CONTRIBUTIONS TO INFORMATION REQUIREMENTS FOR THE IMPLEMENTATION OF RESOURCE DIRECTED MEASURES FOR ESTUARIES

Volume 2

Responses of the biological communities to flow variation and mouth state in two KwaZulu-Natal temporarily open/ closed estuaries

R Perissinotto · A Blair · A Connell · NT Demetriades · AT Forbes · TD Harrison · K Iyer · M Joubert · I Kibirige · S Mundree · H Simpson · D Stretch · C Thomas · X Thwala · I Zietsman

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Water Research Commission

Contributions to Information Requirements for the Implementation of Resource Directed Measures for Estuaries

Volume 2
Responses of the biological communities to flow variation and mouth state in two KwaZulu-Natal temporarily open / closed estuaries

REPORT TO THE WATER RESEARCH COMMISSION

BY

THE CONSORTIUM FOR ESTUARINE RESEARCH AND MANAGEMENT

EDITOR

JB Adams, University of Port Elizabeth

AUTHORS

R Perissinotto[1], A Blair[3], A Connell[3]
NT Demetriades[1], AT Forbes[1], TD Harrison[3], K Iyer[1] M Joubert[1] I Kibirige[1]
S Mundree[3], H Simpson[1], D Stretch[2], C Thomas[1], X Thwala[1], I Zietsman[2]

[1] School of Life & Environmental Sciences, G. Campbell Building, University of Natal, Durban 4041, South Africa
[2] Centre for Research on Environmental, Coastal & Hydrological Engineering (CRECHE), University of Natal, Durban 4041, South Africa
[3] Division of Water, Environment & Forestry Technology, CSIR, PO Box 17001, Congella 4013, Durban, South Africa

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EXECUTIVE SUMMARY

The aim of this programme was to provide data and understanding to support the estuarine component of the Resource Directed Measures programme of the Department of Water Affairs and Forestry. This was a multidisciplinary programme of the Consortium for Estuarine Research and Management (CERM). The Amazon sub-committee of CERM administered the project. There were three projects that are reported on in Volumes 1 and 2. The capacity building and technology transfer components of the study are reported on in the Executive Summary.

PROJECT OBJECTIVES

Project 1: Improving the biodiversity importance rating of SA estuaries (Volume 1).

1. Improvement of existing database through collation of existing data.
   Recent Co-ordinated Water Bird counts (CWAC) were entered in a database. Recent fish data were identified but were not included due to the expense. Existing invertebrate data for all estuaries were collated in an excel database.

2. Filling information gaps with field work : summer bird counts in Wild Coast estuaries.
   Birds were counted in 67 estuaries along the Transkei coast in January-February 2002. A total of 2206 waterbirds and 40 waterbird species were recorded in the estuaries. Excluding terns, only 983 birds were counted in estuaries in total. Mangrove estuaries tended to have higher numbers of waders, and of birds in general. Ordination and multi-dimensional scaling analysis did not reveal any significant patterns in bird communities within the Transkei estuaries. Indeed, even though permanently open estuaries had more intertidal area, they did not support much higher numbers or variety of waders, as might have been expected. There was no major change in the importance ratings of Transkei estuaries as a result of the data generated in this study, due to the small numbers of birds involved. However, the importance ratings for Transkei estuaries are now accurately known, and a few of the larger estuaries are expected to be fairly important for birds, especially those supporting rare species such as Mangrove Kingfisher.

3. Predictors of biodiversity.

This research is reported on in three chapters in Volume 1:

- *A field study of the intertidal invertebrate fauna of 16 warm temperate estuaries.*
- *Predicting invertebrate species richness on the basis of broadscale estuary characteristics*

Meetings were held with team members and other interested parties on the selection of estuaries for study and the methodology, as well as the emphasis of the study. Due to budget constraints field sampling only took place in the warm temperate zone. Intertidal invertebrates were sampled in 16 estuaries. This project has shown that each estuary is highly unique in character and that intertidal invertebrate communities are distinct and cannot be accurately predicted on the basis of simple environmental measures. A large proportion of species recorded were only present in one or two of the estuaries sampled. Species richness per estuary was the most predictable attribute and was strongly correlated with the slope of the river and estuary mouth condition.

The study then attempted to develop a predictive model for invertebrate species richness for all estuaries using available data on some warm temperate estuaries. The variables considered were those that are readily available for most South African estuaries. Some 538 invertebrate taxa have been recorded in South African estuaries. These were separated by habitat into intertidal benthic, subtidal benthic and planktonic invertebrates.

This study is the first attempt at collating the invertebrate data of all South African estuaries, and provides new insight into the relative contribution that these sub-communities make to estuarine biodiversity. The research found that there is relatively little overlap between species recorded in the plankton, subtidal and intertidal macrobenthos and suggests that data on invertebrate biodiversity for only one or two of these groups would not adequately represent the biodiversity of an estuary.

Significant regression models were obtained for total species richness and subtidal benthic species richness. Subtidal species richness was significantly correlated with the area of submerged macrophytes (typically eelgrass Zostera capensis*), and the area of mangroves. When estuary types were analysed separately subtidal species richness was significantly correlated with mean annual runoff in both cases.*

For warm temperate permanently and temporarily open estuaries, species richness as a proportion of the 'species pool' was significantly positively correlated with estuary size. The research thus showed that estuary size is the most important variable determining the number of potential species that actually occur in an estuary. Estuary size is correlated with habitat area, habitat diversity and estuary type (i.e. permanently open estuaries are larger than others). The predictive equations for invertebrate species richness were used to determine estuarine importance scores for all South African estuaries. In the revised importance rating there was no correlation between the new scores and the old invertebrate scores, further supporting earlier concerns that the original index was inadequate.

4. Updating the importance rating of all South African estuaries.
   Existing and field study data were used to produce an updated importance rating database for South African estuaries. Despite the radical changes in the invertebrate importance scores and the significant changes in the bird importance scores, the overall ratings of estuaries were not greatly affected.

Project 2: Quantifying water quality changes that affect different taxa (submitted as a CD).

Quantitative information on the response of estuarine taxa to changes in water quality was collated. A CD has been compiled containing the data (tolerance bands and exposure times) and information tables in MS Excel format. The CD is available for use in future Resource Directed Measures studies. The data are contained in the directory, ***Estuarine Water Quality Database****. Each biotic group is in a sub-directory, microalgae, macrophytes, invertebrates and fish. Each constituent is in a separate file in the sub-directories e.g. salinity.xls, dissolved oxygen.xls. This format has been used so that the information can be updated easily and transferred into a database format. The system variables considered were salinity, turbidity/suspended solids and dissolved oxygen. For the microalgae temperature and pH were also included. Inorganic nitrogen, phosphate and ammonium were the nutrients considered. The effect of toxic substances (e.g. trace metals) was only considered for invertebrates and fish.*

<u>Project 3</u>: Responses of the biological communities to flow variation and mouth state in two KwaZulu-Natal temporarily open – closed estuaries. (Importance of the river-estuary interface zone in temporarily open / closed estuaries – Volume 2).

1. Literature review.

The literature review by Perissinotto et al. *on the "Ecology of South African temporarily open/closed estuaries: a review of current knowledge" was e-mailed to the CERM members for input. After the inclusion of relevant inputs this review paper was submitted to the Journal of Marine Systems for possible publication. The aim of the synthesis was to provide an overview of the existing knowledge on the major biological, physical, chemical and management issues that affect dynamics of these key but highly threatened ecosystems.*

2. Quantifying the response of the biota to changes in flow and mouth condition.

The Consortium for Estuarine Research and Management held a workshop on 24 October 2001 at the University of Natal, Durban, to discuss the terms of reference and hypotheses for the study, and to select a research team. The Mdloti and Mhlanga estuaries were chosen as the study sites and the research team consisted of participants from the University of Natal, Durban and the CSIR, Durban with Prof Renzo Perissinotto as the project leader. Monthly sampling was conducted from March 2002 to March 2003 for physico-chemical, microalgal and zooplankton measurements. Macrobenthos, fish and birds were sampled quarterly. An unusual rainfall pattern was observed during the year of this study, with anomalous high precipitation recorded during the winter (July 2002). In terms of breaching events, 13 of these were recorded at the Mhlanga and 9 at the Mdloti. The study showed that provided the flows are sufficient to sustain residence times of the order of 1-2 days, then the mouth will stay open.

The original hypotheses proposed for the study were that during the open phase of the mouth, a river-estuary interface zone is established that is characterized by higher biodiversity and biomass compared to the closed phase of the mouth and that biotic communities in the REI zone during the open phase of the mouth are distinct from downstream communities. These hypotheses were revised as KwaZulu-Natal estuaries are perched which prevents a distinct REI zone from forming.

The study showed that during the open phase there was strong freshwater inflow followed by low water levels. Highest biodiversity and biomass occurred when the mouth was closed (phytoplankton, microphytobenthos, zooplankton). The maximum microphytobenthos biomass (601 mg chl-a.m^{-2}) occurred in December 2002, after a period of closure of about 15 days. Zooplankon (copepod nauplii) attained peak levels within 2 to 4 weeks following a rainfall event, with temperature and state of the mouth being the main factors controlling the delay in this response. The study hypotheses for macrobenthos (i.e. a different fauna develops after the open state with filter feeders becoming dominant due to food availability three months after mouth closure) could not be tested because there was no unbroken three month closure period. The open mouth state did not increase the diversity of birds as much of the areas exposed after breaching became supratidal rather than intertidal and the drier habitats did not support any of the small, shallow-burrowing prey species.

Many of the study hypotheses were focussed on freshwater input and its importance in introducing nutrients and stimulating biotic response. However both the Mdloti and Mhlanga estuaries receive treated sewage effluent and therefore available nutrients were never a problem. Indeed the Mhlanga Estuary was found to be eutrophic. Phytoplankton biomass was 303 mg chl-a m^{-3} in the lower reaches of the estuary in October 2002, this is the highest reported for any South African estuary. The hypotheses also focussed on the importance of mouth opening in stimulating biotic response. In both estuaries mouth breaching occurred regularly due to increased flow from the sewage plants, and the question is rather not how frequently the mouth should open in a year, but rather how long the mouth should remain closed. Frequent breaching results in a disturbed system where biomass can never build up.

Another important hypothesis that was tested was that overtopping of the berm is important for the recruitment of estuarine dependent invertebrates and fish. Overtopping was indirectly demonstrated, when six 20 mm Diplodus sargus were netted in the Mhlanga while storm-swell overtopping was occurring (September 2002). Direct, conclusive evidence of recruitment through wave overtopping was obtained at the Mhlanga where juveniles of three fish species were netted in the incoming waves at the peak of the spring high tide in August and September 2003. The berm overtopping study showed minimal recruitment of invertebrates during the closed phase as only a few ghost crabs, mole crabs, nematodes and *siphonophorans were collected during the six sampling exercises at Mhlanga and the two exercises at the Mdloti Estuary.*

Recruitment of juvenile fish and invertebrates via berm overtopping appears to be relatively small at the estuaries studied. Coarse sediments with steep, exposed beaches characterize them. Conversely KZN South Coast estuaries e.g. Lovu, Msimbazi, have finer sediments, and less steep beach slopes. Here recruitment via overtopping was observed repeatedly and on a much larger scale than at the Mhlanga/Mdloti estuaries.

One component of the study that was not addressed was the short-term responses of microalgae and zooplankton to the closing of the mouth. The objective of this investigation was to provide information on the time-scale of recovery of the primary producers and consumers after a mouth-breaching event and re-closure of the estuary. By December 2003 the mouth had not breached and therefore this component of the study could not be included in the final report

Capacity building

Resource Directed Measures (RDM) workshops

The objective of this component of the programme was to organise workshops on the RDM methodology to increase the pool of experts capable of doing this work. A workshop was held in Port Elizabeth in June 2002. Participants included Department of Water Affairs and Forestry personnel from the Port Elizabeth, East London, King Williams Town and Cradock offices and from the RDM office in Pretoria. This was also a training workshop for the Thukela RDM study participants.

Two workshops were held in 2003, one in Durban and one in Cape Town. Participants included DWAF personnel, Umgeni Water, Durban Metro, KZN Wildlife, CSIR, Ninham Shand, City of Cape Town, Western Cape Nature Conservation, Department of Environmental Affairs, Namibian Ministry of Agriculture, Water and Rural Development and UPE. From the feedback received it is obvious that participants were very appreciative of these training workshops. The DWAF RDM directorate will undertake training in the future. In the short-term when no formal training was occurring the Amazon project filled an important gap. The feedback from the RDM workshop participants as well as the training materials have been passed on to DWAF so that it can be incorporated in the design of future training workshops.

Mdloti and Mhlanga Rapid Resource Directed Measures studies

A rapid RDM study commissioned by DWAF was completed on the Mdloti Estuary in February 2002 and on the Mhlanga Estuary in April 2003. Representatives from local authorities attended the workshop as well as students and trainees from the University of Natal Durban and CSIR Environmentek. This study provided a first estimate of the freshwater requirements of the Mdloti and Mhlanga estuaries.

These reserve studies could not have been completed without the research and understanding gained from the Amazon research project "Responses of the biological communities to flow variation and mouth state in temporarily open / closed estuaries." In addition these reserve studies have improved capacity at the UND and Environmentek (Durban). Researchers now have the confidence and capacity to undertake other estuary RDM studies in the future.

Technology transfer

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Scientific papers (pending)

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SUMMARY OF NEW KNOWLEDGE GENERATED FROM THE RESEARCH

PROJECT 1 Improving the biodiversity importance rating of SA estuaries

Comprehensive field counts of all the birds of the Transkei coast were completed. Invertebrates were sampled in 16 warm temperate estuaries and from these data a predictive model for invertebrate community attributes was developed based on broadscale estuary characteristics. New knowledge on minimal sampling strategies for different estuarine taxa were proposed and an updated importance rating of all South African estuaries was produced.

PROJECT 2 Quantifying water quality changes that affect different taxa

The objective of this component of the programme was not to generate new knowledge but rather to collate existing data so that information gaps could be identified. Data were mainly sourced from overseas studies indicating the lack of detailed studies on South African taxa and their water quality responses.

PROJECT 3 Responses of the biological communities to flow variation and mouth state in temporarily open / closed estuaries

Mouth condition

This research has made an important contribution to the knowledge of processes controlling the dynamics of temporarily open/closed estuaries. The results seem to support others which show that TOCEs experience a period of biological rejuvenation some time after natural breaching, followed by a period of maximal productivity shortly after the re-closure of the mouth (Whitfield 1992, Nozais *et al.* 2001). For microphytobenthos this was approximately two weeks and for zooplankton it was two-four weeks after mouth closure.

New insights on the process of mouth breaching have been gained. Observations have shown that high seepage flows through the sandbar to the sea cause erosion of the beach face and slumping of the sandbar. Wave overtopping was important in lowering the height of the berm, which promoted overflow and scour.

Recruitment

The research has shown that overtopping of the berm can result in fish recruitment but minimal invertebrate recruitment. The dominance of estuarine dependent species in the catches, almost to the exclusion of other marine species, demonstrates a remarkable ability of the small larvae to manoeuver and orientate in the surfzone for the purpose of recruitment to the estuaries. *Rhabdosargus holubi* juveniles were of a similar size, approximately 28 days post hatch. This single cohort in the catches begs the question whether adult spawning, and hence juvenile arrival at the beaches, for recruitment to estuaries might be linked to the lunar cycle in this species.

Sewage discharge

The study has highlighted the problems associated with sewage discharge to temporarily open / closed estuaries e.g. algae blooms and low oxygen concentrations. Sewage discharge also increases base flow, thus increasing the water level in the estuary causing premature mouth breaching. The Mhlanga Estuary was shown to be in an unstable state due to the frequent breaching of the mouth. The greater abundance and biomass of fish in the Mdloti Estuary compared to the Mhlanga Estuary was related to the greater water retention within the system (i.e. longer periods of mouth closure).

RECOMMENDATIONS FOR TECHNOLOGY TRANSFER OF RESULTS

PROJECT 1 Improving the biodiversity importance rating of SA estuaries.

The updated biodiversity importance ranking of all SA estuaries will be used by DWAF in future RDM studies. The importance rating of a system is an important step in the RDM process that is used to set the recommended ecological category of the estuary.

PROJECT 2 Quantifying water quality changes that affect different taxa.

The database on the CD will be available to be used in all RDM studies. In this way people will utilise the available information as well as provide further data for the improvement of the current database.

PROJECT 3 Responses of the biological communities to flow variation and mouth state in two KwaZulu-Natal temporarily open / closed estuaries

The results from this project were used to set the ecological flow requirements of the Mdloti and Mhlanga estuaries (DWAF RDM studies). Transfer of results has thus taken place and indeed the research increased the confidence of the RDM assessments. This project has provided good baseline data for future monitoring of the Mdloti and Mhlanga estuaries. Important new knowledge was generated in this project that had direct application to management problems.

This has already been implemented in the re-direction of sewage effluents away from the Mhlanga and Mdloti estuaries. The new water treatment system planned by Ethekweni Water Services (Durban Metro) will now discharge to the Mgeni Estuary. The study has highlighted the problems of sewage discharge to temporarily open / closed estuaries (i.e increased flow and unnatural breaching). In particular the eutrophication processes at Mhlanga Estuary can serve as a case study for testing responses in similarly affected estuaries.

RECOMMENDATIONS FOR FUTURE RESEARCH

PROJECT 1 Improving the biodiversity importance rating of SA estuaries.

Future studies should address the following needs:

1. Completion of the habitat area and total area estimates for all estuaries. This includes checking the existing area data in cases where there are doubts about accuracy. It would be particularly useful if the estuaries were digitized in a geographic information system (GIS).
2. Sampling of invertebrates from a lot more systems. This would improve the accuracy of the models produced in this study, and would also provide useful baseline data for future RDM assessments, especially rapid or desktop assessments.
3. An in depth study of the trade-off between sampling effort and data quality for estuarine fishes, allowing an evaluation of the Harrison data used in the index versus use of the other more comprehensive data-sets, and implications for the fish importance rating.
4. A total recount of birds of all the South African estuaries to update the counts which are mostly over 20 years old.
5. A sensitivity analysis of the weightings in the importance index. Testing of the index is needed, to investigate the effects of, for example, only retaining the biotic elements, or reducing the weighting of the zonal type rarity index.

PROJECT 2 Quantifying water quality changes that affect different taxa.

The collation of available information in one database can be used to identify gaps. Overall the study has shown that there is little ecophysiology and ecotoxicology data for most of South Africa's estuarine biota. Thus there is plenty of scope for detailed studies on individual taxa.

