.

From the past slides, which were viewed, the dominant species belonged to the Genera *Navicula*, *Nitzschia* and *Amphora*. In addition, when we look at the dominant species that were found in the St.Lucia estuarine system now, the dominant Genera in the system are species belonging to the Genera *Navicula*, *Nitzschia* and *Amphora*. Therefore it can be stated from the results, that the diatoms that were found in 1968 compared to 2005- 2006 has not changed with regard to the Genera but the species have evolved over the years.

Diatoms found in St. Lucia estuarine system

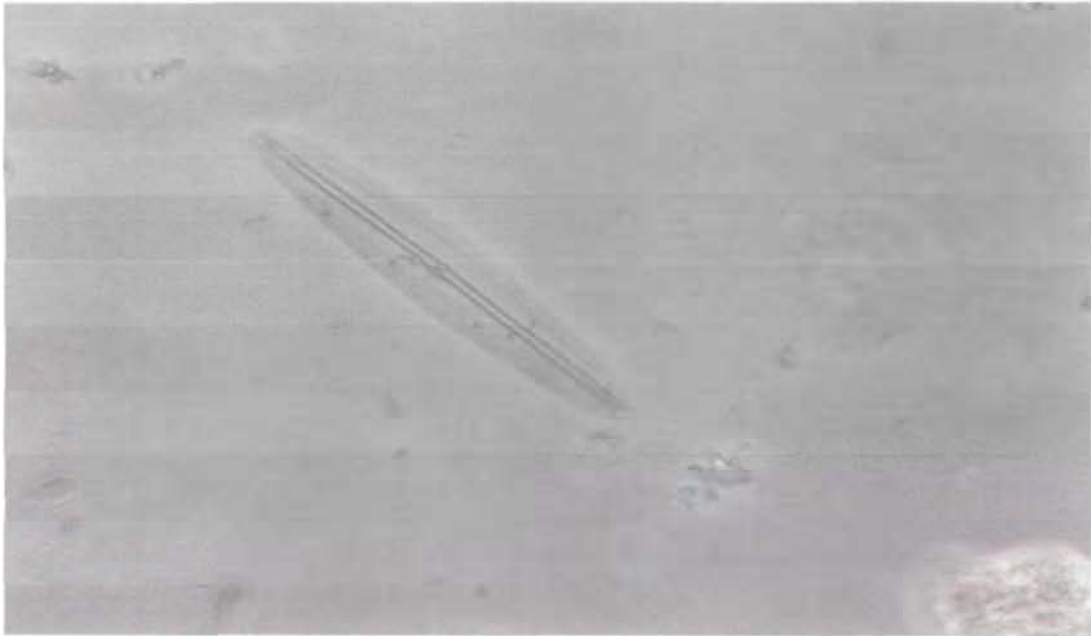


Plate 5.7: ***Frustulia rhomboides***

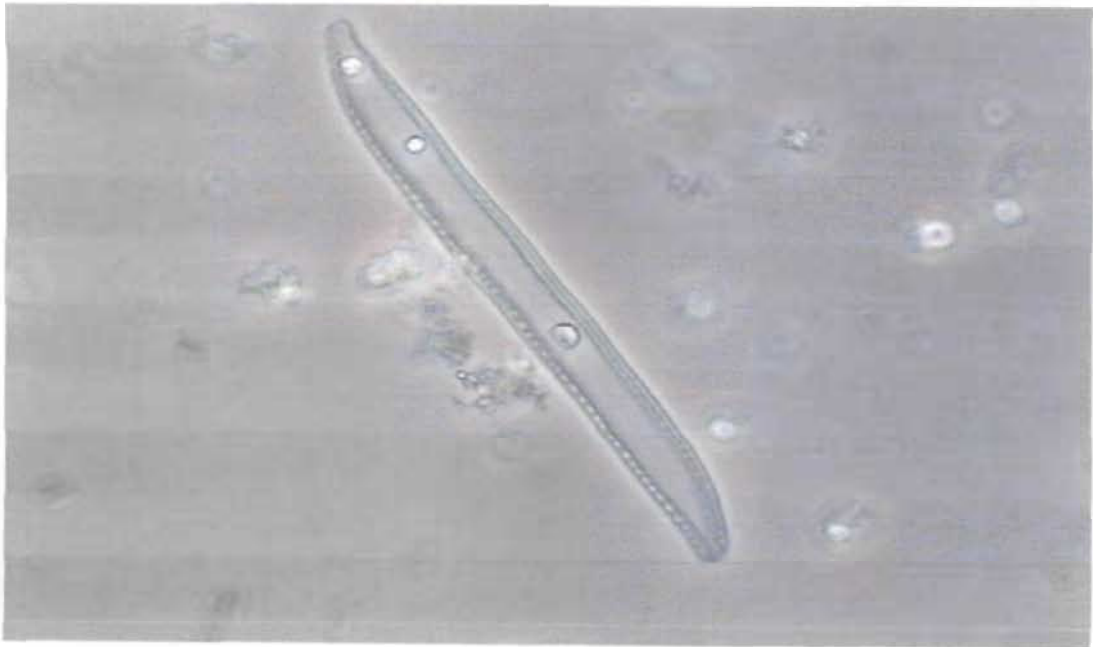


Plate 5.8: ***Nitzschia scalpelloides***

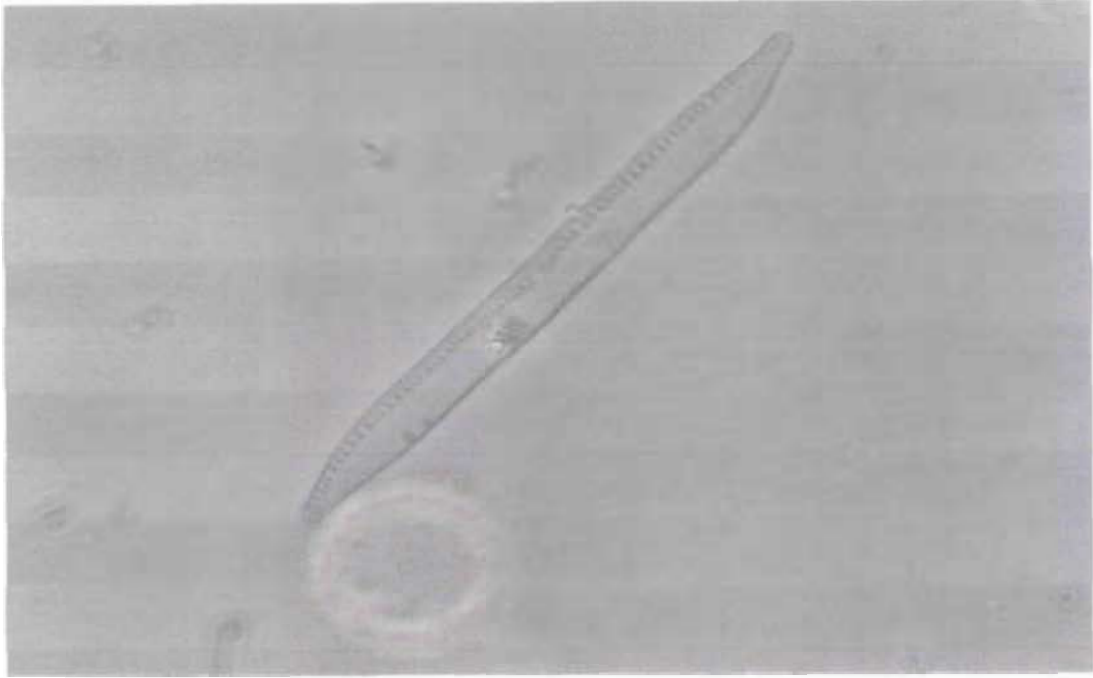


Plate 5.9: *Nitzschia pellucida*

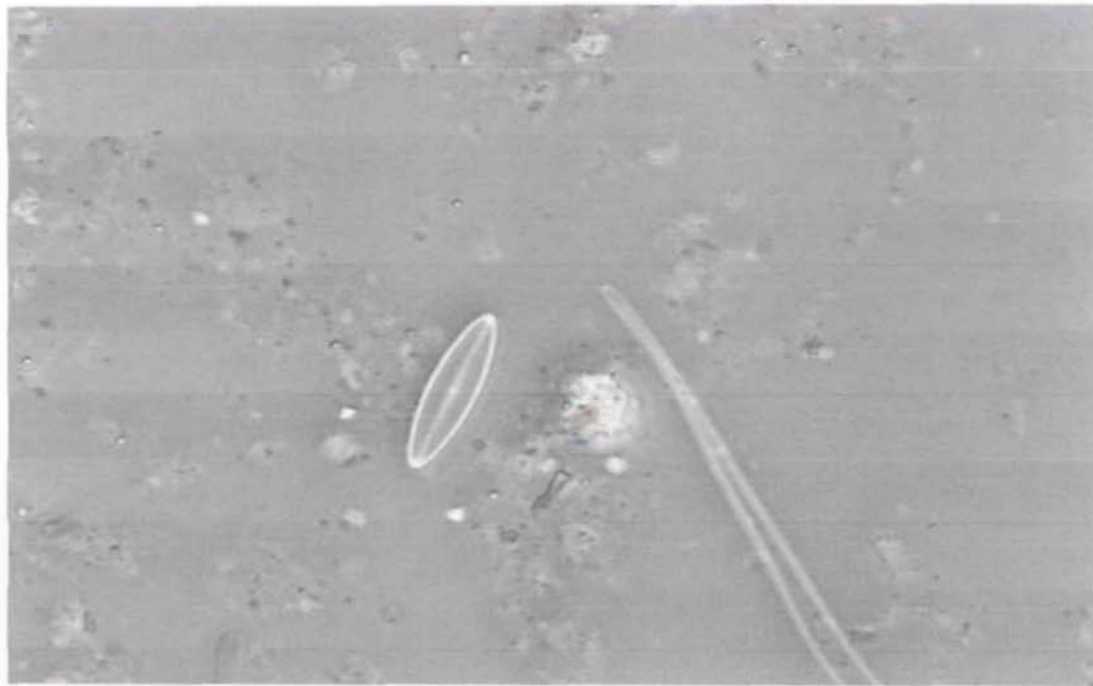