PROJECT 3 Responses of the biological communities to flow variation and mouth state in two KwaZulu-Natal temporarily open / closed estuaries

This study has emphasized the need for medium / long term monitoring programmes for key estuaries. Climate change and an increase in the frequency of droughts will have a major influence on the mouth dynamics of temporarily open / closed estuaries. It is recommended that a programme on the effects of climate change on estuaries be initiated. Research into the mechanisms of mouth breaching is necessary particularly in light of the response of the mouth to slumping as a result of outflow of estuary water. Investigations of this nature should be extended to temporarily open / closed warm temperate estuaries.

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TABLE OF CONTENTS

PROJECT 3 (VOLUME 2):

RESPONSES OF THE BIOLOGICAL COMMUNITIES TO FLOW VARIATION AND MOUTH STATE IN TWO TEMPORARILY OPEN / CLOSED KWAZULU-NATAL ESTUARIES

LIST OF ABBREVIATIONS

ANOVA: Analysis of Variance
Chl-a: Chlorophyll-a
CERM: Consortium for Estuarine Research and Management
CSIR: Council for Scientific and Industrial Research
CWAC: Co-ordinated Water Bird Counts
DIN: Dissolved Inorganic Nitrogen
DIP: Dissolved Inorganic Phosphorous
DO: Dissolved Oxygen
DW: Dry Weight
DWAF: Department of Water Affairs and Forestry
EMDS: Estuarine Marine Dependent Species
FDC: Flow Duration Curve
KZN: KwaZulu-Natal
MAE: Mean Annual Evaporation
MAP: Mean Annual Precipitation
MAR: Mean Annual Runoff
MPB: Microphytobenthos
MSL: Mean Sea Level
PAR: Photosynthetically Available Radiation
RDM: Resource Directed Measures
SD: Standard Deviation
SL: Standard Length
TOCE: Temporarily Open/Closed Estuary
UPE: University of Port Elizabeth
WLS: Water Level Sensor
WR90: Surface Water Resources of SA, 1990 (Midgley *et al.* 1994)
WRC: Water Research Commission
WWTW: Waste Water Treatment Works

LITERATURE REVIEW

Ecology of South African temporarily open/closed estuaries: a review of current knowledge

ABSTRACT

The vast majority (71%) of South Africa's 258 estuaries are currently temporarily open/closed (TOCE), remaining closed to the ocean during the dry season while exhibiting an open phase with duration proportional to the intensity of runoff during the rainy season. The aim of this synthesis is to provide an overview of the existing knowledge on the major biological, physical, chemical and management issues that affect the dynamics of these key but highly threatened ecosystems.

The state of the mouth is perhaps the single most important factor that drives the ecology of TOCEs. Mouth dynamics is in turn determined by the water balance which is largely controlled by river inflows. There have been few detailed studies of the water balance for South African estuaries, so that the relationship between flows and mouth dynamics remains specific to the individual case studies that have been reported. An attempt to assimilate some of the available information in terms of residence times is attempted in this review, although the quality of the data is probably low.

The regular alternation of open and closed phases makes the physico-chemical dynamics of TOCEs a great deal more fluctuating and complicated than that of permanently open estuaries. Mouth closure cuts off all tidal exchanges with the ocean, resulting in prolonged periods of stagnation during which salinity and temperature stratification may occur, along with dissolved oxygen and nutrient depletion in the water-column. During mouth breaching, on the other hand, thorough mixing is accompanied by scouring of the sediment and a dramatic increase in the silt load and turbidity of the water.

Microalgae are key primary producers in TOCEs, and while phytoplankton biomass in these systems is lower than in permanently open estuaries, microphytobenthic biomass has been found to be much higher in TOCEs than in their permanently open counterparts. Productivity, however, seems to indicate an opposite trend to that of biomass, with water-column values exceeding those of benthic microalgae.

During the closed phase of TOCEs, absence of tidal currents, cleaner water and greater light penetration can result in the proliferation of submerged macrophytes. These may be brackish or freshwater types depending on the salinity. The commonest emergent macrophyte is the reed *Phragmites australis.* Loss of tidal action and high water levels, however, also result in the absence or disappearance of mangroves and adverse effects on salt marsh vegeration.

Zooplankton are the main primary consumers both in the water-column and within the upper sediment microlayer, due to their marked diel migrations. The prolonged period of mouth closure that TOCEs experience generally leads to poor levels of zooplankton taxonomic diversity, but also to the biomass build-up of a few dominant species.

There is a paucity of up to date information on the meiofauna of TOCEs. Recent data from the Mdloti Estuary in KZN indicate greater meiobenthic abundances during closed phases and a dominance of the community by nematodes.

The macrobenthos of TOCEs is typically a sub-set of the community found in large, permanently open systems where intertidal areas support a variety of species and an open mouth allows more migration. Macrobenthic densities in TOCEs are frequently higher than in open systems, but these numbers often reflect an abundance of small, pioneering species and not necessarily a high biomass.

The dominance of estuarine and estuarine-dependent marine fish in any TOCE is an indication of the viable/important nursery function of that system. *Rhabdosargus holubi* and several Mugilidae species exhibit an extended spawning season and the ability of juveniles to recruit into TOCEs not only when the mouth opens, but also during marine overwash events.

Concerning birdlife, waders are generally absent or uncommon in TOCEs because of the absence of intertidal feeding areas. Most species that do occur are piscivorous and include herons, egrets, kingfishers, cormorants, darters, pelicans and fish eagles which catch a variety of fish species either from the surface or by diving and swimming underwater.

Under the assumption that the current concept of sustainable management for TOCEs is aimed at protecting their ecological integrity, biodiversity and nursery function, the key issues involved in their management are: artificial breaching; freshwater abstraction; catchment land use; floodplain encroachment; sandwinning operations; utilization of bioresources and recreational activities.

1) Introduction

The South African coast has 258 estuaries and 183 of these (approximately 71%) are currently classified as temporarily open/closed systems, or TOCEs (Whitfield, 2000). The vast majority of these systems (248) are situated along the eastern (Indian Ocean) seaboard. Conversely, the west or Atlantic coast exhibits only 10 proper estuarine systems, although Heydorn & Tinley (1980) identified a number of other small non-estuarine systems on this coast. Many of these systems comprise dry riverbeds that only carry water at times of exceptional rainfall (Heydorn, 1991).

In terms of biogeography, South African estuaries are subdivided into 3 main climatic regions: (1) cool temperate from the Orange River to the Cape Peninsula; (2) warm temperate from False Bay to the Mbashe River; and (3) subtropical from the Mbashe to Kosi Bay. There is thus only one TOCE in the cool temperate region, while the warm temperate and the subtropical regions exhibit respectively 84 and 93 such systems (Whitfield 2000).

A key feature of TOCEs is their small river catchments (Whitfield 1992). For example, in the subtropical region about two-thirds of the TOCEs have catchments smaller than 100 km^2. As a result, during the dry season and under low river flow conditions they are closed off from the sea by a sandbar (beach berm), which forms at the mouth. Flow rates during this period are generally low, say of order 20% of the mean annual runoff (MAR) (Huizinga & van Niekerk, CSIR 2002). Following periods of high rainfall and freshwater runoff, the water level inside the estuary may rise until it equals or exceeds the height of the sandbar at the mouth (Whitfield 1992, Wooldridge & McGwynne 1996). Breaching may then occur with a rapid drop in the water level, often exposing large areas of substratum that may have been submerged for long periods and colonised by a rich community of algae and animals. River conditions may briefly dominate the estuary during breaching events, when flow rates significantly exceed MAR (Huizinga & Van Niekerk, CSIR 2002). However, once the freshwater inflows again decrease to of order the MAR or lower, a typical estuarine open phase can occur, with regular tidal exchange and seawater penetration into the middle and upper reaches. The tidal prism often remains small relative to the estuary's volume in the case of South African TOCEs (Whitfield 1992). The open phase ends when the sandbar at the mouth is regenerated by along-shore and cross-shore sand movement in the surf-zone (usually within weeks). This leads to another closed phase during which the only seawater inflow is provided by wash-over at the peak of the spring tide or during storm surges. Depending on the climatic conditions and rainfall patterns in the catchment areas, closure periods may vary naturally from days to months or even years. This extremely dynamic situation has important implications for the full range of physical and chemical parameters of TOCEs and ultimately for their ecological functioning. The main physico-chemical factors in this regard are temperature, salinity, sediment structure, light penetration, oxygen concentration and nutrients availability.

2) Hydrodynamics: residence times and inlet dynamics

Changes to the flows into estuaries can have a significant impact on mouth dynamics and therefore on the overall functioning of estuarine systems. The increasing pressures on water resources have generally reduced the amount of flow into estuarine systems that implies an increasing proportion of time that TOCEs are closed. The implications for water quality and the general ecological health of these systems can be severe, although they are not fully understood. Case studies are reported by De Decker (1987), Grange et al (2000), Reddering (1988), Schumann & Pearce (1997), Whitfield & Wooldridge (1994), Whitfield & Bruton (1989), Whitfield (1993),. The interrelationships between flow and estuarine morphology are discussed in reviews by Harrison, Cooper & Ramm (2000), Cooper, Wright & Mason (1999), and Cooper, (2001). Schumann, Largier & Slinger (1999) provide a general review of various aspects of estuarine hydrodynamics in South Africa. The relationship between flow and the mouth state of TOCE systems in South Africa is of particular relevance to the management of these ecosystems and is the focus of the present review. Several case studies have provided information on the flow rates associated with specific inlet conditions. Obviously low flows are generally associated with closed mouth conditions, while relatively high flows (and in particular flood events) are associated with the opening of the mouth. However, actual flow magnitudes are specific to each case study estuary, and have little or no relevance to others. "What defines a "high" flow rate? i.e. what is the relevant scaling for these flows which could be used to establish general principles that may be applicable to all estuaries of broadly similar type? For example, it seems obvious that the size of an estuary influences the critical flow rate required to open the mouth. Such scalings, that require more detailed consideration of the water balance of estuarine systems, have not yet been established. Some broad (mainly qualitative) principles concerning the issues of flow and mouth dynamics of SA estuaries have emerged from case studies, and are discussed by Huizinga (2000), Huizinga and Van Niekerk (2002). For example, in the case of South African estuaries, the energetic wave climate means that direct wave action and associated cross-shore sediment transport plays a relatively larger role in mouth dynamics in the case of smaller TOCE systems.

The hydrological functioning of a TOCE system can be conceptualised in terms of a dynamic storage system with variable inputs and outputs as depicted schematically in figure 1. Freshwater inputs comprise mainly of the catchment's runoff. Outputs comprise losses due to evaporation/transpiration, seepage through the berm, and variable outflows/inflows to the sea depending on the state of the mouth or inlet. A closed estuary would breach naturally when water levels rise to equal or overtop the frontal berm (or sandbar) that separates it from the sea. The maintenance of an open state depends on a complex balance between sediment removal by scouring and sediment deposition by wave action. Once an opening is formed, tidal exchange flows can also contribute to the sediment dynamics at the mouth. If the estuary is "perched" at elevations above the mean sea level, then tidal prisms are usually small and the estuary may essentially drain most of its storage capacity when it breaches.

If the storage volume of the system depicted in figure 1 is S(t), then a simple water balance implies that

$$\frac{dS(t)}{dt} = I(t) - O(t)$$

where $I(t)$ and $O(t)$ are the grouped inputs and outputs from the system respectively. If S_0 is some reference storage, for example the maximum storage capacity of the closed estuary prior to breaching the frontal berm, then a residence time may be defined from the water balance as

$$\frac{dS(t)}{dt} = \frac{S_0}{T_R} = I(t) - O(t)$$

whence

$$T_R = \frac{S_0}{I(t) - O(t)}$$

Residence time is an important indicator of the hydrodynamic functioning of an estuary and as such can have important implications on the ecological functioning of the system. As defined above, it can be interpreted as the time scale between breaching events, given a certain inflow/outflow scenario. TOCEs with generally short residence times, would regularly overtop and/or breach the frontal berm and therefore be open to the sea for a longer proportion of time than those with relatively longer residence times. Actual values of T_R would vary seasonally (due to seasonal variations in rainfall, runoff and evaporation), and be related directly to water levels (due to their effect on seepage losses). A detailed water balance analysis is required to accurately determine actual residence times. Moreover, the available information on South African TOCEs is not sufficient to estimate the outflows (e.g. by seepage) during closed mouth conditions. To obtain some insight into the water balance characteristics of typical South African TOCE systems an indication of typical residence times may be obtained by ignoring outputs from the estuaries and using the mean annual runoff (MAR) as representative of the inputs. Whence an average residence time may be defined as

$$\overline{T}_R = \frac{S_0}{MAR}$$

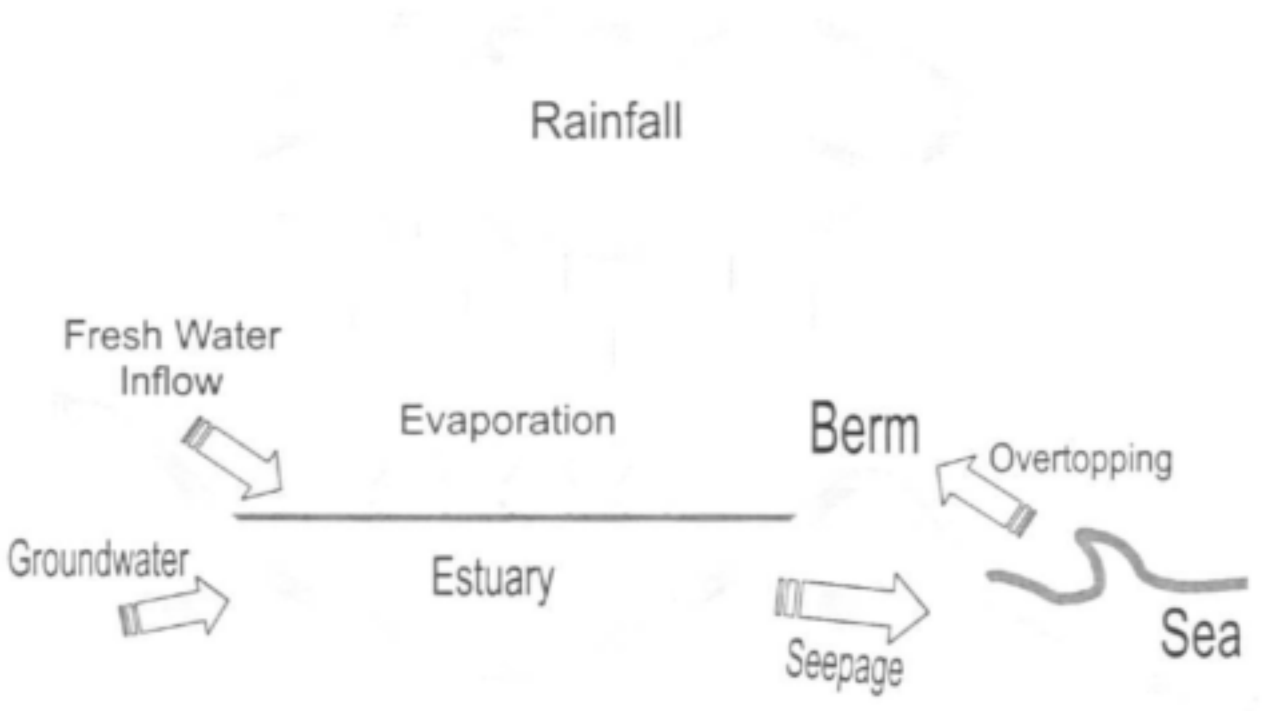

Figure 2 : Conceptual water balance for a temporary open/closed estuary

The surface area of an estuary, and the maximum water level changes (or average depth) can be used to estimate S_0 while MAR estimates are available for all South African catchments from Midgely *et al.* (1994). A comprehensive compilation of KZN estuary data, including estuarine morphology (average depth, surface area, etc) and mouth statistics, is available in the reports by Begg (1978, 1984a) and Ramm et al (1985 – 1986). Heydorn & Tinley (1980) and Heydorn & Grindley (1983) give similar, but less detailed information on Cape estuaries[1]. The available data for KZN estuaries has been used to estimate average residence times for TOCEs in that region, and to investigate any relationship between residence time and the proportion of time that the estuaries are closed. A synthesis of the data is plotted in figure 3.

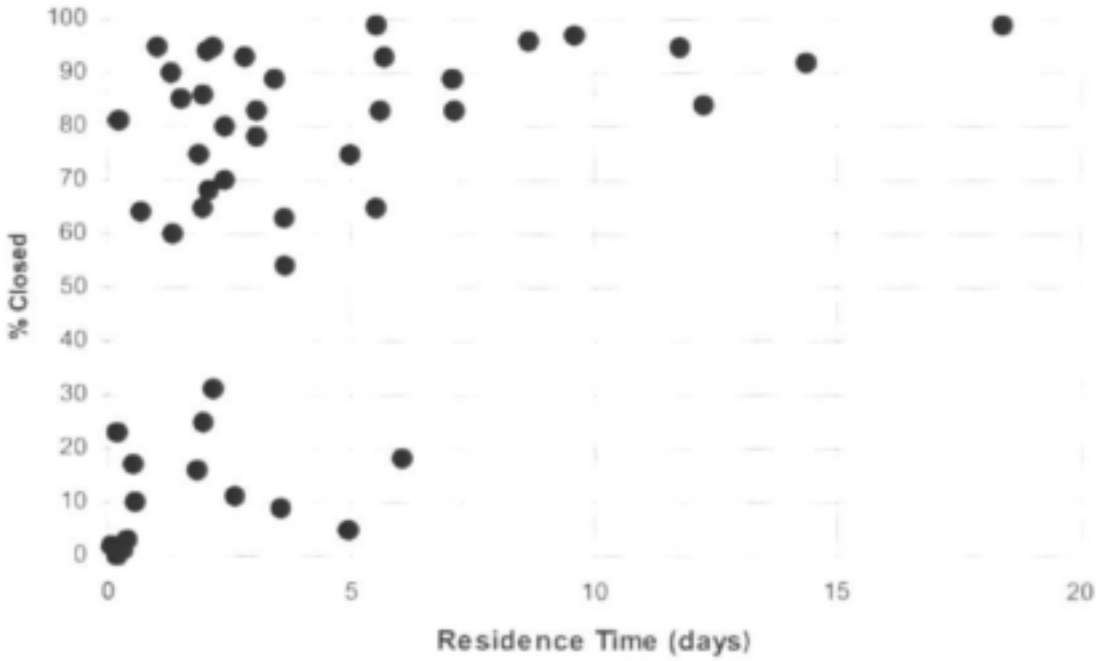

Figure 3 The relationship between average residence times and mouth state for KZN estuaries

It can be seen from figure 3 that, as expected, there is a general trend where longer average residence times are associated with estuaries that have a higher proportion of closed mouth conditions. Estuaries with average residence times, based on MAR, longer than about 5 days are typically closed for 80% or more of the time. There is however considerable scatter in the data for cases where average residence times are shorter than 5 days i.e. there is only weak (if any) indication of an association between residence time and mouth statistics in these cases. The scatter may in part be due to inconsistencies in the way the data were estimated e.g. for the estuary area and average depths). Another factor could be that the MAR is not representative of seasonal flows,

[1] The source book by Whitfield (2000) contains a summary of available scientific information for all SA estuaries, while Alcock (1999) lists available information on water resources.

particularly for small catchments that have highly variable runoff. However the most important factor is likely to be the exclusion of any accounting for the outflows in the definition of average residence times. Most TOCEs in KZN are perched with maximum water levels 2 – 3m above mean sea level. Seepage outflows through the berm can be particularly important in such cases and can significantly increase actual residence times.

Jezewski and Roberts (1986, see also Alcock, 1999) carried out the first analysis of the freshwater requirements of estuarine systems in SA. Their report contains an analysis of the annual evaporative requirements (i.e the freshwater inflow required to balance direct losses to evaporation), as well as a "flooding" requirement. The latter, defined as a 2-year return period flood, was suggested as the flow required to flush the systems, including breaching the sandbar at the mouth where applicable. It is interesting to note that the total freshwater requirements for the estuaries included in that study were found to be generally less than 5% of the natural MAR. These estimates are significantly lower than those inferred by more recent detailed and holistic analyses incorporating aspects of the biological functioning of the estuaries. An example is a recent study of the Mdloti Estuary (CSIR, 2002). In that study, the effects of flow on mouth state was a key element in determining the biological responses to flow changes. Four mouth states were identified for the estuary as indicated in the Table 1. Also included in the table are the residence times corresponding to each state. It can be seen from Table 1 that closed or semi-closed mouth conditions correspond to residence times longer than a week, while open mouth conditions occur when the residence times are of order a few days. The generality of these estimates remains to be tested.

STATE	DESCRIPTION	FLOW (m^3/s)	FLOW/MAR	$T_R = S_0/FLOW$ (days)
1	Closed, no tidal prism	0 – 0.3	0 – 0.15	> 40
2	Semi-closed, no tidal prism, some outflow	0.3 – 2.0	0.15 – 0.85	7 - 40
3	Open, with tidal prism	2.0 – 5.0	0.85 – 2.20	2 -6
4	Open, no tidal prism, river dominated	> 5.0	> 2.20	< 2

Table 1. Mouth states for the Mdloti Estuary (after CSIR, 2002)

Sedimentation of estuaries is an important issue that is intrinsically related to hydrodynamics. Specific case studies are described e.g. by Cooper (1989, 1990, 1993a,b,c, 1994a,b, 2001), Cooper et al (1990, 1999), Reddering (1988), Reddering et al (1990). Alcock (1999) reviews further information on sedimentation.

The breaching of TOCEs is thought to play an important role in maintaining an approximate equilibrium with respect to sedimentation. Changes to breaching patterns due to flow changes can be expected to have severe impacts on the morphology of the estuaries concerned.

The wave climate at an estuary's location, and associated cross-shore and along-shore sediment transport, are key factors in the opening/closing dynamics of TOCEs. The timescales associated with these sedimentation processes, when compared to the residence time of the estuary (as discussed above), may be expected to determine the duration of the open phase (see also Cooper, 2001, Huizinga, 2000). However, quantitative information on these time scales is not readily available. In the region of the Mdloti estuary referred to above, northward along-shore sediment movement has been estimated to exceed 1000 m^3/day (see e.g. Schoonees, 2000). Allowing several tidal cycles for cross-shore redistribution of the sediment, it would appear that the sedimentation time scale is short, perhaps 2 - 4 days. This corroborates the results discussed above (see Table 1).

In summary, case studies in South Africa have attempted to identify critical flow regimes associated with different inlet conditions for individual TOCE systems. These studies have provided valuable experience and insight into the inlet hydrodynamics of small TOCEs . However, this has not been generalized into universal scaling relationships for deducing the influence of flow on mouth state for TOCEs. This requires detailed analysis of the water balance of typical systems and remains an important topic for future research.

3) Physico-chemical environment

The material used in this review is listed in Table 2 below. Only substantial investigations, covering at least some seasonal/annual surveys are included in the table. Much of these data are, however, very old, often inaccurate and even obsolete. Snapshot measurements, taken during a 1992-1999 survey by Harrison et al. (2000) in virtually all South African TOCEs, are not included in the table, although some data derived from these are discussed in the review. Information from the Nhlabane system in northern KZN and from the Bot River in the south-western Cape is included in the discussions despite the fact that these are classified as coastal lake systems by Whitfield (2000). This was done because of similarities to TOCEs in some of the biotic features, the amount of information available on these two systems and the strong impression that knowledge of these larger systems would enhance understanding of the typically smaller TOCEs.

System	Reference
Diep/Rietvlei (cool temperate)	Grindley & Dudley 1988, Hughes *et al.* 1993
Wildevoëlvlei (cool temperate)	Heinecken 1985
Silwermyn (warm temperate)	Heinecken 1982
Sand (warm temperate)	Morant & Grindley 1982
Eerste (warm temperate)	Grindley 1982
Lourens (warm temperate)	Cliff & Grindley 1982
Sir Lowry's Pass (warm temperate)	Heinecken *et al.* 1982
Rooiels (warm temperate)	Heinecken 1982
Buffels, Oos (warm temperate)	Heinecken *et al.* 1982
Onrus (warm temperate)	Heinecken & Damstra 1983
Hartenbos (warm temperate)	Bickerton 1982
Groot Brak (warm temperate)	Morant 1983, Slinger *et al.* 1995, Allanson & Winter 1999
Piesang (warm temperate)	Duvenage & Morant 1984
Groot, Wes (warm temperate)	Morant & Bickerton 1983
Seekoei (warm temperate)	Esterhuysen 1983, Bickerton & Pierce 1988, Whitfield and Bruton 1989
Kabeljous (warm temperate)	Bickerton & Pierce 1988
Boknes (warm temperate)	Fromme 1986
Kasuka (warm temperate)	Fromme 1986
Kleinemonde, Wes (warm temperate)	Badenhorst 1988, Day 1981
Kleinemonde, Oos (warm temperate)	Badenhorst 1988, Day 1981
Quinira (warm temperate)	Wiseman *et al.* 1993
Nyara (warm temperate)	Perissinotto *et al.* 2000; Walker *et al.* 2001
Quko (warm temperate)	Burns *et al.* 1988
Mtamvuna (subtropical)	Day 1981, Begg 1984
Zolwane (subtropical)	Begg 1984
Sandlundlu (subtropical)	Begg 1984
Ku-Boboyi (subtropical)	Begg 1984
Tongazi (subtropical)	Begg 1984
Kandandhlovu (subtropical)	Begg 1984
Mpenjati (subtropical)	Begg 1984, Perissinotto *et al.* 2002, Kibirige2002
Umhlangankulu (subtropical)	Begg 1984
Kaba (subtropical)	Begg 1984
Mbizana (subtropical)	Begg 1984
Mvutshini (subtropical)	Begg 1984
Bilanhlolo (subtropical)	Begg 1984
Uvuzana (subtropical)	Begg 1984
Kongweni (subtropical)	Begg 1984
Vungu (subtropical)	Begg 1984
Mhlangeni (subtropical)	Begg 1984
Zotsha (subtropical)	Begg 1984
Boboyi (subtropical)	Begg 1984
Mbango (subtropical)	Begg 1984
Mtentweni (subtropical)	Begg 1984
Mhlangamkulu (subtropical)	Begg 1984
Damba (subtropical)	Begg 1984
Koshwana (subtropical)	Begg 1984
Intshambili (subtropical)	Begg 1984

Mzumbe (subtropical)	Begg 1984
Mhlabatshane (subtropical)	Begg 1984
Mhlungwa (subtropical)	Begg 1984
Mfazazana (subtropical)	Begg 1984
Kwa-Makosi (subtropical)	Begg 1984
Mnamfu (subtropical)	Begg 1984
Mtwalume (subtropical)	Begg 1984
Mvuzi (subtropical)	Begg 1984
Fafa (subtropical)	Day 1981, Begg 1984
Mdesingane (subtropical)	Begg 1984
Sezela (subtropical)	Begg 1984
Mkumbane (subtropical)	Begg 1984
Mzinto (subtropical)	Begg 1984
Mzimayi (subtropical)	Begg 1984
Mpambanyoni (subtropical)	Begg 1984
Mahlongwa (subtropical)	Begg 1984
Mahlongwana (subtropical)	Begg 1984
Ngane (subtropical)	Begg 1984
Umgababa (subtropical)	Day 1981, Begg 1984
Msimbazi (subtropical)	Begg 1984
Lovu (subtropical)	Begg 1984
Little Manzimtoti (subtropical)	Begg 1984
Manzimtoti (subtropical)	Begg 1984
Mbokodweni (subtropical)	Begg 1984
Mgeni (subtropical)	Begg 1984,
Mhlanga (subtropical)	Whitfield 1980a,b, Begg 1984,
Mdloti (subtropical)	Begg 1984, Blaber *et al.* 1984, Mundree 2001, Nozais *et al.*2001
Tongati (subtropical)	Blaber *et al.* 1984, Begg 1984
Mhlali (subtropical)	Begg 1984
Seteni (subtropical)	Begg 1984
Mdlotane (subtropical)	Begg 1984
Nonoti (subtropical)	Begg 1984
Zinkwasi (subtropical)	Begg 1984
Siyaya (subtropical)	van der Elst *et al.* 1999, Schleyer & Roberts 1987

Table 2. Physico-chemical studies carried out in South African TOCEs

Temperature

Seasonality and regional climate are clearly dominant factors affecting TOCEs temperature. Seasonal ranges are largest in the cool temperate region on the Atlantic coast and smallest in the subtropical part of the eastern seaboard. A typical annual temperature range would be from approximately 11 to 24 °C in a cool temperate TOCE (Rietvlei/Diep, Day 1981), 18 to 30 °C in a warm temperate TOCE (e.g. Nyara, Perissinotto et al. 2000) and from 19 to 28 °C in a subtropical TOCE (e.g. Mpenjati, Perissinotto et al. 2002). These ranges are similar to those observed in permanently open estuaries, although they tend to be generally smaller owing to the extremely slow freshwater inflow and the isolation from the sea that they experience during much of the year (Day 1981). Coastal upwelling events may be locally important in affecting the temperature of west and south coast TOCEs during their open phase (Schumann et al. 1999).

These factors also affect the formation of horizontal and vertical thermal gradients. The surface temperature gradients established along the estuary when the mouth is open do not persist after its closure, but the vertical gradients do, since density depends on salinity (Day 1981). So, during their closed phase, TOCEs virtually lack any temperature gradient along the length of the estuary, with differences of < 2 °C recorded from mouth to head both in winter and summer (Oliff 1976, Perissinotto *et al.* 2000, Perissinotto *et al.* 2002). On the other hand, vertical differences can be very substantial, and more marked than in an average permanently open system. This is because of poor vertical mixing, which leads to the establishment of vertical density gradients, whereby the denser high-salinity waters penetrated from the ocean during the open phase settle at the bottom of the water-column. On the eastern seaboard, this results in a winter/spring gradient (the typical period of the closed phase) of increasing temperature with depth. This indicates that the salinity gradients that are established during the summer open phase may persist for some time after the closure of the mouth, to determine the vertical density structure at the onset of the closed phase (Day 1981). Mixing and new fresh/seawater intrusions will eventually occur and dominate the density structure of the water-column through the rest of the open phase. On the west coast, the pattern is expected to be somehow reversed, as warmer waters develop during the summer closed phase over the more dense and cold ocean water penetrated during the winter/spring open phase (i.e. winter rainfall area).

Salinity

During their open, tidal dominated phase, TOCEs experience similar horizontal and vertical salinity gradients as those typically observed in permanently open estuaries. At times these may be very marked, with salinity stratification prevalent in deep TOCEs. For instance, in the Mtamvuna Estuary, a 15 $^{0}/_{00}$ isohaline normally lies at a depth of 1-2 m, with a halocline between 2-3 m and salinities gradually increasing to 30-35 $^{0}/_{00}$ below this layer down to the bottom, which may attain a maximum depth of approximately 10 m (Oliff, 1976, Day 1981). However, in broad shallow TOCEs (e.g. Mdloti), the fetch of the wind is generally sufficient to ensure complete mixing from surface to bottom and no salinity stratification is noted during either the closed or open phases (Blaber et al.1984, Grobbler et al. 1987, Mundree 2001). When TOCEs are breached, riverine conditions normally prevail for some time, with freshwater replacing all resident water types.

Salinities in closed estuaries will vary as in open systems, but typically on a time scale ranging from days to weeks rather than hours as in open systems where tidal effects are significant. Horizontal salinity gradients tend to break down once the marine influence is cut off, although some contrast between the mouth and head areas may remain, depending on seepage and overtopping at the bar and the magnitude of freshwater input at the head. Mesohaline (5-18‰) and oligohaline (0.5-5‰) conditions prevail during the closed phase in shallow systems and in the upper layers of deeper systems. In areas of high seasonal rainfall, such as the Western Cape and the KZN coasts, the sustained freshwater inflow into TOCEs results in salinities dropping to near freshwater conditions. On the KZN coast, this effect is compounded by the perched nature of TOCEs. Limnetic conditions (0.1-0.5‰) may then prevail through most of the closed phase, with salinities seldom exceeding 1‰ even in the lower reaches (e.g. Mdloti, Nozais et al. 2001). At the opposite end, in areas of low rainfall and high evaporation rate, or during periods of severe droughts, hypersaline conditions (> 40‰) may occur. An example of this was recorded in the Seekoei River in the Eastern Cape, where hyper-saline levels of up to 98‰ resulted in mass mortality of many of its fish species (Whitfield & Bruton 1989, Whitfield 1995).

Irradiance

Photosynthetic available radiation (PAR, 400-700 nm) in TOCEs may vary widely, in response to weather conditions, time of the day and suspended solids in the water. Maximum levels of surface irradiance recorded are in the range of 1638-2300 $\mu mol.m^{-2}.s^{-1}$ (Perissinotto *et al.* 2000, Nozais *et al.* 2001, Perissinotto *et al.* 2002). During the open estuarine phase, as a general rule, the suspended solids concentration in the water-column decreases from high to low tide and from the mouth to the upper reaches (Day 1981).

Light attenuation (K_d) in the water-column shows a highly significant and positive relationship with rainfall, obviously reflecting the increase in turbidity as a result of runoff from the catchment area (Nozais *et al.* 2001). During the closed phase of a TOCE, K_d may vary between 0.4 and 3 m^{-1} (Perissinotto *et al.* 2000, Perissinotto *et al.* 2002), with > 1% of the surface light intensity reaching the bottom of the water-column at any time. Conversely, when the opening of the estuary leads to heavy silt loading and a considerable increase in turbidity levels, K_d normally attains values of 8-29 m^{-1} (Perissinotto *et al.* 2000, Nozais *et al.* 2001). During this period, the percentage of surface light intensity reaching the bottom is often < 0.1%, thus resulting in aphotic conditions at the sediment surface as well as through part of the water-column. In other words, the depth of the euphotic zone drops to values much shallower than the estuary's total depth. This of course has important, negative implications for the photosynthetic efficiency of microalgae. It is, however, possible that a significant amount of light may be available to low-light adapted organisms even below the 0.1% level.

Available Secchi disc measurements from past surveys indicate that values may range from minima of 5-10 cm, during periods of mouth breaching, to maxima in excess of 3 m during the driest part of the closed phase (Day 1981, Begg 1984). More recent measurements of turbidity in units of nephelometric turbidity (NTU) show a wide range of values for TOCEs, from 0.2-7 NTU during the closed phase to 75-90 NTU at the onset of mouth breaching (Cooper et al. 1993, Harrison unpubl. data).

Sediment structure

Sediment scouring is one of the main features that characterize the onset of the open phase in TOCEs. Large amounts of sediment accumulate during the closed phase, mainly in the distributary channels, and this results in generalized shoaling. The process is reversed regularly by breaching events at the mouth, during which the fluvial flood is sufficient to remove a substantial proportion of the accumulated sediment, thereby deepening the estuarine channels and eliminating entire banks (Reddering & Rust 1990).

Estuarine sediments include organic and mineral particles of a wide range of sizes and composition. The organic content of a sediment tends to increase with the fineness of the deposit (Gray 1981). Marine sediment input is almost exclusively inorganic and occurs through three main processes: tidal inflow (e.g. formation of flood-tidal deltas); barrier overwash; and wind action (Cooper et al. 1999). Fluvial inputs, on the other hand, include a large organic component, mainly in the form of plant detritus, and an inorganic component of sand, silt and clay that is generally proportional to the rainfall rate, the degree of soil erosion in the area, the geology of the catchment and its agricultural management. Cape TOCEs often exhibit flood-tidal deltas, rather than fluvial sand and marine sand constitutes the bulk of estuarine sediments (Reddering & Rust 1990). In KZN, on the other hand, TOCEs sediments are dominated by river-borne deposits, as a result of the high relief of their drainage basins, the high rainfall of the region and their perched nature.

During the open phase, the sediment distribution in TOCEs is very similar to the pattern observed in any permanently open system. This means an abundance of coarse/medium sand (0.25-2 mm) towards the mouth and a predominance of silt (4-63 µm) and mud/clay (< 4 µm) in the middle and upper reaches (Day 1981). This pattern is well reflected in the results obtained in the Mhlanga Estuary in 1978 (Whitfield 1980b). At that time, the lower reaches of the Mhlanga were dominated by medium/coarse sand, the middle reaches by medium/fine sand and the upper reaches by very fine sand/silt (Whitfield 1980b). A significantly different situation was however observed in the nearby Tongati and Mdloti estuaries during the 1989/81 study of Blaber *et al.* (1984). While the lower and middle reaches of these two systems exhibited the same particle size composition of their Mhlanga counterparts, there was a virtual absence of silt in their upper reaches. The substratum was also relatively uniform throughout their length. This was attributed to an increase in the flow rate and frequency of mouth breaching in these two systems, compared to the Mhlanga (Blaber *et al.* 1984). Grains of different sizes are found in any sediment type, however well-sorted sediments tending towards an homogeneous type are typical of high current velocity situations. Conversely, poorly sorted, or highly heterogeneous sediments, are generally encountered in areas or during periods of low current activity (Gray 1981).