Plate 5.10: *Navicula aequorea*

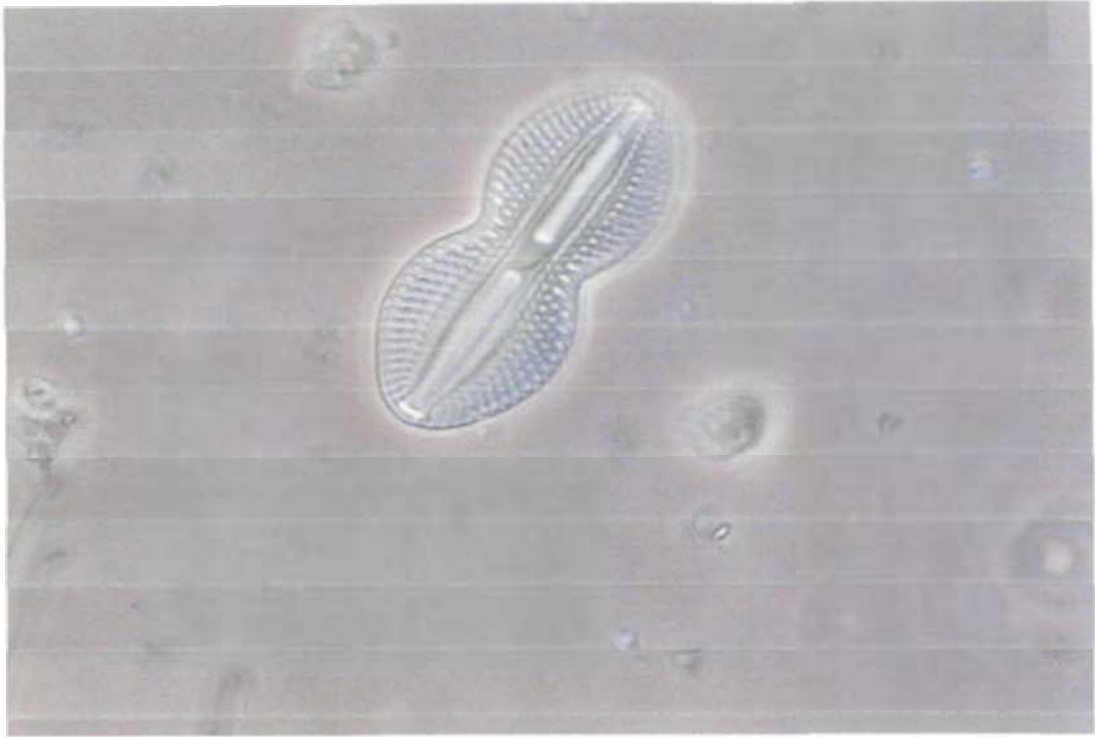


Plate 5.11: **Diploneis SP1**

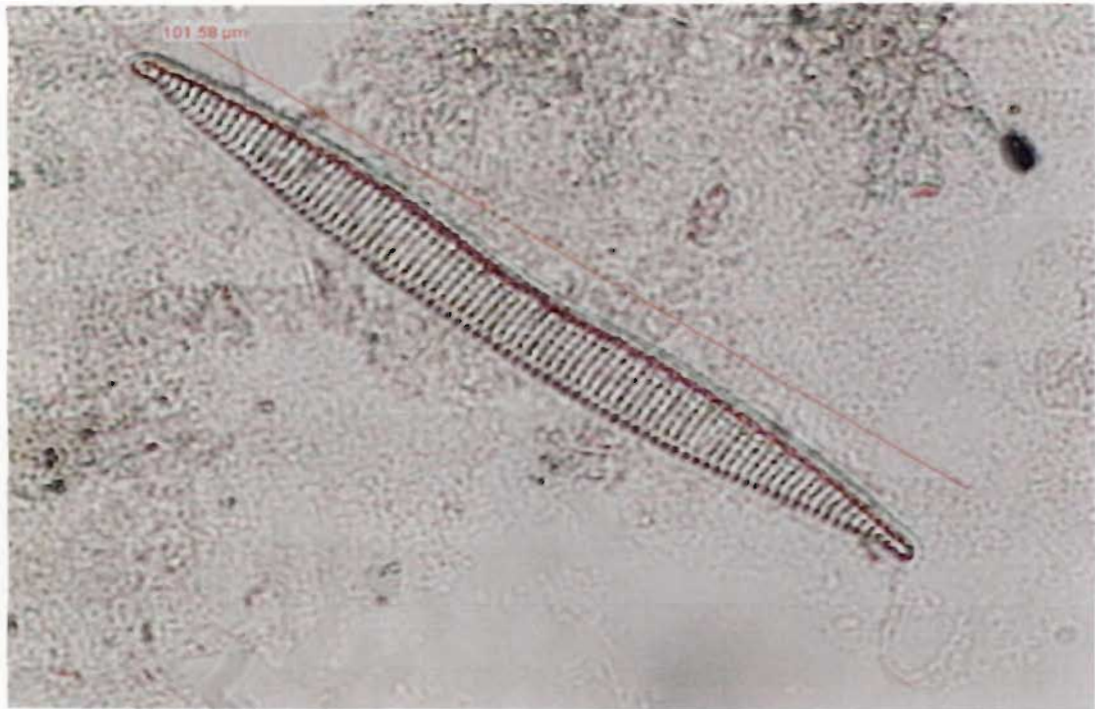


Plate 5.12: ***Hantzschia* sp1**

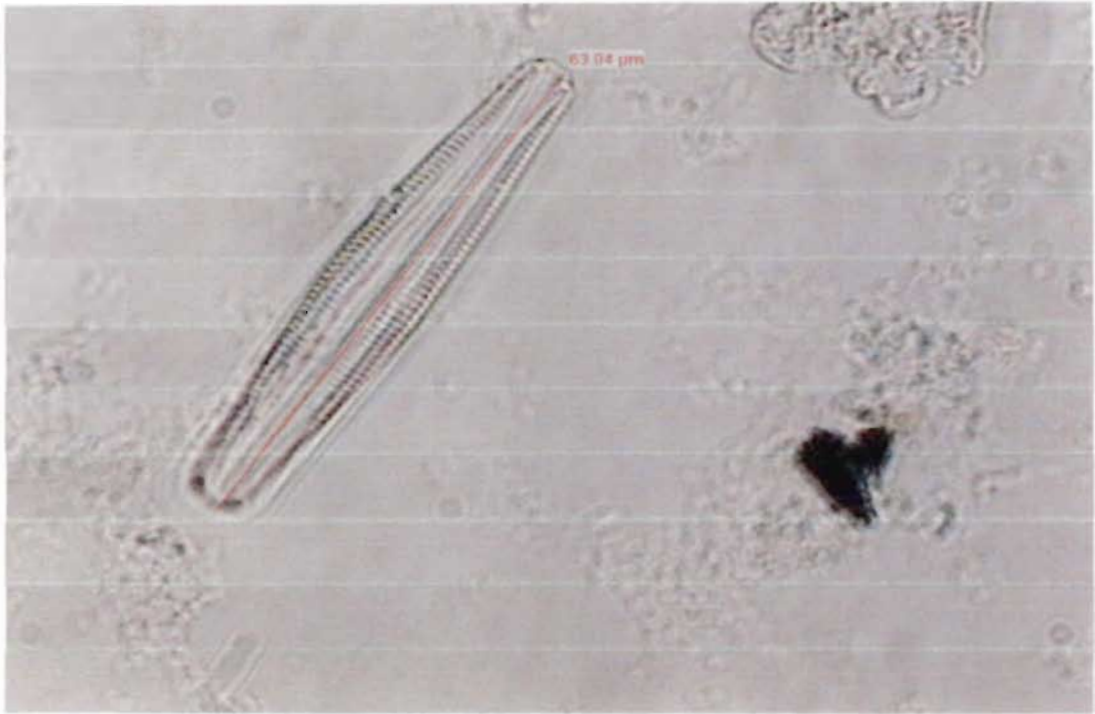


Plate 5.13: *Nitzschia clausii*

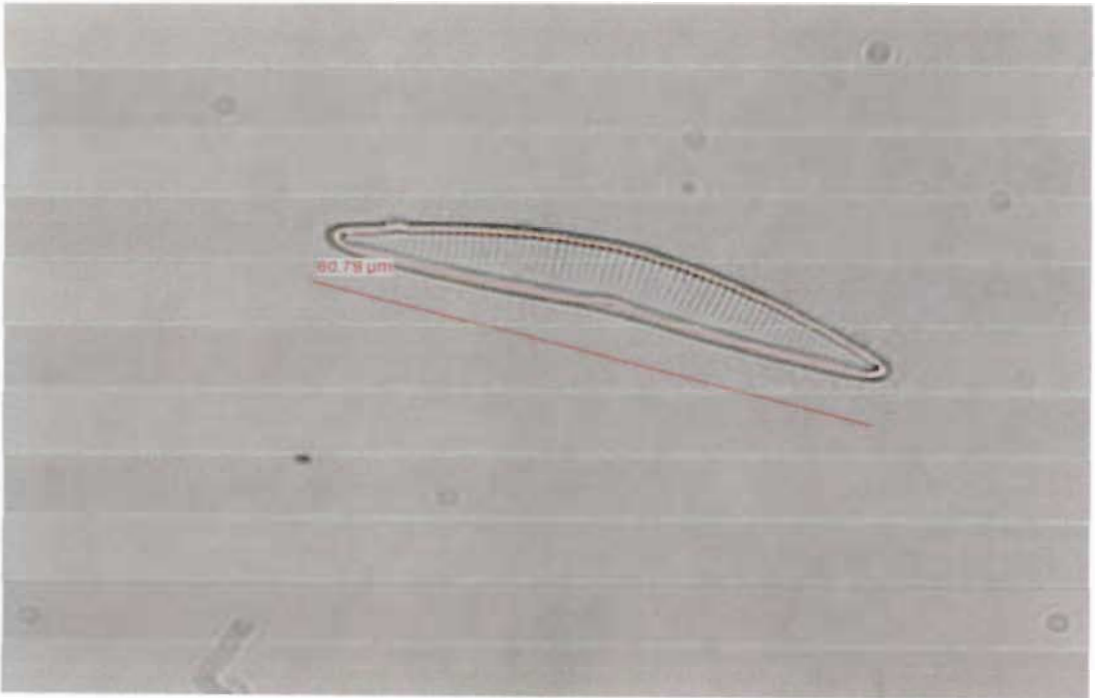


Plate 5.14: *Amphora arcus*



Plate 5.15: *Navicula rhynchocephala*

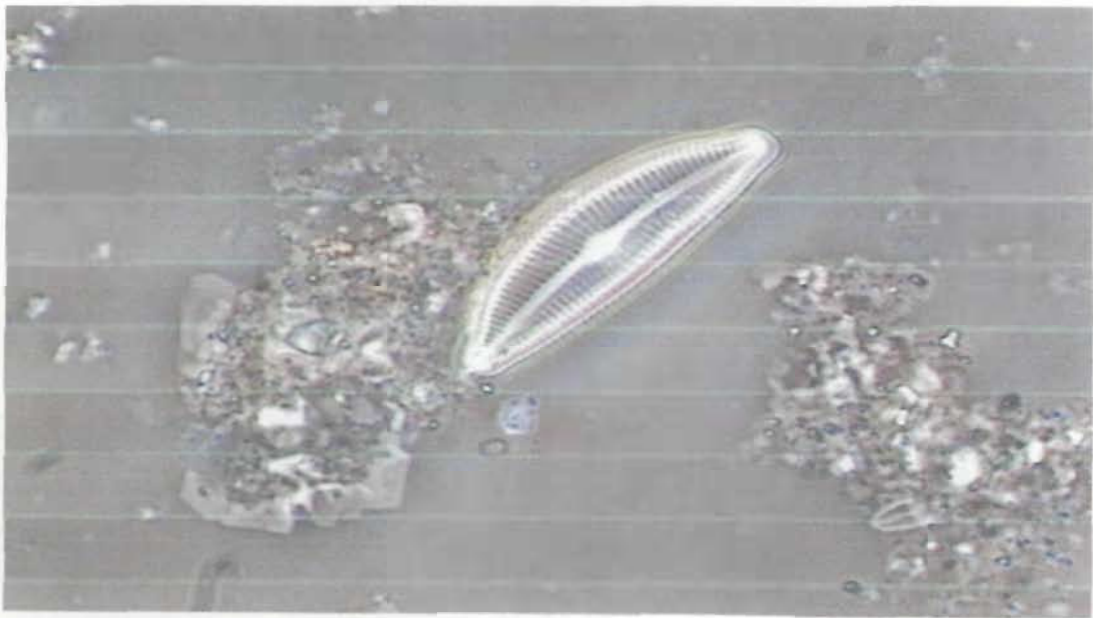


Plate 5.16: *Cymbella turgidula*

CHAPTER SIX

6. OVERALL CONCLUSION AND RECOMMENDATIONS

6.1. OVERALL CONCLUSION

The chemical analysis of water provides a good indication of the chemical quality of the aquatic system, however, it does not integrate ecological factors and don't necessarily reflect the ecological state of the system (Karr *et al.