Dissolved Oxygen

In the absence of tidal currents and strong freshwater inflow, oxygen levels in closed systems may decline in bottom waters, particularly if stratification develops and if there is any organic accumulation on the bottom. During the closed phase, the oxygen concentrations in the deeper layers of TOCEs depend on the ratio of area to depth and the circulation due to the wind (Day, 1981). Systems like the Umgababa in KwaZulu-Natal and the Kleinmonds in the Eastern Cape are relatively broad and shallow and do not show any evidence of substantial oxygen depletion even in the deepest areas(but are these "deep"areas as deep as deep areas in other estuaries?.

This is so because wind action is sufficient to prevent the formation of thermoclines and the water is well oxygenated to the bottom (Oliff 1976, Day 1981, Begg 1984).

Narrower and more sheltered TOCEs may, however, experience anoxic or hypoxic conditions, particularly towards the end of the closed phase, after a prolonged period of stagnation. This situation is well illustrated by the results obtained during the Nyara Estuary study, in the Eastern Cape (Walker *et al.* 2001). In March, at the onset of the closed phase, the upper 2 m of the water column of the estuary was well oxygenated (4-7 $mgO_2.l^{-1}$), while lower oxygen conditions characterised the deeper water near the sediments (3-4 $mgO_2.l^{-1}$). This reached a low of less than 1 $mgO_2.l^{-1}$ at a depth of 3.5 m in September. Finally, oxygen concentrations rose to their highest levels (>9 $mgO_2.l^{-1}$) in November, during a flood event that caused breaching at the mouth (Walker *et al.* 2001). Similar situations have been reported for subtropical TOCEs in KwaZulu-Natal. For instance, in 1980-81 the Tongati Estuary experienced semi-anoxic waters (< 25% saturation) throughout its length during much of its closed phase, from May till October (Blaber *et al.* 1984). The nearby Mdloti during the same period experienced only a slight hypoxia (< 50%) in the bottom waters during January, May and August (Blaber *et al.* 1984). Saturation levels of < 20% have also been observed during periods of mouth closure in the Mtamvuna Estuary, below the 2m halocline (Oliff 1976, Begg 1984). Semi-anoxic conditions, with O_2 concentrations of ≤ 1 $mgO_2.l^{-1}$, have been recorded at the bottom of several KZN TOCEs during the Estuarine Health Index project, in October-November 1992 (Harrison *et al.* 2000). These include the Kandandhlovu, the Kongweni, the Little Manzimtoti, the Manzimtoti, the Mhlanga and the Tongati (Cooper et al. 1993).

Nutrients

The dynamics of macronutrient cycling in TOCEs are, in general, poorly understood since for only a few estuaries are there more than snapshot data available. Whole-year surveys with monthly sampling frequency have recently been carried out at the Mdloti, a largely modified estuary with significant input of treated sewage waters (Nozais *et al.* 2001, Mundree 2001), and at the Mpenjati, a relatively pristine system (Perissinotto *et al.* 2002, Kibirige 2002). Results indicate that macronutrient concentrations are generally higher in the Mdloti than in the Mpenjati Estuary, with a marked increase observed in both systems during the open phase. Dissolved inorganic nitrogen (DIN, sum of nitrate & ammonia) concentrations ranged from 0.5 to 204 μM in the Mdloti (Mundree, 2001; Nozais *et al.*, 2001) and from 1.6 to 17.7 μM in the Mpenjati (Perissinotto *et al.*, 2002). Concentrations of dissolved inorganic phosphorous (DIP, orthophosphate) varied between 0.3 and 5.8 μM in the Mdloti and between 0.2 and 1.2 μM in the Mpenjati. In these TOCEs, DIN:DIP molar ratios were usually well above the Redfield ratio of 16:1 during the open phase, but often below this critical value during the closed phase, thus indicating a potential limiting effect of nitrogen during the closed phase and phosphate during the open phase.

Prior to these studies, a snapshot survey of the nutrient status of the Mdloti and Tongati in November 1981 (open or closed?) had concluded that the lowest soluble reactive phosphate values of 20-30 $mg.l^{-1}$ recorded in the Mdloti may have been sufficiently low to limit primary production (Blaber et al., 1984). Most of the nitrogen in Tongati was in the form of ammonia possibly derived from treated sewage. The high nitrate values recorded in the Mdloti, with up to 340 $mg.l^{-1}$, were also attributed to the use of agricultural fertilizer on adjacent sugarcane fields. In the Groot Brak Estuary, mean nitrate values were 0.8 μM in August 1992 and increased to 1.7 μM in November, as a result of an increase in freshwater inflow during this period (Slinger *et al.* 1995). Further snapshot measurements of nutrient concentrations in TOCEs were made by Harrison *et al.* (2000) in virtually all South African systems during the period 1992-1999. NO_3 (last time called nitrate) values obtained during this survey ranged from <1 to 5547 $\mu g.l^{-1}$ while PO_4 ranged from <1 to 6392 $\mu g.l^{-1}$. Try to use consistent terms and units.

4) Phytoplankton & microphytobenthos

The material used in this review is listed in the table below.

System	Reference
Groot Brak (warm temperate)	Adams & Bate 1999
Nyara (warm temperate)	Perissinotto *et al.* 2000, Walker *et al.* 2001, Perissinotto *et al.* in press
Mpenjati (subtropical)	Kibirige 2002, Perissinotto *et al.* 2002, Perissinotto *et al.* in press
Mhlabatshane (subtropical)	Watt 1998
Fafa (subtropical)	Oliff 1976, Grindley 1981
Manzimtoti (subtropical)	Watt 1998
Mdloti (subtropical)	Blaber *et al.* 1984, Nozais *et al.* 2001, Mundree 2001, Perissinotto *et al.* in press
Tongati (subtropical)	Blaber *et al.* 1984

Table 3. Major microalgal studies carried out in South African TOCEs

Studies on water-column microalgae (phytoplankton) are available only from two warm temperate TOCEs in the Eastern Cape and three subtropical systems on the KZN coast. Data on TOCEs' benthic microalgae (microphytobenthos) are available for the same two warm temperate TOCEs and for six of the subtropical estuaries of KZN. Snapshot measurements of phytoplankton chlorophyll-a biomass were made by Harrison *et al.* (2000) in virtually all South African TOCEs during the period 1992-1999. Values obtained during this survey ranged from < 1 to 10.92 $\mu g.l^{-1}$. These data are, however, not discussed in this review in view of the uncertainties associated with the methods employed and the different periods during which they were collected in the various estuaries.

Composition and distribution

There are currently limited data on the taxonomic composition and distribution of phytoplankton in South African TOCEs. Flagellates and cyanobacteria (blue-green algae) have been reported as dominant components of the phytoplankton of the Groot Brak Estuary, with smaller numbers of diatoms, dinoflagellates and euglenoids also present (Adams & Bate, 1999). Diatom species were found to occur over a wide salinity range from freshwater to virtual marine conditions (salinity 35‰). In the Nyara study of 1997, diatoms were dominated by the genera *Navicula, Amphiprora, Gyrosigma* and *Nitzschia* Walker *et al.* 2001). Their concentrations ranged from 10^3 to 10^5 cells.l^{-1} and were consistenly lower than picoplankton and dinoflagellate numbers (Walker *et al.* 2001).

An extensive taxonomic study of benthic diatoms is currently being carried out at a number of estuaries all around the country, as part of a dedicated Water Research Commission (WRC) project. Among these are two TOCEs: the Mpekweni in the Eastern Cape and the Manzimtoti in KwaZulu-Natal (Bate *et al.*, 2002). This once-off sampling study is largely aimed at evaluating the usefulness of these microalgae in the assessment of ecosystem health, by showing associations between estuarine water quality and dominant microphytobenthic species. In the sediments of the Tongati and Mhlanga, Blaber *et al.*, (1984) reported the common presence of unidentified species of the genera *Navicula*, *Nitschia*, *Synedra*, *Spirulina*, *Ocillatoria*, as wells as euglenoid flagellates. A whole year, spatio-temporal study of the benthic diatom community of the Mdloti Estuary has shown a clear dominance of four genera and 26 species: 13 *Navicula* spp., 6 *Nitschia* spp., 3 *Amphora* spp. and 2 *Achnanthes* spp. (Mundree 2001). Species composition and dominance have been related to water quality in a comparative study of the Mhlabatshane and the Manzimtoti (Watt 1998). The naturally oligotrophic Mhlabatshane sediments were found to be dominated by the diatom genera *Acanthidium* and *Diploneis* , while the nutrient enriched and polluted Manzimtoti sediments contained mainly small species of *Navicula* and *Nitzschia*. A total of 44 diatom species were identified in the Manzimtoti and 21 in the Mhlabatshane (Watt 1998).

Distribution data have indicated that dominance of a diatom taxon is largely dependent on sediment type and the open/closed state of the estuary. In the Mdloti, benthic microalgae were found in substantial numbers at depths of up to 5 cm below the sediment surface, well below the photic zone (Mundree, 2001). Their distribution at such depths may represent an important stock of potential primary producers in TOCEs and is likely to be related to active migration, physical hydrodynamic conditions and/or bioturbation by deposit feeders. In terms of relationships with sediment type, vertically homogenous microphytobenthic chl-a concentrations were recorded in the upper 5 cm of sediment in the lower reaches of the Mdloti Estuary, where coarse sand dominated the sediment structure. Conversely, chl-a was concentrated in the upper 2 cm in the upper reaches of the estuary where fine silt made up the bulk of the sediment (Mundree 2001).

Biomass and primary productivity

Recent studies in South African TOCEs have shown that phytoplankton biomass ranges from 0.09-15.4 mg chla m^{-3} (Oliff 1976, Perissinotto *et al.* 2000, Nozais *et al.* 2001, Perissinotto *et al.* 2002). These values are lower than those measured in permanently open estuaries where maximum values often exceed 20 mg chl a m^{-3} and occasionally even 100 mg chl a m^{-3} (Adams & Bate, 1999). Conversely, microphytobenthic biomass has been found to be generally higher in TOCEs than in permanently open estuaries, with values ranging from 1.4-616 mg chl a m^{-2} (Adams & Bate 1994, 1999; Nozais *et al.* 2001). The maximum concentrations obtained are the highest values ever recorded for any South African estuary and among the highest reported in the literature (Cahoon et al. 1999, Adams & Bate 1999, Perissinotto et al. 2002).

Microphytobenthic biomass has been shown to contribute a significant fraction of the total primary biomass in TOCEs, equaling or even exceeding that of phytoplankton in the overlying water. Studies in the Mdloti, Mpenjati and Nyara estuaries have reported microalgal biomass in the sediment to be 1 to 3 orders of magnitude higher than in the water-column (Nozais et al., 2001; Perissinotto et al., 2003). Such results have been attributed to the prevailing conditions in these systems, which include low turbidity and current speed, a more stable sediment and salinity environment and a large nutrient pool available within the substratum (Adams & Bate, 1999). General spatial and temporal patterns of phytoplankton and microphytobenthic biomass have been found to be largely controlled by factors such as salinity, exposure (desiccation), water currents, sediment hydrodynamcs, granulometric composition (Adams & Bate, 1999; Mundree, 2001), and by interactions between light, nutrient availability and grazing pressure (Mundree, 2001; Perissinotto et al., 2000; Nozais et al., 2001; Perissinotto et al., 2003).

The first productivity study carried out in a TOCE was that of Oliff (1976) at the Fafa Estuary, on the KZN south coast. Only phytoplankton productivity was measured on that occasion and values obtained ranged from 0.32 to 7.50 $mgC.m^{-3}.h^{-1}$ (^{14}C method) and from 1.65 to 29.93 $mgC.m^{-3}.h^{-1}$ (oxygen method) (Oliff 1976, Grindley 1981). More recently, the comparative productivity of phytoplankton and microphytobenthos has been investigated in the Mdloti and Mpenjati estuaries (Perissinotto et al., 2003). Contrary to the biomass pattern, microalgal productivity in these two systems seems to be higher in the water-column than within the sediment, often by an order of magnitude. These results have shown a phytoplankton production range of 3-208 $mgC.m^{-2}.h^{-1}$ and a microphytobenthic production range of 0.23-16.4 $mgC.m^{-2}.h^{-1}$, with no clear evidence of substantial differences between open and closed phases. These preliminary findings need to be confirmed with further measurements. If the trend is indeed consistent, then it could mean that: 1) either the large microalgal biomass observed within the sediment is actually produced in the water-column and only later sinks to the bottom; or 2) the biomass produced in the water-column is constantly removed by grazers, while that within the sediment remains virtually unutilized by consumers. Both of these hypotheses are equally viable and have important but different implications for the trophic functioning of TOCEs. Studies on the rate of exchange of microalgal species and biomass between the water-column and the sediment of TOCEs are thus needed, as are measurements of grazing rates on both phytoplankton and microphytobenthic stocks.

Trophic role

Microalgae not only constitute an important carbon source for local benthic-pelagic food webs, but also provide an essential link between inorganic compounds and organic matter available to the higher trophic levels and top predators (Miller *et al.*, 1996; Mortazavi *et al.*, 2000). Studies at the Mpenjati and Nyara estuaries have indicated that the grazing impact of zooplankton on microalgal biomass in the water-column is very substantial (Perissinotto *et al.*, 2003). Current research is now underway to estimate the degree of microalgal utilization as an energy source by other components of the food web, such as meroplanktonic larvae and meiobenthic organisms. The ultimate fate of microalgae in the ecosystem and their degree of direct consumption by primary and higher level consumers is of fundamental importance to the energy balance of TOCEs.

5) Macrophytes

The following synthesis is based partially on the estuary specific references listed in the table below. Estuaries are listed from west to east.

Estuary	**Region**	**Reference**
Rietvlei/Diep	Cool temperate	Grindley & Dudley 1988
Eerste	Warm temperate	Grindley 1982
Buffels (Wes), Elsies, Sir Lowry's Pass, Steenbras, Buffels (Oos)	Warm temperate	Heinecken, Bickerton & Morant 1982
Rooiels	Warm temperate	Heinecken 1982
Onrus	Warm temperate	Heinecken & Damstra 1983
Hartenbos	Warm temperate	Bickerton 1982
Groot Brak	Warm temperate	Morant 1983
Keurbooms/Bitou, Piesang	Warm temperate	Duvenage & Morant 1984
Groot (Wes), Sout	Warm temperate	Morant & Bickerton 1983
Qinira	Warm temperate	Wiseman, Burns & Vernon 1993
KZN estuaries	Sub-tropical	Begg 1978, Hiralal 2001

Table 4. Major macrophytic studies carried out in South African TOCEs

Adams *et al.* (1999) succinctly point out that "estuarine macrophyte species composition, biomass and productivity, like all plant communities, is determined by a complex interaction between numerous abiotic and biotic factors: irradiance, temperature, nutrient availability, grazing and competition. By the nature of their habitat, estuarine macrophytes are also subjected to varying salinity and fluctuating water levels".

In selecting from the above environmental parameters in terms of gradients on different scales, it is clear that a condition such as salinity can vary within or between estuaries but a parameter such as average temperature will show a steady latitudinal change. South Africa is big enough for such latitudinal changes to occur and accordingly there is a change in estuarine vegetation from more tropical species to the north to more temperate types in the south. This change is most clearly demonstrated by the mangroves of which five species occur at Kosi Bay, but two of these *viz.* *Ceriops tagal* and *Lumnitzera racemosa*, do not occur any further south. The other three species fade out in the Eastern Cape. Mangroves are, however, not ideal examples because their presence in estuaries depends on a permanently open mouth and the resultant tidal action. In temperate areas the rice grass *Spartina maritima* is similarly absent from blind estuaries where there is no tidal exchange (Day 1981). Species whose presence or absence is less strongly linked to tidal environments but which are more common in more temperate estuaries include some of the sedges *Scirpus spp.* and the salt marsh inhabiting *Limonium spp.* and *Salicornia spp.* Others such as *Juncus kraussii, Phagmites australis, Ruppia maritima* and *Zostera capensis* may occur in suitable habitats along the entire South African coast (Day 1981).

Relevant environmental parameters

TOCEs exhibit several features of relevance to plants that separate them from permanently open systems. The absence of year-round tidal influence, which obviously causes submergence or exposure of intertidal areas twice a day, causes closed systems to maintain a relatively constant water level between disruptions by natural breaching following overtopping of the bar or by anthropogenic disturbance. Natural breaches are generally seasonal and linked to rainfall cycles although, on rare occasions, storm events at sea can cause waves that overtop the berm from the seaward side that ultimately can cause a breach. Their timing is nevertheless often erratic and unpredictable. Tidal cycles of water level may resume following breaching but high tide levels may not coincide with the high levels attained during closed phases. This variability will obviously affect the nature of the plant community at the land-water interface.

Extended periods of mouth closure associated with high water levels and inundation of salt marshes will result in a die-back of plants. Data are inadequate to cover all species but stress appears to occur after 2-3 weeks. Following breaching, buried seed banks will, however, contribute to rapid re-establishment (Adams *et al.* 1999).

Normally submerged macrophytes exposed by a fall in water level will eventually die back and while there are interspecific differences, survival is generally measured in terms of days. A resilient seed bank again contributes to the re-establishment of these plants (Adams *et al.*1999)

Water movements associated with tidal currents have both direct and indirect effects. According to Adams *et al.* (1999), quoting several examples, growth of submerged plants is significantly reduced by direct current effects at speeds of 0.5 $m.s^{-1}$. Currents also affect the amount of suspended matter in the water column and hence the turbidity and light penetration as well as the general stability and composition of the substratum. The absence of currents in closed systems, and hence the possibility of clearer water, greater light penetration and a more stable substratum should enhance the growth of submerged macrophytes, assuming the nutrient regime is suitable.

The salinity in closed systems will depend on the balance between freshwater inflow, bar overtopping during high tides, evaporation and any groundwater seepage. Salinity tolerance ranges shown by estuarine macrophytes commonly vary from near freshwater to 40-60‰ (90‰ for *Ruppia*) in species such as *Zostera capensis* and *Ruppia cirhosa* (Adams *et al.*1999) and are often therefore suited to the degree of fluctuation found in TOCEs. Under conditions of extended near fresh water, species such as *Potamogeton spp.* and *Nymphaea spp.* may become dominant.

Available information

Published, although possibly out-dated information on the macrophytes of individual temporary/open closed systems is available for the KwaZulu Natal estuaries. These data are the result of surveys carried out by Begg in the 70's (Begg 1978) and for a selection of estuaries from the Orange to the Great Kei, incorporated into the "Estuaries of the Cape" programme carried out by the National Research Institute of Oceanology of the Council for Scientific and Industrial Research in the early 80's (Heydorn & Tinley 1980). Estuaries within the old Transkei geographical area were neglected and this gap has still not been adequately filled. Use has therefore been made of the general review produced by Day (1981) and the more recent synthesis by Adams *et al.* (1999) in order to draw an overall picture.

The over-riding impression that appears from the Begg (1978) collation of data indicates that the KZN TOCEs are predominantly small, that the vegetation is dominated by emergent species, particularly *Phragmites australis* and that rooted submerged species are rare. Whether this represents the historical situation is difficult to say, as these systems have arguably been characterized in the long term by floods and rapid flows as a result of the relatively high rainfall and steep coastal gradient. Associated turbidity and coarse sediments would further restrict the growth of submerged rooted plants. The freshwater swamp tree *Barringtonia racemosa*, sometimes erroneously referred to as the freshwater mangrove, occurs in localised patches in sub-tropical regions where conditions have presumably been adequately stable for long enough for it to establish itself. It appears to be quite tolerant of extended periods of root immersion and seems to be more adversely affected by periods of exposure during open mouth conditions. Tidal environments which could support salt marshes are typically, although not universally, occupied by mangroves and in the generally narrow valleys which contain the smaller systems there is little area for salt marsh development. Water logged areas tend to be dominated by *P. australis*, or at slightly higher elevations by *Hibiscus tiliaceus*, or the land has been drained and is being farmed. From the Mlalazi northwards, the coastal gradient is flatter and estuaries are fewer, larger and typically more open. The only exception is the Nhlabane, about 25 km north of Richards Bay, which has been considerably impacted by the construction of a barrage but has supported *Potamogeton* sp. and *Nymphaea* sp. during periods of stable low salinity. There are no records of *Ruppia* spp. in any KZN TOCEs. On the other hand, the occurrence of *Zostera capensis* has been reported regularly from the Umgababa Estuary from the early 1950s till 1989 (Begg 1984, Hiralal 2001). However, it has not been recorded since the latter date in any KZN TOCEs (Hiralal 2001).

Estuaries that were within the old Transkei boundaries, now included in the eastern part of the Eastern Cape, often referred to as the Wild Coast, have not been well described. Steep gradients in the northern section of this coastline resemble those of KZN and would presumably generate a similar type of environment and a similar plant community. Many of these small systems do not seem to have been impacted to the same extent as KZN estuaries. This possibility needs to be investigated because it might give an indication of what the KZN situation may have been in its unspoilt state. This could potentially help in the process of rehabilitation.

The following comments are based on data from 12 estuaries between East London and the Rietvlei/Diep at Milnerton, on the northern outskirts of Cape Town (Table 4). The available information is possibly slightly out-dated but the CSIR "Green book" reports still represent the best available summaries. Emerging from these summaries is the conclusion that the TOCEs to the south and west of the Kei River show differences from those of KZN. This relates particularly to submerged macrophytes which are rare in KZN systems but were recorded in five of the 12 Cape estuaries. Species included *Ruppia maritima*, *Ruppia spiralis*, *Zostera capensis* and *Potamogeton pectinatus*. In the Diep River *Aponogeton* sp. and *Zannichellia aschersoniana* have also been found. These species do have different salinity preferences and their seasonal presence or distribution may vary within any one system. A feature common to both the KZN and the Cape systems is the nature of the emergent species of which the reed *Phragmites australis* was the most common; other wide spread species in this category include the rush *Juncus kraussii* and the sedge *Scirpus* spp. The latter is generally associated with lower salinity or freshwater seepage.

Salt marshes are features of temperate rather than tropical estuaries in South Africa and are best developed, most diverse and most clearly defined in areas where there are gentle topographic gradients on either side of the water and substantial tidal exchange. Along the KZN coast, they seem to be replaced by *Phragmites*, perhaps because the high rainfall rate observed here washes the salt out of the soil. This does not mean that they are excluded from sub-tropical and temperate TOCEs, but the species which do occur typically form a sub-set of the full suite of species occurring in permanently open systems. The most commonly mentioned species in the Cape surveys as well as in Day's summary (Day 1981) are *Juncus kraussii*, *Sarcocornia* spp., *Sporobolus* sp. and *Triglochin* spp. *Salicornia* spp. and *Limonium* spp. are more common in the cooler estuaries of the southern and western Cape. As mentioned previously, these species cannot tolerate extended submersion and it should not be expected that the periodicity of the rise and fall in water levels typical of TOCEs will allow a stable zonation of diverse fringing plant communities to develop in these areas.

6) Zooplankton

The material used in this review is listed in table 5.

System	Reference
Kabeljous (warm temperate)	Schlacher & Wooldridge 1995
Nyara (warm temperate)	Perissinotto et al. 2000, Walker et al. 2001, Perissinotto et al. in press
Mpenjati (subtropical)	Kibirige 2002, Kibirige et al. in press, Perissinotto et al. in press
Umgababa (subtropical)	Grindley 1981
Mhlanga (subtropical)	Whitfield 1980 a, b
Mdloti (subtropical)	Blaber et al. 1984
Tongati (subtropical)	Blaber et al. 1984
Siyaya (subtropical)	Connell et al. 1981, van der Elst et al. 1999

Table 5. Major zooplankton studies carried out in South African TOCEs

Meaningful data on TOCEs' mesozooplankton are only available for two estuaries situated in the warm temperate region of the Eastern Cape and for six subtropical estuaries on the Kwazulu-Natal coast. The only study on microzooplankton is from the Nyara Estuary in the Eastern Cape (Walker et al. 2001).

The prolonged period of mouth closure that TOCEs experience generally leads to poor levels of zooplankton taxonomic diversity. This is in contrast to the situation observed in their permanently open counterparts, where neritic marine species penetrating with the tides often increase the species richness to levels even in excess of a hundred species, particularly in the mouth region (Grindley 1981). The few studies conducted so far in South African TOCEs show that during the closed phase (coinciding with the winter/spring seasons on the southern and east coasts) the mesozooplankton community is composed of few taxa (10-20), with 2-5 species normally accounting for ≥ 80% of total numerical abundance (Perissinotto et al. 2000, Kibirige 2002). These include consistently the calanoid copepods *Pseudodiaptomu hessei* and *Acartia natalensis* as well as a few unidentified species of harpacticoids, cyclopoids such as *Halicyclops* spp. and *Thermocyclops emini*, the amphipod *Grandidierella lignorum* and isopods such as *Cirolana fluviatilis* (Whitfield 1980, Connell et al. 1981, Grindley 1981, Blaber et al. 1984, Schlacher & Wooldridge 1995, Perissinotto et al. 2000, Kibirige 2002). Their larval stages, and particularly copepod nauplii, are always the most numerically abundant group during this phase. Where conditions at the mouth are favourable to the maintenance of salinity levels above 10‰, the mysids *Gastrosaccus brevifissura* (e.g. Mpenjati, Kibirige 2002) and *Rhopalophthalmus terranatalis* (e.g. Kabeljous, Schlacher & Wooldridge 1995) are also among the dominant components of the mesozooplankton community during the closed phase. Elsewhere, where limnetic or oligohaline conditions prevail throughout the estuary, such as in the perched TOCEs on the KZN north coast, mesozooplankton may include large proportions of *Prionospio* spp. polychaetes (Blaber et al. 1984), the cladocerans *Moina micrura* & *Ceriodaphnia reticulata* (Connell et al. 1981) as well as ostracods and the bivalve *Musculus virgiliae* (Blaber et al. 1984).

A study on the diel vertical migration of the mesozooplankton during the closed phase was undertaken in the Kabeljous Estuary in February 1992 (Schlacher & Wooldridge 1995). This has shown that in the absence of any horizontal-displacing movement (tidal or freshwater flows) the patterns of vertical ascent/descent during the nighttime is substantially simpler than under open mouth conditions, for all species. Predictably though, there are still some marked differences between species. For instance, while the copepod *P. hessei* and the mysid *R. terranatalis* exhibit only one bout of upward migration after sunset, the harpacticoids have a more bimodal behaviour, with one peak after sunset and a second one around midnight. Other species show a short peak after sunset, followed by either a progressive disappearance from the water-column during the rest of the night (i.e. *G. lignorum*) or by marked fluctuations throughout the night (i.e. *C. fluviatilis*) (Schlacher & Wooldridge 1995).

A totally different situation is observed during the open phase, normally in the spring/summer seasons. The initial period of river/freshwater dominance is typified by the sudden, temporary removal of the dominant components of the closed phase and the appearance of freshwater species, all the way down to the mouth. These may include chironomid and other insect larvae (Blaber et al. 1984), several species of cladocerans, ostracods and rotifers (Connell et al. 1981, van der Elst et al. 1999). Conversely, the setting in of a proper estuarine open phase, with substantial penetration of seawater during the tidal flood, results in a large-scale appearance of numerous taxa of neritic origin. These include many species of copepods (mainly calanoids), gastropod & bivalve veligers, barnacle cypris, chaetognaths and several other taxa. Larval forms, including those of fish, penaeid shrimps and macrurans (Whitfield 1980, Kibirige 2002) are prominent and contribute the bulk of the recruitment process into these estuaries.

The microzooplankton survey carried out in the Nyara Estuary in 1997 (Walker et al. 2001) showed relatively large concentrations of ciliates, tintinnids and heterotrophic/mixotrophic dinoflagellates during the closed estuarine phase. Numerical abundances of ciliates/tintinnids and dinoflagellates ranged between 10^4 and 10^5 cells.l^{-1}, respectively. In terms of carbon content, the most important group was the dinoflagellates, particularly unidentified species of *Protoperidinium* (Walker et al. 2001). This appears to be a condition typical of an ecosystem strongly affected by the microbial loop and driven to a large extent by *in-situ* regenerated, rather than new, nutrients. Such a situation will, of course, be reversed during the open phase, when allochthonous nutrients enter the system once again via fresh water runoff and tidal influence.

The high zooplankton diversity observed in permanently open estuaries, compared to TOCEs, is in sharp contrast to the relatively low biomass that individual taxa normally exhibit within their zooplankton communities. On the other hand, the low average diversity of TOCEs is more conducive to the biomass build-up of a few dominant species (Grindley 1981). Under this point of view, it may be expected that TOCEs achieve higher gross zooplankton productivity rates than permanently open estuaries, given their short period of communication with the open ocean during an average annual cycle. This would minimize competition with neritic species and also reduce the losses due to exchange with the ocean. This would enhance their role as nursery areas for marine and estuarine fish and crustacean species, compared to the permanently open systems. However, the recruitment capacity for such species is probably hampered by the short-term and small-scale nature of their communication with the open ocean. This in turn would reduce the impact of the consumers on the zooplankton stock of these estuaries. Both factors above may in the end explain the very high zooplankton biomass levels that have recently been observed in two TOCEs, the Nyara in the Eastern Cape (Perissinotto et al. 2000, Walker et al. 2001) and the Mpenjati on the KZN south coast (Kibirige, 2002, Kibirige et al. in press). In the Nyara, night-time abundance estimates ranged from

$3.8\cdot10^4$ to $6.5\cdot10^6$ ind·m^{-3} and biomass from 0.002 to 2.03 g DW·m^{-3}. The Mpenjati exhibited night-time abundance levels of $1.8\cdot10^4$-$2.2\cdot10^5$ ind·m^{-3} and a biomass of 0.393 – 2.18 g DW·m^{-3}. In all cases, values were much higher during the closed phase than during the open phase of the estuary. The above biomass levels show a large variation, with values ranging from the highest ever recorded in South Africa to almost the lowest. This suggests that there are aperiodic bursts in growth (during the closed phase) followed by periods of depression (during the open phase) (Whitfield 1980, Perissinotto et al. 2000). The important point here is that, on average, the biomass values obtained in these TOCEs were 2-5 times higher than the average values reported for the most and the least productive South African permanently open estuaries (Wooldridge 1999, Perissinotto et al. 2000). Combined with the relatively poor phytoplankton stocks observed in the water-column of TOCEs (see the earlier section on microalgae), this is a situation that may result in prolonged periods of imbalance between autotrophic food availability and herbivore food demands in the pelagic subsystem.

Mesozooplankton grazing studies were carried out at the Mpenjati between 1998 and 2001 (Kibirige 2002, Kibirige & Perissinotto 2003). During the winter closed phase of 1999, autotrophic food consumption by the three dominant species of zooplankton at the Mpenjati ranged between 34 and 70% of water-column biomass, thus suggesting that at times these rates may exceed the total daily phytoplankton production of the estuary (Perissinotto *et al.* 2003). These rates are very high, compared with those reported from similar studies undertaken in South African permanently open estuaries, where grazing rates are normally within the range of 4-40% of available phytoplankton biomass (Froneman 2000, Grange 1992). Thus, in TOCEs phytoplankton alone may not be able to sustain the entire energetic demands of the consumers throughout the year. Measurements of $\delta^{13}C$ and $\delta^{15}N$ ratios in the three dominant zooplankton species of the Mpenjati and in their possible food sources (particulate organic matter, detritus and microphytobenthos) show that each grazer derives most of its energetic requirements from a specific and unique food source within the same trophic level (Kibirige *et al.* 2002). This strategy may minimize inter-specific competition and hence improve the utilization of the food sources available in these estuaries.

7) Meiofauna

The first and only specific review of South African estuarine meiofauna was produced over 20 years ago by Dye & Furstenburg (1981), as part of the pioneering synthesis of South African estuarine knowledge edited by Day (1981). Much of the following summary is drawn from this review with an emphasis on those areas that are relevant to TOCEs.

The term "meiobenthos" was coined by Mare (1942) in order to separate the macrobenthos from the microbenthos and conventionally refers to organisms passing through a 1.0 or 0.5 mm sieve but retained by a 0.045 mm mesh. Separation on the above basis would result in the inclusion of larvae of some macrobenthic species, which would eventually outgrow the meiofauna size range and are therefore described as temporary forms. The permanent meiofauna is dominated to a very large degree by the Nematoda followed by harpacticoid, mystacocarid and ostracod Crustacea, halacarid mites, Kinorhyncha, Gastrotricha, Archiannellida, Tardigrada, Turbellaria and Rotifera. There are also some specialized forms of typically larger sized organisms such as Hydrozoa, Nemertina, Bryozoa, Gastropoda, solenogasters, Holothuroidea, Tunicata, Priapulida and Sipunculoidea.

The review of South African data by Dye & Furstenburg (1981) was based very largely on their own work in the Swartkops Estuary but they also refer to the Kromme and Berg. All of these are permanently open systems. In a much later review (de Villiers, Hodgson & Forbes 1999), there is further reference to work on the Mngazana (Dye 1983a,b), another open system, and also to the Bot River (de Decker & Bally 1985) which, while not considered a TOCE (Whitfield 2000), provides the closest available approximation to such a system in the south-western Cape. Dye & Furstenburg (1981) suggest a very broad average meiofaunal density in open estuaries of *ca.* 1 000 individuals per 10 cm^2. The Bot River density data are unfortunately quoted as numbers of individuals per cm^3, which makes comparison difficult. De Decker & Bally (1985), however, quote Dye & Furstenburg (1981) as giving an average density of approximately 100 animals per cm^3 in South African estuaries and compare this with their own very much lower figures of 0.008 – 1.5 ind.cm^{-3}, which clearly indicated an impoverished fauna. Diversity was also low at seven recorded taxa, primarily platyhelminthes, nematodes and oligochaetes, versus 52 genera of nematodes in the Swartkops (Dye & Furstenburg 1981). The low abundance and diversity were attributed to the sediment characteristics of the system.

The only available data from a TOCE were derived from the Mdloti Estuary, a small system about 27 km north of Durban that was monitored from June 1999 to August 2000 (Haines 2000). Meiofaunal abundance was higher during closed mouth periods, peaking in July 1999 at 1785 ind.10 cm^{-2}. Densities decreased to less than 500 ind.10 cm^{-2} in all months during open mouth conditions, from November 1999 to April 2000. On average, nematodes contributed 61% numerically of the total meiofauna during the year. The unexpected appearance of the astigmatid mite *Tyrophagus putrescientiae* in the meiobenthic samples is discussed by Marshall *et al.* (2001). This species occurred in most months and constituted 98.6% of the samples in November 1999, a period when the mouth was open (Haines 2000). Density of the Mdloti Estuary meiobenthos during closed mouth phases appears to be comparable with figures obtained from open systems such as the Swartkops (Dye & Furstenburg 1981), but are much lower during open mouth conditions. The significance of the presence of the mite *T. putrescientiae* (Marshall *et al.* 2001) and its role in benthic dynamics remains to be clarified.

8) Macrobenthos

The estuaries referred to in this review of the macrobenthic fauna and the relevant references are summarized in table 6.

System	Reference
Nhlabane * (subtropical)	Forbes & Demetriades (2000, 2002); Vivier, Cyrus, Jerling & Cilliers (1998)
Tongati (subtropical)	Blaber, Hay, Cyrus & Martin (1984)
Mdloti (subtropical)	Blaber, Hay, Cyrus & Martin (1984)
Mhlanga (subtropical)	Whitfield (1980)
Gqutywa (warm temperate)	Teske & Wooldridge (2001)
Mtati (warm temperate)	Teske & Wooldridge (2001)
Mpekweni (warm temperate)	Teske & Wooldridge (2001)
Old Womans (warm temperate)	Teske & Wooldridge (2001)
East Kleinemonde (warm temperate)	Teske & Wooldridge (2001)
West Kleinemonde (warm temperate)	Hill (1965)
van Stadens (warm temperate)	Teske & Wooldridge (2001)
Kabeljous (warm temperate)	Teske & Wooldridge (2001)
Bot/Kleinmond * (warm temperate)	De Decker (1987)

Table 6. Major macrobenthic studies carried out in South African TOCEs. (* = Not a proper TOCE, but included due to its relevance to closed phase dynamics).

Over 50 years ago, Day (1950) recognized that the South African estuarine fauna was dominated by species of marine origin, which favoured or required sheltered conditions. He recognized five components of which four, *viz.* stenohaline marine, euryhaline marine, estuarine and migratory were marine in origin and in combination occupied the bulk of any estuarine system. The freshwater component was restricted to low salinities at the heads of the systems. A much greater degree of euryhalinity was therefore apparent in the estuarine fauna than in the freshwater fauna.