*, 2000). Therefore, biological assessment is a useful alternative for assessing the ecological quality of an aquatic ecosystem, since biological communities integrate environmental effects of water chemistry, in addition to the physical, chemical and biological characteristics of estuary and river systems.

The conditions of the water quality in the Mnweni River system was reflected in the physical, chemical and biological parameters, as well as the species composition of the microphytobenthos community that was found in the system. The relative abundance and the differences in specific sensitivity of certain diatom species to the changes in the flow character, anthropogenic modification, influences on the river bank and catchment land use was significant in relation to diatom assemblage composition that was found at the 5 sampling sites. The results of the diatom composition in the Mnweni river system, is a clear indication that the diatom found at the sites were typical of clean or mildly enriched water conditions.

The results of the research that is present in the St. Lucia estuarine system, is a relatively clear indication that the general distribution patterns in the estuary are controlled by salinity levels. The changes in the community structure in the estuarine system indicate the variable nature of estuarine benthic communities under different salinity conditions. Contrary to the drought

conditions that are being experienced, the microphytobenthic biomass did not exhibit any significant differences. A possible reason for this may be due to The rapid recovery ability that microphytobenthic communities exhibit in response to physical, chemical and biological disturbances that are occurring in their habitat. As stated by Underwood, 2004 & Perissinotto *et al.*, 2006; that recent studies indicate that the recovery time scale of microphytobenthos that are exposed to physical, chemical disturbances may be extremely short.

The findings in the St. Lucia estuarine system provide a relative utility of diatom communities for the use in bioassessment of estuarine conditions. Therefore, this study does indicate that diatom associations may be successfully employed to assess the quality of an estuarine system.

6.2. RECOMMENDATIONS

Despite their ecological importance, practical usefulness and broad studies that have been conducted to determine the dominants and subdominants of microphytobenthic diatoms in selected estuaries and rivers, an in-depth study of patterns of diatom taxa distribution and underlying causes are largely unexplored in South African literature.

Much of the diatom information in South Africa are purely taxonomic in nature therefore there is a need to gather information on dominant species that are found in South African estuarine and river systems.

I would like to suggest that future research investigate primary productivity and grazing of microphytobenthos assemblages in rivers and estuaries.

Lastly, a study of the effect that river flow has on the microphytobenthic biomass would be of interest, since abstractions and impoundments of rivers has reduced the amount of river water that flows into estuaries. A study that is undertaken like this will enable us to understand and also predict the changes and determine the limit of acceptability of reduced river water input.

The management and monitoring of rivers and estuaries is crucial in order to ensure the ecological integrity of these systems.

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