The macrobenthic fauna of South African estuaries in general is dominated by errant and sedentary polychaetes; cirripedes, amphipods and isopods amongst the lower Crustacea; penaeid, caridean and anomuran prawns and brachyurans amongst the higher Crustacea and finally bivalve and gastropod molluscs. While other groups occur, they are far less common. Sub-tropical species such as the penaeid prawns and some of the crabs tend to become more common further north but many other species are remarkably widespread (Day 1981). The number of species recorded in any one estuary varies from single figures to over 300 in the case of Knysna (de Villiers *et al.* 1999). High numbers are typically associated with large, open tidal systems with high habitat diversity. According to de Villiers *et al.* (1999), permanently or mainly open estuaries tend to have richer macrobenthic communities, usually >60 species, while systems which are closed for long periods usually have fewer than 35 species. Day (1964) attributed such low figures to a limited recruitment of individuals from the sea. More recently Teske & Wooldridge (2001), based on studies in the Eastern Cape, have argued that diversity in permanently open systems is adversely affected by freshwater dominance. They also showed that faunal diversity in TOCEs, while conforming to the pattern above, may be intermediate between systems such as Knysna and freshwater-dominated systems.

Although there is no simple relationship between species richness and the frequency or duration of open mouth periods, the macrobenthic fauna of closed systems represents a sub-set of the much larger number of species that has been recorded in large, permanently open systems. Species that disappear from closed estuaries are not replaced by other more specialized species.

The most obvious absence in closed systems is of species such as the fiddler crabs *Uca* spp., the soldier crab *Dotilla fenestrata*, *Macrophthalmus* spp., and the marsh and mangrove crabs of the genus *Sesarma*. All of these species have a strong dependence on intertidal areas for feeding and the loss of this habitat through the absence of tidal action will be a strong excluding factor. The occurrence of migratory species, which have an obligatory marine phase in their life cycles, such as the penaeid prawns, the anomuran mudprawn *Upogebia africana* and the mangrove crab *Scylla serrata* will be dependent on the coincidence of migration periods with an open mouth condition. If this condition occurs, substantial recruitment may follow as in the case of *S. serrata* in the West Kleinemonde Estuary in the Eastern Cape, which was described by Hill (1975). The life cycle of *U. africana* has only recently been elucidated by Wooldridge & Loubser (1996), in terms of the migration of the larval phases from the spawning estuarine population to the inshore marine environment and back again. This aspect of the life cycle explains its absence from any system that closes for any significant length of time. Conversely, the sand prawn *Callianassa kraussi* which, like *U. africana*, is a burrower in estuarine sediments, does not have a planktonic larval phase and has no marine phase in its life cycle. It is therefore capable of survival in both open estuarine systems, where it may co-occur with *U. africana*, although there are sediment preferences that tend to separate the populations, and also in closed systems where it can be a major component of the benthos.

On the basis of the above arguments it is clear that the number of species surviving in temporarily open/closed estuaries is progressively reduced by the loss of tidal action and the link with the marine environment. Assuming that the surviving species are independent of the above factors the next parameter of significance is arguably salinity. As previously stated, hypersalinity, *i.e.* salinity greater than seawater, is unusual in closed estuaries although it is becoming more of a problem because of water abstraction from the feeder rivers. Hyper-salinity in systems such as St Lucia is at times a highly significant factor but as this condition has historically been associated with an open mouth condition it is of no relevance here, apart from noting that known estuarine invertebrates in high salinity conditions remain isosmotic and that the upper limits of tolerance are at salinities of about 50-60 (Hill 1981). Low salinities are more typical of closed systems. The broad salinity tolerance range of 5-55 for "most South African estuarine (macrobenthic) species" (de Villiers *et al.* 1999) is therefore pertinent. While five might be the lower tolerance threshold for a majority of species, empirical evidence of tolerance levels for all stages in the life cycles of resident estuarine species is sparse. It is noteworthy therefore that the benthic fauna of Lake Sibaya, formerly an estuary but now an isolated freshwater lake, includes the typical estuarine polychaete *Ceratonereis keiskama* and several crustaceans, such as the tanaid *Apseudes digitalis*, the amphipods *Corophium triaenonyx*, *Grandidierella lignorum* and *Orchestia ancheidos* and the crab *Hymenosoma orbiculare* (Allanson, Hill, Boltt & Schulz 1966). It is clear from the above that low salinities in closed estuaries are a contributing factor to a reduction in species richness although a few species can survive extended or even indefinite periods of fresh water conditions.

Whitfield (1980) recorded nine taxa in the Mhlanga Estuary in 1978. Of these, the polychaetes *Ceratonereis erythraensis* and *Dendronereis arborifera* and the amphipod *Corophium triaenonyx* were the most important in terms of contributions to the energy content per m^2. Blaber *et al.* (1984) recorded nine and 16 taxa respectively in the Tongati and Mdloti estuaries during 1980/81. Oligochaetes were common in both systems. The combination of the three polychaetes *C. erythraensis*, *D. arborifera* and *Desdemona ornata* was significant in the Mdloti, while a fourth polychaete *Prionospio* sp. dominated the benthos in the Tongati. In all cases the only significant group of freshwater origin was the Chironomidae.

TOCEs are a major feature of the South African coastline and to some extent compensate for their generally small size by their comparative abundance, in relation to permanently open systems. By comparison with open systems, they have their own characteristics and many of the biotic features can be explained on the basis of known aspects of the biology of the actual or potential residents. The nature and functioning of these systems is significantly influenced by the mouth condition and it has been indicated that abnormally frequent breaching is likely to adversely affect these processes. At the other end of the scale is the question of the effect of abnormally extended periods of closure. Long term data sets which could possibly resolve this question are extremely limited and possibly the best worked system in this regard is the Nhlabane Estuary, about 25 km north of Richards Bay. This system has been monitored for about 10 years through a period of extended closure to a regime of abnormally frequent breaching. Unpublished data indicate that the system supported up to about 40 species of macrobenthic invertebrates but there was typically a high dominance by the polychaetes *C. keiskama*, *D. arborifera*, *D. ornata* and *Prionospio* sp., the amphipods *C. triaenonyx* and *Grandidierella* spp. and the tanaid *A. digitalis*. These include the same species that have managed to survive in the freshwater Lake Sibaya. Total individual densities in the Nhlabane reached 20 to 25 000 per m^2, declining to as low as 2 000 after flood events or after prolonged closure and declining salinities (Vivier *et al.* 1998). Similar trends were recorded by Forbes & Demetriades (2000, 2002), who found that a decline in overall density was accompanied by a decline in the numbers of taxa recorded, *i.e.* uncommon species became progressively more rare and less frequently recorded, but did not necessarily disappear.

All the above information was collected using a Zabalocki-type Ekman grab, which samples the upper 10-15 cm of sediment depending on the texture. It was noted in the Nhlabane (Forbes & Demetriades, unpubl. data) that substantial numbers of the burrowing bivalve *Hiatula lunulata* were also present, but the bulk of the population was beyond the reach of the grab. The distribution of this species is poorly known and it should be appreciated that our general knowledge of the benthic faunas of TOCEs is equally poor and, while we have some knowledge of the species involved, other details of their biology, interactions, dispersal mechanisms and population dynamics are generally not available.

All the above data relate to KZN estuaries and it is therefore worthwhile to compare these places with other areas along the South African coast. The Bot River on the south coast, although classified as an estuarine lake system and not a TOCE has an "excellent" information base (Whitfield 2000). Here, natural breaching is rare (de Decker 1987) but frequent artificial breaching allows comparison of the system and its responses with regularly breaching TOCEs in other parts of the country. Sampling before and shortly after breaching, as well as after a three month recovery period, indicated a fall in species numbers from 23 to 15 followed by an addition of only one further species after the recovery period. These numbers are similar to those recorded in the KZN and Eastern Cape systems while the same or closely related species, *viz.* the tanaid *A. digitalis*, the polychaetes *Capitella capitata* and *Prionospio pernana* and the amphipod *Melita zeylanica* were the most abundant. Total individual densities ranged between *ca.* 2 000 to 8 000 per m^2, which is again comparable with results from KZN and the Eastern Cape. The biomass was dominated by the bivalve *Arcuatula capensis* and, although this species is not a burrower, there is the interesting parallel with the Nhlabane system where another bivalve *H. lunulata* was a highly noticeable, although unquantified, component of the benthos. De Decker (1987) has suggested that in the Bot River estuary, under extended periods of closure, the fauna would tend towards that of a coastal lake but breaching interrupts this by eliminating 'lagoonal" species, while low salinities during closed periods reduce the numbers of marine colonizers, thereby maintaining a very low species diversity.

Teske & Wooldridge (2001) investigated 13 Eastern Cape estuaries of which 7 were TOCEs. They recorded 72 taxa of which 38 occurred in TOCEs. This latter number is in keeping with the figures for open/closed systems mentioned above. For their analyses they selected 14 of the most abundant species of which the commonest were the polychaete *C. capitata*, the cumacean *Iphinoe truncata*, the amphipod *C. triaenonyx*, the tanaid *A. digitalis* and chironomid fly larvae. These are broadly the same species that have been recorded in KZN and in the Bot River. The total individual density per m^2 in the different estuaries in all three geographical areas were broadly similar, ranging from *ca.* 2 000 to 8 000 in the Bot River (de Decker 1987), 2 000 to 12 000 in the Eastern Cape (Teske & Wooldridge 2001) and 2 000 to 20 000 in the Nhlabane (Vivier *et al.* 1998; Forbes & Demetriades 2000, 2002). It is of interest that despite the similarity of the commonest species in the permanently open and the TOCE systems investigated in the Eastern Cape, total individual benthic densities in the latter type of estuary were often two to four times higher than in permanently open systems. This possibly confusing situation indicates that an understanding of benthic dynamics in TOCEs as well as open systems, is going to depend on a greater appreciation of biological interactions. The benthic grabs used in these studies do not effectively sample deep burrowing species such as the thalassinid prawns *C. kraussi* and *U. africana*. The same criticism has already been made above in relation to burrowing bivalves. Although *U. africana* does not survive in closed systems, this and the other two taxa are all highly significant contributors to estuarine benthos and benthic biomass and can be major modifiers of the benthic environment. Their presence or absence will certainly have considerable effects on the functioning of these systems. Unfortunately there was no indication of the status of these species in the Eastern Cape estuaries surveyed. Quantification of these effects or interactions will depend on the results of longer term monitoring and more intensively focussed investigations than have been done to date.

9) Fish

The material used in this revision is listed in table 7.

System	Reference
Sand (cool-temperate)	Morant (1991), Quick & Harding (1994), Clark *et al.* (1994), Harrison (1998)
Eerste (cool-temperate)	Morant (1991), Clark *et al.* (1994), Harrison (1998)
Lourens (cool-temperate)	Morant (1991), Harrison (1998)
Rooiels (cool-temperate)	Morant (1991), Harrison (1998)
Buffels (Oos) (cool-temperate)	Morant (1991), Harrison (1998)
Seekoei (warm-temperate)	Dundas (1994)
Kabeljous (warm-temperate)	Dundas (1994)
Van Stadens (warm-temperate)	Dundas (1994)
East Kleinemonde (warm temperate)	Cowley & Whitfield (2001), Cowley *et al.* (2001), Vorwerk *et al.* (2001)
Klein Palmiet (warm-temperate)	Vorwerk *et al.* (2001)
Mpekweni (warm-temperate)	Vorwerk *et al.* (2001)
Mtati (warm-temperate)	Vorwerk *et al.* (2001)
Mgwalana (warm-temperate)	Vorwerk *et al.* (2001)
Bira (warm-temperate)	Vorwerk *et al.* (2001)
Gqutywa (warm-temperate)	Vorwerk *et al.* (2001)
Ngculura (warm-temperate)	Vorwerk *et al.* (2001)
Zolwane (subtropical)	Begg (1984a; 1984b)
Sandlundlu(subtropical)	Begg (1984a; 1984b)
Ku Boboyi (subtropical)	Begg (1984a; 1984b)
Kandandlovu (subtropical)	Begg (1984a; 1984b)
Umhlangankulu (subtropical)	Begg (1984a; 1984b)
Mvutshini (subtropical)	Begg (1984a; 1984b)
Bilanhlolo (subtropical)	Begg (1984a; 1984b)
Uvuzana (subtropical)	Begg (1984a; 1984b)
Kongweni (subtropical)	Begg (1984a; 1984b)
Vungu (subtropical)	Begg (1984a; 1984b)
Mhlangeni (subtropical)	Begg (1984a; 1984b)
Zotsha (subtropical)	Begg (1984a; 1984b), Harrison & Cooper (1991), Harrison & Whitfield (1995)
Boboyi (subtropical)	Begg (1984a; 1984b)
Mbango (subtropical)	Begg (1984a; 1984b)
Mtentweni (subtropical)	Begg (1984a; 1984b), Harrison & Connell (2002)
Mhlangamkulu (subtropical)	Begg (1984a; 1984b)
Damba (subtropical)	Begg (1984a; 1984b), Harrison & Whitfield (1995)
Koshwana (subtropical)	Begg (1984a; 1984b)
iNtshambili (subtropical)	Begg (1984a; 1984b)
Mzumbe (subtropical)	Begg (1984a; 1984b)
Mhlabatshane (subtropical)	Begg (1984a; 1984b)
Mhlungwa (subtropical)	Begg (1984a; 1984b)
Mfazazana (subtropical)	Begg (1984a; 1984b)
Kwa-Makosi (subtropical)	Begg (1984a; 1984b)

Mnamfu (subtropical)	Begg (1984a; 1984b)
Mvuzi (subtropical)	Begg (1984a; 1984b)
Fafa (subtropical)	Begg (1984a; 1984b), Harrison & Connell (2002)
Mdesingane (subtropical)	Begg (1984a; 1984b)
Sezela (subtropical)	Begg (1984a; 1984b), Ramm *et al.* (1987), Harrison & Connell (2002)
Mkumbane (subtropical)	Begg (1984a; 1984b)
Mzinto (subtropical)	Begg (1984a; 1984b)
Mzimayi (subtropical)	Begg (1984a; 1984b)
Mpambanyoni (subtropical)	Begg (1984a; 1984b)
Mahlongwa (subtropical)	Begg (1984a; 1984b)
Mahlongwana (subtropical)	Begg (1984a; 1984b)
Ngane (subtropical)	Begg (1984a; 1984b)
uMgababa (subtropical)	Begg (1984a; 1984b)
Msimbazi (subtropical)	Begg (1984a; 1984b)
Lovu (subtropical)	Begg (1984a; 1984b)
Little Manzimtoti (subtropical)	Begg (1984a; 1984b)
Manzimtoti (subtropical)	Begg (1984a; 1984b)
Mbokodweni (subtropical)	Begg (1984a; 1984b)
Mhlanga (subtropical)	Begg (1984a; 1984b), Whitfield (1980a; 1980b; 1980c), Harrison & Whitfield (1995)
Mdloti (subtropical)	Begg (1984a; 1984b), Blaber *et al.* (1984)
Tongati (subtropical)	Begg (1984a; 1984b), Blaber *et al.* (1984)
Mhlali (subtropical)	Begg (1984a; 1984b)
Seteni (subtropical)	Begg (1984a; 1984b)
Mdlotane (subtropical)	Begg (1984a; 1984b)
Nonoti (subtropical)	Begg (1984a; 1984b)
Zinkwasi (subtropical)	Begg (1984a; 1984b)
Siyaya (subtropical)	Van der Elst *et al.* (1999)

Table 7. Fish studies carried out in South African TOCEs

This table shows that published scientific information exists that deals with fish studies on at least 80 South African TOCEs. These studies have covered a variety of topics, including basic ichthyofaunal surveys (e.g. Millard & Scott, 1954; Harrison 1997a; 1997b; 1998; 1999a; 1999b) and comparative studies including seasonal variations in fish community structure (e.g. Begg 1984a; Bennett, 1989; Dundas, 1994; Harrison & Whitfield, 1995; Cowley & Whitfield, 2001; Vorwerk *et al.*, 2001). Other studies include trophic structure (e.g. Whitfield 1980b; 1980c) and fish recruitment mechanisms (e.g. Whitfield, 1980a; Harrison & Cooper, 1991; Cowley *et al.*, 2001). Several studies also looked at the degradation and rehabilitation of estuaries (e.g. Blaber *et al.*, 1984; Ramm *et al.*, 1987; van der Elst, 1999). The following is a summary of the most relevant findings from all these studies.

From the results of basic fish surveys of west coast estuaries, Harrison (1997a) concluded that the majority of the smaller outlets along the west coast (Atlantic Ocean) are of little value as habitat for fishes. Due to the arid conditions of the area, many of these systems comprise dry riverbeds and only carry water at times of exceptional rainfall (Heydorn, 1991). On the southwest Cape coast, Millard & Scott (1954) described the physico-chemical characteristics of the Diep Estuary; fish communities were also included in that study. Heavy rainfall during winter and spring results in the Diep Estuary opening to the sea, at this time salinity is reduced and turbidity increases; in summer, the estuary closes and the system may become hypersaline (Millard & Scott, 1954). In spite of these highly variable environmental conditions, the fishes in the Diep are dominated by estuarine and estuarine-dependent marine species indicating a viable nursery function (Millard & Scott, 1954; Harrison, 1997b).

From a review of the status of the estuaries of False Bay, Morant (1991) found that most of the systems have been modified through human activity. Harrison (1998) concluded that five TOCEs in the False Bay area, the Sand, Eerste, Lourens, Rooiels and Buffels (Oos) still appear to provide a viable nursery function and estuarine and estuarine-dependent marine fishes were a dominant component of their fish fauna. Quick & Harding (1994) investigated the management of the Sand Estuary for recreation and as a fish nursery. The Sand Estuary was regarded as an important nursery area for several marine species and a number of management measures were proposed to maintain acceptable water quality and enhance the recreation and fish nursery role of the system (Quick & Harding, 1994). Clark *et al.* (1994) compared the ichthyofauna of the Sand and Eerste estuaries and that of the adjacent surf-zones. The Sand and Eerste estuaries were dominated by a few species; the abundance of marine migrant species was found to be higher in the Eerste Estuary and this was attributed to differences in the duration of connection with the sea. Although estuarine and adjacent surf-zone ichthyofauna assemblages were similar, fish densities in the Sand and Eerste estuaries were considerably higher than those recorded in the adjacent surf-zones and this was ascribed to the higher spring and summer water temperatures together with protection from piscivores (Clark *et al.*, 1994).

Bennett (1989) compared the fish communities of a permanently open, a seasonally open (TOCE) and a normally closed estuary in the southwestern Cape. He reported marked seasonal changes in the fish fauna of the Kleinmond TOCE. In winter, freshwater input breaches the estuary, river currents are strong and temperatures are low; few fish species and individuals are present during this period. With the onset of the summer dry season, freshwater discharge declines, temperature rises and species enter the estuaries in increasing numbers. When the system closes, these fishes remain in the estuary until it opens again the following winter (Bennett, 1989). Harrison (1999a) found that the Kleinmond, Onrus and Ratel TOCEs on the southwest Cape coast were dominated by both estuarine and estuarine-dependent marine species, indicating a viable nursery function. Estuarine and estuarine-dependent marine fishes also dominated the ichthyofauna of the Klipdrifsfontein and the Blinde estuaries on the southern Cape coast (Harrison, 1999b).

Dundas (1994) compared the ecology of the fishes in three TOCEs in the Eastern Cape, the Seekoei, Kabeljous and Van Stadens. Generally, the physico-chemical characteristics and the fish species composition of the estuaries were similar. The Kabeljous Estuary was found to have a higher abundance and diversity relative to the other two systems; it was suggested that a possible explanation for this was the low level of human disturbance and lower water transparency (Dundas, 1994). Increased freshwater discharge often results in these estuaries breaching, however, mouth opening events were also found to be caused by high waves lowering the sand bar at the mouth to a level that enabled an outflow channel to form. Although changes in physico-chemical conditions were noticed after an opening event, these were generally small; the abundance and diversity of fishes, however, generally decreased following an opening event; this was ascribed to the emigration of marine migrant species and possibly poor juvenile recruitment into these systems (Dundas, 1994).

Cowley & Whitfield (2001) described the ichthyofauna characteristics of the East Kleinemonde TOCE in the Eastern Cape. Estuarine resident and estuarine-dependent marine fishes were well represented in the system, with juvenile estuarine-dependent marine fishes dominating the fish community. The East Kleinemonde was found to have a lower species diversity relative to similar subtropical KwaZulu-Natal TOCEs and nearby open estuaries. The reasons given for this were the biogeographical position of the system (warm-temperate) as well as the estuary being closed for much of the year. The numerical dominance of certain estuarine-dependent marine species, such as *Rhabdosargus holubi* and several Mugilidae species was attributed to their extended spawning season and the ability of juveniles to recruit into the estuary not only when the mouth opens, but also during marine overwash events (Cowley & Whitfield, 2001). Cowley *et al.* (2001) noted that the larval fish assemblage in the surf-zone adjacent to the East Kleinemonde Estuary were mainly estuarine-associated marine fishes and that species such as *R. holubi* uses overwash events as a mechanism to recruit into estuarine nursery areas.

Vorwerk *et al.* (2001) compared the ichthyofauna of ten estuaries on the former Ciskei coast of the Eastern Cape. Eight of these estuaries were TOCEs and dominated by resident estuarine and estuarine-dependent marine fishes. They noted little seasonal change in the ichthyofaunal structure and attributed this to the predominantly closed nature of these systems preventing large immigrations or emigrations of species.

Begg (1984a; 1984b) conducted a comparative study of 62 KwaZulu-Natal estuaries. He concluded that open estuaries were important nursery areas for estuarine-dependent marine species, while closed estuaries contained largely resident and freshwater species. From a study of three TOCEs in KwaZulu-Natal, Harrison & Whitfield (1995), found that estuarine-dependent marine species were an important component of the ichthyofauna of these systems. They also noted seasonal changes, with different fish assemblages dominating the ichthyofauna at different periods. This was linked to the spawning and migration patterns of the various species as well as the hydrological regime of each estuary. During winter, these estuaries are normally closed and water levels and food and habitat availability are high; at this time, estuarine-dependent marine fishes dominate the fish community. With the onset of the summer rains, these systems breach and adult and sub-adult estuarine-dependent marine species migrate to sea, while juveniles begin recruiting into these systems. Spring and summer are also peak breeding periods for resident estuarine and freshwater species, resulting in an increase in their contribution to the overall ichthyofauna. The reduction in water levels (and water volume) following breaching also results in the concentration of fishes in the lower reaches, further contributing to the increase or dominance in estuarine and freshwater species. The rise in water level, following mouth closure in autumn allows the redistribution of estuarine and freshwater fishes into the upper reaches, leaving estuarine-dependent marine species to dominate the middle and lower reaches (Harrison & Whitfield, 1995).

Whitfield (1980a) examined factors influencing the recruitment of juvenile fishes into the Mhlanga Estuary, KwaZulu-Natal. He found that the extended spawning strategy of many estuarine-dependent marine species enabled juveniles to enter the system, which was closed in winter and open in spring. High numbers of species were also reported in November, coinciding with the peak immigration period for this group of fishes (Whitfield, 1980a). Harrison & Cooper (1991) described a mechanism whereby juvenile estuarine-dependent marine fishes actively recruit into TOCEs during short open phases, even under strong current velocities. Whitfield (1980a) found that resident estuarine species also have an extended breeding season, which serves as a buffer against sudden breaching events. Many species were found to breed during the stable closed phase enabling fry to utilise a winter peak in zooplankton.

The distribution of many fish species in the Mhlanga was also governed largely by the abundance of the preferred food items (Whitfield, 1980b). The most important food source in the Mhlanga was benthic floc (detritus and associated micro-organisms), which supported more than 90% of the fish biomass in the system (Whitfield, 1980c). The highest standing crops of fish food resources were also found to occur during the closed phase and this was attributed to the stability of the physical environment. Following breaching, food resources declined considerably due to the prolonged exposure of large areas of the estuary bed and the flushing effect of floodwaters (Whitfield, 1980c).

Blaber *et al.* (1984) studied the ecology of two degraded TOCEs in KwaZulu-Natal, the Mdloti and Tongati. The fish fauna of both systems were considered impoverished and were dominated by mullet. The food chain from benthic floc to iliophagous fish, however, was still found to be viable in these degraded systems (Blaber *et al.*, 1984). Ramm *et al.* (1987) documented the recovery of the degraded Sezela Estuary. This system was devoid of fish life because of chronic industrial waste discharge, primarily of an organic nature. Following the implementation of a rehabilitation programme, which included improved waste treatment, a mouth breaching policy and increased river flow into the system, improvements in water quality and fish fauna were documented. Van der Elst *et al.* (1999) have also reported on a multi-disciplinary study to rehabilitate the Siyaya catchment and estuary. Agricultural activities in the catchment had resulted in increased siltation and infilling in the estuary. This, together with reduced recruitment from the sea, resulted in an increase in freshwater fish species reported in the estuary (van der Elst *et al.*, 1999).

10) Birds

Reviews of estuarine avifauna are restricted to the earliest synthesis by Siegfried (1981) and a more recent assessment by Hockey & Turpie (1999). In neither case was there any attempt to contrast permanently open estuaries and TOCEs and the following is an attempt to extract the relevant details from these accounts and supplement them with likely scenarios on the basis of described physico-chemical differences between the two estuarine types.

One hundred and twenty seven species of birds have been recorded from the estuaries of South Africa and of these, 68 species feed on invertebrates (Siegfried 1981). Many of the 68 are wading types, which migrate to southern Africa during the austral summer from breeding grounds in the northern hemisphere. The vast majority of these use intertidal areas in estuaries where they feed on surface or shallow burrowing organisms and consequently play little role in the functioning of closed, non-tidal estuaries. The major food resource for birds in closed estuaries is the fish fauna and consequently the major bird species using these habitats are piscivorous. The fish resource supports a variety of species that use different feeding techniques. Grey *Ardea cinerea* and goliath herons *Ardea goliath*, little *Egretta garzetta* and great white egrets *Casmerodius albus* wade and spear fish in the shallows. The kingfishers, typically malachite *Alcedo cristata*, pied *Ceryle rudis* and giant *Ceryle maxima* feed by diving and snatching fish near the surface with their bills. Larger fish are taken from the surface by fish eagles *Haliaetus vocifer* and ospreys *Pandion haliaetus* using their talons. White *Pelicanus onocrotalus* and pink backed *Pelicanus rufescens* pelicans also take fish near the surface although these two species are relatively uncommon in TOCEs. Deeper swimming fish are taken by reed *Phalacrocorax africanus* and white breasted cormorant *Phalacrocorax carbo* and darters *Anhinga melanogaster*. None of these piscivores is migratory although they are obviously highly mobile and able to take advantage of systems where successful recruitment of juvenile fish has occurred, as noted by Blaber (1973) in the west Kleinemonde where white breasted cormorant significantly reduced the population of *Rhabdosargus holubi*.

Clearly all of the above species could also utilize open estuaries in which the fish resource is likely to be more consistent, but as they are all visual feeders, the lack of currents in closed systems and the resulting greater water clarity is likely be an attractive feature.

11) Management issues

A recent detailed review of the management aspects of South African estuaries is given by Morant & Quinn (1999). Apart from some of the more general and typical management problems, TOCEs are also exposed to some unique issues related to their periodical isolation from the oceanic environment. A preliminary assessment of the conditions of all South African estuaries was made by Whitfield in 1995. This indicates that 112 (61.5%) out of a total of 182 TOCEs are in good to excellent conditions, with the majority of them situated in the former homeland regions (i.e. Transkei & Ciskei), where little industrial, agricultural and residential development would have occurred compared to the rest of the coastline. The remaining 70 TOCEs (38.5%) fall within the fair to poor category and are mainly situated in KZN (44). A more recent and thorough assessment was also carried out by Harrison *et al*. (2000), on the basis of estuarine geomorphology, ichthyofauna, water quality and aesthetics. In this case, only 23 TOCEs scored good to excellent on all accounts, while 14 TOCEs scored moderate to very poorly. All the others exhibited either a mixture of low and high scores (92) or had not been sampled for all the components required (53).

Under the assumption that the current concept of sustainable management for TOCEs is aimed at protecting their ecological integrity, biodiversity and nursery function, the following can be regarded as key issues.

a) Artificial breaching

The interference with the mouths of estuaries by artificial breaching and the construction of sandbars is known to have adverse effects on the natural functioning of such systems (Begg 1984, de Decker 1987, Morant & Quinn 1999, Harrison & Connell 2002). A study of KZN estuaries has highlighted the broad ecological implications of natural and artificial breaching (Begg 1984). In the Wastern Cape, artificial breaching of the Bot River Estuary was found to have an immediate impact on the resident benthic community as well as a long-term deleterious effect on the species richness and diversity (de Decker 1987). Frequent artificial breaching can indeed have serious negative effects on estuarine ecosystems, but occasional artificial breaching as a substitute for natural breaching is not necessarily bad. Breaching is a natural phenomenon and is essential in order to establish periodical connection with the sea, thereby allowing the immigration of juvenile fish and invertebrates, tidal exchanges and the re-establishment of salinity gradients along the estuary. Reduction in river flow has often resulted in serious reduction in the occurrence of open mouth conditions. Under these circumstances, artificial breaching can be used in a controlled way to increase the occurrence of open mouth conditions, in order to approach the natural frequency of breaching.

Many examples of urban encroachment exist, particularly in KZN TOCEs. The direct impacts of this poor urban planning are obvious, with loss of habitat from activities such as permanent hardening of shorelines, construction of jetties and launch ramps, development of large residential complexes and the destruction of riparian vegetation. One of the major indirect impacts however, is the breaching of estuary mouths to lower water levels in order to prevent flooding and damage to badly sited floodplain infrastructure. Jurisdiction over estuaries and management of estuary mouths in KZN is currently vested with Ezemvelo KZN Wildlife (EKZNW) by a 1994 KZN amendment to the Seashore Act 21 of 1984. The increase in development (including illegal and inappropriate historical) and population pressure along the KZN coastline has resulted in a concomitant increase in breaching requests to management staff as well as frequent illegal breaches of a number of estuaries along the KZN coast. Social conflict between various stakeholder groups is now becoming more common as landowners and various user groups clamour for artificial breaching in opposition to the advice and concerns of conservation agencies, estuarine scientists and various environmentally aware organizations. The understanding of estuarine function has however advanced significantly since the 1980's and the addition of new legislation such as the National Water Act 36 of 1998 and the pending Coastal Management Act has further enhanced the need and provided a potential mechanism for increased management of this potentially detrimental activity.

While it is true that a combination of an open mouth, tidal action and freshwater input are generally positive factors in estuarine function this should not be interpreted as implying that the frequent closure of many estuarine systems, both large and small, renders them biologically or ecologically insignificant and therefore less deserving of sympathetic development. A variety of fish species as well as invertebrates such as the sand prawn and the mangrove crab all survive quite adequately in TOCE systems and may in fact be favoured by the conditions that exist during such periods. The life cycles of many of these species are geared to the natural cycles of opening and closing, and arbitrary artificial breaching can adversely influence the successful completion of these cycles.

TOCEs provide a range of goods and services, which require careful management to ensure their long-term sustainability. Available evidence strongly indicates that artificial breaching may have long term impacts upon the sediment dynamics, biota and basic functioning of an estuary, seriously jeopardizing its ability to provide these good and services. Artificial opening of an estuary mouth, when the water level is below that at which breaching naturally occurs, results in a reduced scour potential. In the long term this leads to accumulation of sediments in the estuary mouth, thereby compounding the original problem.

The effect of artificial breaching on the biota is frequently more severe. Past studies in KZN have suggested that the community structure of the benthos of the Mdloti Estuary was greatly altered by the frequent artificial breaching of the lagoon observed 16 times over a two year study period (Begg 1984). This is supported by more recent studies on the meiofauna of the Mdloti Estuary, which have shown that the physico-chemical changes associated with breaching, impact heavily on meiofaunal abundance and biomass (Nozais *et al.* in press). Immediate negative impacts on the resident plankton and benthos in TOCEs are also evident after breaches, as well as longer term effects which are the more important in terms of species composition (Perissinotto *et al.* 2002, Jairam 2002). The impacts of artificial breaching on the macrofauna of the Bot River estuary have been described as "disastrous" (De Decker 1987), with repeated and unpredictable breaches producing an unstable system with very low species diversity. This is indicated by the presence of only 27 benthic species in this estuary, a much lower total than other comparable Eastern Cape estuaries (*e.g.* 137 spp in the Klein River), and with only six species consistently abundant and comprising 95% of the biomass.

It has previously been mooted that the icthyofauna maybe a component of the biota which benefits from the frequent opening of an estuary. There is evidence to suggest that this is in fact not the case and that the long term effects of reduced productivity and changed physico-chemical regimes may negatively impact on estuarine-dependent fish. Many of the predominantly closed TOCEs in KZN are dominated by estuarine-dependent marine fishes and provide an important nursery function for this group (Harrison & Whitfield 1995, Harrison & Connell 2002). The spawning and migration patterns of these fishes appear to be linked to the natural seasonal breaching patterns of these systems (Whitfield 1998). Spawning takes place during late autumn, winter and spring and recruitment into estuaries takes place when increased rainfall and river flow breaches the mouths of these systems (Harrison & Connell 2002). Full breaching is in fact not required, as juvenile fish are able to use wave-overtopping events at high tide as a means of entry into estuaries (Cowley *et al.* 2001). Once the estuaries close, habitat, nutrients and food availability increase thereby providing ideal conditions for growth and survival. Unseasonable flushing of these systems reduces the nursery function by removal of food resources and premature flushing of juveniles.

Artificial breaching can have similar massive impacts on the avifaunal community utilizing estuaries (Quinn 1998) and huge changes in avian biomass have been reported following breaches. A good example exists from a study on the Bot River estuary, where avian biomass fell from a peak of 40,000 kg to only 700 kg after two breaches (Heÿl & Currie 1985).

The information presented above indicates that artificial breaching can have serious long-term impacts upon the sediment dynamics and biota of an estuary. The major problem associated with artificial breaching is linked to the fact that it often occurs unseasonably. For instance, most often the highest water levels in KZN TOCEs occur during the drier winter months when these systems are closed. This results in the backflooding of inappropriately sited developments and cultivated land and concomitantly causes an increase in requests to breach, and illegal interference with the mouths of these systems commonly occurs.

b) Impoundments, abstractions & discharges from water treatment works

Combinations of abstractions, impoundments and discharges from sewage treatment works have the capacity to drastically change flow regimes, water quality and mouth behaviour, thereby affecting seasonality of habitat availability and fish migration opportunities.

Upstream impoundments can have major impacts on runoff inputs into estuaries. Total flow volumes are reduced and the temporal characteristics of the flow are altered. High peak, short duration flows have reduced peaks and longer time scales after routing through a storage reservoir. This can change the flow-duration characteristics of the inflows and thereby the mouth dynamics of TOCEs. Impoundments also change the sediment loadings in rivers by trapping sediment in the reservoirs. This may give rise to reduced sediment loads into a downstream estuary. Abstractions (e.g. for irrigation) also reduce inflows to estuaries, although in a somewhat different way. Whereas impoundments tend to have their most profound effects on high flows (floods), abstractions tend to occur mostly in low rainfall times and therefore have their most significant impact on low flows.

Discharges from wastewater treatment works can significantly increase flows into small estuaries, usually by providing a near constant "capping flow". By contributing significantly to the flow, especially in low-flow periods, these discharges can reduce residence times and play a key role in estuary inlet dynamics during those periods. These capping flows are generally small relative to flood (high) flows, and therefore do not significantly affect the influence of these flows on the estuary functioning. In assessing the affects of flow changes on the functioning of TOCEs for management purposes, it can therefore be important to distinguish the exact nature of the flow changes, as indicated in the above examples. In the case of changes in high flows, the effect on mouth dynamics can be predicted with high confidence because the water balance of the estuary is inflow dominated during high flow events (which generally lead to open mouth conditions). The affect of changes to the lower flow regimes can be more uncertain, since in those situations the water balance is no longer dominated by inflows. Outflows and losses (e.g. due to seepage and evaporation) must also be accounted for in order to predict the affects on water levels and mouth dynamics.

Discharges from sewerage treatment plants can also severely impact the inorganic chemistry of estuarine waters. Nutrient loading can be particularly drastic and result in widespread eutrophication of the entire estuary. This is especially likely to happen during the closed phase of a TOCE, when the residence time of waters inside the estuary exceeds the time scale of growth of the local microalgal community (Allanson & Winter 1999). With very prolonged residence times, eutrophication will eventually lead to an escalation in the biological oxygen demand thoughout the water-column and eventually to the establishment of anoxic conditions in the deeper layers. Mass mortality of fish and crustaceans may follow.

c) Catchment land use

Deforestation, wetlands destruction and overgrazing in catchment areas are major causes of soil erosion in South Africa. The consequences of this are particularly important for TOCEs, as increased sedimentation and silt loading can have dramatic effects on benthic communities and can also reduce the availability of radiation for photosynthetic microorganisms within the water-column

(Morant & Quinn 1999, Nozais et al. 2000). Land use practices can also impact on the runoff characteristics (volumes and time scales) of TOCEs.

These issues are particularly important for the management of TOCEs because of their sensitivity to factors that influence mouth dynamics. Most TOCEs in South Africa have small catchments, which enhances their sensitivity to land use changes. It is therefore particularly important for management decision-making concerning TOCEs to adopt a holistic approach that includes the entire catchment system. Midgely *et al* (1994) apply rainfall-runoff modelling to estimating the affects of land-use changes on river flows, which allows simulated natural streamflow to be used as a reference condition for management decisions. There can be large uncertainties in estimating the affects of land-use changes, but procedures to do so are included in Midgely *et al* (1994). More recently Smakhtin (2000) discussed simple methods to include the effects of land use changes into flow-duration analyses.

Estuarine water quality can also be affected by leaching of fertilizers and pesticides from the numerous types of plantations that have replaced the natural vegetation in most catchments. Citrus, pineapple and sugar cane are probably the cultures most directly responsible for the agrochemical pollution of South African TOCEs (Begg, 1978, Morant & Quinn 1999).

d) Flood plain encroachment

Due to the shelter they provide against coastal winds and waves, TOCEs have become favourite sites of residential developmnent. On the western part of the Eastern Cape and on the KZN south coast the process has escalated to the point that most TOCEs in the area now exhibit urban/resort settlements, often with dense human populations (Morant & Quinn 1999).

Encroachments on the flood plains of estuaries for agriculture are common in the South African context, e.g. in KZN sugar-cane has often been grown right down to the banks of the normal river channel. Such developments generate pressure for artificial mouth breaching during times of high water levels (i.e. closed mouth conditions) at TOCEs.

Other encroachments on the estuarine flood plain, such as those due to the construction of road/rail bridges and their associated approach embankments (or other similar structures) introduce local scour and sedimentation patterns that could influence mouth-breaching patterns. Since mouth dynamics is a critical driver of the physico-chemical environment of TOCEs, such factors can have a severe impact on the overall functioning of the estuaries concerned, and should be included in future management decisions. Unfortunately, this was generally not the case in the past and a large number of road railway and pipeline bridges were constructed over TOCEs, often very close to the sea, thus compromising the ecological functioning of entire estuarine sections (e.g. Heydorn & Bickerton 1982).

e) Sand-winning operations

Sand-winning operations are widespread on South African rivers, particularly in Kwazulu-Natal (KZN). The extraction of sand from KZN estuaries and rivers for building purposes appears to be largely an economic activity in these systems. A recent study has shown that 99% of all sandwinning in KZN occurs in river systems (Mundree 2003), typically in the lower reaches but at times extending into the estuary, as in the Mpenjati. With the growing demand for sand in the building industry, sandwinning in estuarine systems is likely to become more widespread.

The excavation of large amounts of sand not only increases water turbidity but also results in the removal of many ecologically important sediment-associated micro-organisms from these systems. Thus sandwinning has a direct impact on the structure and biodiversity of important groups, such as microphytobenthos, meio- and macrofauna, and a cascading negative impact on the riverine and estuarine ecosystem functioning. The higher trophic levels, like fishes and birds, are generally driven out of the system due to the noise and the sediment disturbance generated by the sandwinning operation. Although the national Department of Minerals and & Energy (DM&E) does have specific guidelines in place to ensure minimum negative impact on the environment, as well as proper rehabilitation of the environment upon closure of the sandwinning operations, the destruction of riparian and associated vegetation is quite extensive. This may lead to the proliferation of invasive alien species in disturbed areas and also result in the modification of existing channels. It is important to note that while riparian and estuarine vegetation can be rehabilitated during and at the closure of a sandwinning operation, the loss of key components of the biotic ecosystem may be irreversible. Other problems directly associated with sandwinning are pollution via discarded machinery, spilt fuel and high noise levels. Also, the production of irregular unstable areas and holes in the channels may cause dangers to local people

Suitable guidelines for management would include the thorough assessment of the ecological sensitivity of the proposed sandwinning site prior to a permit being issued. Currently, the assessment is only based on the impact on the marginal and adjacent vegetation, with virtually no consideration for the biotic components of the water ecosystem itself. Both systems need to be taken into account. Another important recommendation would be to ensure that there is regular monitoring of the river/estuary in which sandwinning is occurring. This would ensure the sustainability and the protection of the biodiversity of these ecosystems and would also help to address any problem that may arise, either through direct or collateral impact.

f) Exploitation of bioresources and recreational activities

South Africa has not been blessed with an abundance of large estuaries, and in the global context, with the exception of systems such as Knysna on the south coast and the St Lucia Lake and Kosi systems in northern KwaZulu-Natal, does not have any estuaries of any size at all. In addition, its open, high energy coastline, while often spectacular, can not be described as user-friendly in comparison with the coastal waterways of, *e.g.* the eastern U.S.A. and parts of eastern Australia. South Africa's natural inland water bodies are few and far between, and those inland water bodies which do exist are very largely artificial impoundments. It is very noticeable, however, that even the smallest of these water bodies present attractions to the boating and/or fishing community, a fact which immediately presents management problems arising out of over-exploitation of living resources and user conflict. Estuaries, including TOCEs, many of which constitute very small bodies of water, have not escaped this onslaught and where access and depth of water permit one can expect that boating, fishing, water ski-ing and the use of jetskis and wetbikes will very often be found sometimes on the same systems.

Fishing or the indications of fishing activity in the form of litter, car tracks or campfire sites are generally apparent. Bait, typically the thalassinid sandprawns *Callianassa kraussi* in closed systems, is extensively collected and the evidence of this activity in the form of disturbed sediments is often visible. At best fishing is done using a rod and line, but a developing problem with far-reaching implications in many TOCEs is the use of gill nets. Lamberth and Turpie (2003) report that gillnetting is currently responsible for the largest annual fish catches in South African estuaries, accounting for about 47% of the total. The small size and shallow nature of most of the TOCE systems, which makes virtually the entire habitat accessible and vulnerable, present a particular problem.

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STUDY RESULTS

Responses of the biological communities to flow variation and mouth state in two temporarily open / closed KwaZulu-Natal estuaries

ABSTRACT

An unusual rainfall pattern was observed during the year of this study, with anomalous high precipitation recorded during the winter (July 2002). This was reflected in the state of the mouth of the Mdloti and Mhlanga, which experienced prolonged periods of open phase during the winter and unusual closed phases during part of the summer. Despite their largely different catchment areas (Mhlanga ~ 100 km^2 ; Mdloti ~ 500 km^2), the two estuaries exhibited very similar proportions of open versus closed mouth periods. However, in terms of breaching events, 16 of these were recorded at the Mhlanga and 9 at the Mdloti. This was largely due to the different capping flows the two systems receive from sewage treatment plants situated in their upper reaches. While the Mhlanga receives an average of 20 Ml of treated water per day, the Mdloti receives only about 8 Ml per day. Sewage discharges not only affect the residence time of water in the estuaries, but also contribute large amounts of macronutrients to the estuarine waters.

The following is a concise summary on the state of the hypotheses that were posed at the onset of this project.

PHYSICO-CHEMICAL ENVIRONMENT

"*With summer rainfall and increased river flow there is an input of nutrients and suspended solids, a decrease in salinity, an increase in water level, the mouth opens but later, due to wave action and sediment supply the mouth closes*"

High rainfall and high flows are statistically linked to an open mouth state, but the relationship between flow and mouth state is indirect i.e. breaching occurs at specific (high) water levels, that are in turn driven by the full water balance of the system, not only inflows. Rainy periods are not necessarily associated with high water levels, nor with low salinity, since if mouth breaching occurs it dramatically lowers water levels and during the open state tidal exchange flows lead to significant influxes of saline water. River flows that have residence times of order 5 days or less, seem to be able to sustain open mouth conditions. This time scale is representative of the time required for mouth re-closure due to sediment redistribution by wave action. Sediment supply due to longshore drift (estimated to exceed 1000 m^3/day) is more than sufficient to close the small inlets that are typical at these estuaries when they breach. The mouth re-closure process is controlled by cross-shore sediment transport (estimated to be of order $15m^3/day/m$) in these cases. Other key findings from the hydrodynamic investigation include the following.

1) Water levels at these perched estuaries ranged from ± 700 mm above MSL (fully open state) to ± 3200 mm above MSL (fully closed). Breaching water levels (and the height of the berm) are about 1.2 times the median significant wave height above spring high tide levels.

2) Mouth breaching is driven directly by high water levels and associated steep hydraulic gradients across the berm. Seepage through the sandbar plays an important role in the process for these highly perched estuaries. Maximum seepage outflows were estimated as 0.25 m^3/s and 0.50 m^3/s at the Mhlanga and the Mdloti respectively, and effectively define critical inflow magnitudes below which breaching does not occur.

3) Tidal exchanges at the Mhlanga during the open phase were 5–30% of the estuary volume, despite the highly perched nature of the estuary. This is due to wave run-up on the steep beaches, which effectively increases tide levels by half the median significant wave height.

4) Mouth re-closure at the Mhlanga took 3 – 6 days under low flow conditions, and occurred at high tide, thereby trapping significant volumes of saline water in the estuary.

MICROPHYTOBENTHOS

"*As a result of increased freshwater flow, microphytobenthos biomass increases due to an increase in nutrient availability, an increase in light and a decrease in grazing. This increase is at a maximum two weeks after mouth closure*".

In the Mdloti Estuary, an increase in freshwater flow, and nutrient availability (dissolved inorganic nitrogen, DIN, and phosphorous, DIP), was related to higher microphytobenthic biomass, particularly during the closed phase. The possibility that grazing pressure exerted by the zooplankton community may have influenced the microphytobenthic community seems likely, as zooplankton abundance and biomass was consistently correlated to microphytobenthic biomass throughout the study period. However, during the open phase of both estuaries a decline in the zooplankton community resulted in an increase in microphytobenthic biomass. Light attenuation does not seem to have played a major role in limiting this biomass during the period of the study, either at the Mdloti or the Mhlanga. Although the precise time-scale required by the local benthic microalgae to reach a maximum peak has not been determined yet, the maximum level observed during the survey (601 mg $chl\text{-}a.m^{-2}$) occurred in December 2002 at the Mdloti, after a period of mouth closure of about 12 days. The highest peak at the Mhlanga, with 313 mg $chl\text{-}a.m^{-2}$, was recorded in June 2002, during an extended period (approx 10 days) of partly open mouth conditions.

PHYTOPLANKTON.

"*As a result of increased freshwater flow, phytoplankton biomass increases due to an increase in light and an increase in nutrient availability following freshwater and seawater mixing, to a maximum of two weeks after mouth closure. Subsequently, grazing will reduce phytoplankton three weeks after closure*".

The lowest phytoplankton biomass occurred during the open phase in both estuaries, while the maximum values occurred during their closed phase. A significant inverse relationship was found between rainfall and phytoplankton biomass during the open phase of the Mdloti Estuary ($r = -0.643$, $p < 0.01$), suggesting that an increase in freshwater flow does not necessarily result in an immediate increase in the biomass of phytoplankton. The most immediate effect of increase in freshwater flow is a decrease in the penetration of light because of increased turbidity. This inhibits the growth of the phytoplankton community. However, data do not provide conclusive evidence for the hypothesis that an increase in light would result in an increase in phytoplankton biomass. Positive correlations between water-column phytoplankton biomass and zooplankton abundance occurred in both estuaries [Mdloti: $r = 0.334$, $p < 0.05$ (sled zooplankton); $r = 0.499$, $p < 0.01$ (WP2 net zooplankton); Mhlanga: $r = 0.534$, $p < 0.01$ (sled zooplankton); $r = 0.455$, $p < 0.01$ (WP2 net zooplankton)]. The impact of top-down control of phytoplankton biomass is strongly suggested by this relationship between phytoplankton biomass and zooplankton abundance, however, no precise time-scale for the maximal impact of grazing pressure on the phytoplankton community could be determined.

ZOOPLANKTON.

"*As a result of increased freshwater flow, zooplankton biomass increases due to an increase in phytoplankton biomass to a maximum four weeks after mouth closure. Freshwater inflow and salinity dropping to < 10 ‰, stimulates hatching of resting copepod eggs after one week*".

Zooplankton abundance and biomass were always higher during the closed phase in both estuaries, as compared to their open phase. This could be attributed to the stability of the system during the closed phase, due to reduced freshwater input and the restricted exchange of water with the sea. There was a consistent correlation between zooplankton abundance/biomass and microalgal biomass. This suggests that zooplankton responds positively to an increase in benthic and pelagic microalgal biomass, both in the Mdloti and the Mhlanga estuaries.

The highest zooplankton levels coincided with periods of prolonged mouth closure. These were very limited at the Mhlanga because of its high frequency of breaching. At the Mdloti, the highest biomass/abundance values were recorded in February 2003, following 22 days of mouth closure. This is about twice as long as stated in the initial hypothesis. Mass hatching of copepod nauplii have been observed a few times after major rainfall events. However, any causal effects with salinity drops have been difficult to test because of the predominant low salinity conditions encountered in the Mdloti. There was a consistent time lag in the response of copepod nauplii abundance/biomass to rainfall. Copepod nauplii attained peak levels within a period of 2-4 weeks following a major rainfall event, with temperature and state of the mouth being the main factors controlling the delay in this response. However, any causal effects with salinity drops have been difficult to test because of the predominant low salinity conditions encountered particularly in the Mdloti.

MACROBENTHOS

"*After the open state, a different macrobenthic community develops, with filter feeders becoming dominant due to food availability by three months after mouth closure. Overtopping of the berm at the mouth and wind resuspension of material within the estuary provides food for filte feeders. Overtopping of the berm is important for the recruitment of estuarine dependent fish and invertebrates*".

The hypothesis that closure would result in a different macrobenthic community after three months has not been demonstrated nor tested, because there has not been an unbroken three month closure period during the study. The Mhlanga was in fact open, or had at least some link with the sea for about half of the study period; closure seldom lasted longer than a month. The Mdloti was closed for longer periods, but again there were no extended periods of closure prior to any macrofaunal survey. Analysis of variation in abundance of the macrobenthos showed that greater abundance did follow periods of closure. The erratic nature of the open/closed periods and the difficulty of obtaining good data on the behaviour of the sand bars means that this relationship would require further testing. The berm-overtopping study showed minimal recruitment of invertebrates during the closed phase, as only a few ghost crabs *Ocypode spp.*, mole crabs *Emerita analoga*, nematodes and siphonophorans were collected during the six sampling exercises undertaken at the Mhlanga and the two at the Mdloti during the year.

FISH

"*The composition of estuarine-associated marine fish species in TOCEs is directly linked to the seasonality and duration of the open mouth condition. The mouth must remain open for a minimum of 7 days during spring and summer in order to ensure optimum recruitment. The longer the mouth stays open the greater the diversity and abundance of fish composition. However, overtopping of the berm does provide a migration route for the recruitment of estuarine dependent fish*".

In both systems, lowest diversity was found in late winter (September survey), after both systems breached in July/August. During all four surveys, biomass was dominated by freshwater (*Clarias* and *Oreochromis)* species, and mugilids. The longest open period (July/August 2002) was followed by the lowest diversity found during the survey (September 2002), which may simply reflect a difficulty of sampling new recruits with seine nets, particularly if the bank profiles are steep.

Overtopping was indirectly demonstrated, when six 20 mm *Diplodus sargus* were netted in the Mhlanga while storm-swell overtopping was occurring (September 2002). Direct, conclusive evidence of recruitment through wave overtopping was also obtained at the Mhlanga, where a few juveniles of three fish species were netted in the incoming waves at the peak of the spring high tide in August and September 2003.

BIRDS

"*The open mouth state will increase the diversity and abundance of bird species due to an increased availability of intertidal/exposed habitat*".

The hypothesis that an open mouth condition would increase the diversity and abundance of the avifauna, and particularly species using inter-tidal areas, was not supported. The only wading birds recorded were four to six common sandpipers *Actitis hypoleucos* in the mid reaches and occasional greenshanks *Tringa nebularia* and grey plovers *Pluvialis squatarola* in the lower reaches of the Mhlanga during summer open mouth periods, but this did not constitute a mass influx. Much of the areas exposed after breaching became supra-tidal rather than inter-tidal and the resulting desiccation would rapidly kill any of the small, shallow-burrowing invertebrates species that occur in these areas when submerged.

INTRODUCTION

The river–estuary interface (REI) has been defined for South African permanently-open estuaries as that area that experiences a salinity drop below 10 ‰ (or $g.l^{-1}$, Bate *et al.* 2002). This is the section of the estuary that coincides approximately with the more traditional definition of middle reaches (Day 1980) and exhibits unique physico-chemical and biological characteristics. In particular, phytoplankton and zooplankton biomass are enhanced at the REI and invertebrate community structure and estuarine-associated fish species distribution are remarkably different in this region, compared to both upper and lower reaches (Bate *et al.* 2002).

In permanently-open estuaries, the size and properties of the REI are determined by a range of factors, including: a) rate and duration of river inflow; b) tidal prism and phase; and c) physico-chemical characteristics of the inflowing river water type. Some of these factors, and especially the rate of freshwater inflow, have been the subject of targeted studies in recent years (see for instance Hilmer 1990, Allanson & Read 1995, Snow *et al.* 2000, Whitfield & Wood 2002). However, no such studies have yet been conducted on the more abundant temporarily open/closed estuaries (TOCEs), which constitute 73% of the total number of South African estuarine systems. Variations in river inflow rates have fundamental effects on the dynamics of TOCEs, as these determine the open versus closed state of their mouth. Reduction in river flow is generally associated with prolonged periods of mouth closure, while increases in freshwater runoff and river inflow can eventually lead to the breaching of their mouth and a variable period of free communication with the ocean. The effects of these two opposite phases on the physico-chemical and biological characteristics of these estuaries are still largely unknown.

The National Water Act (36) of 1998 requires that the amount and quality of water needed by all aquatic ecosystems must be determined. Interference with the natural quantity and quality of river water delivered to an estuary alters the size and character of the REI. This, in turn, influences the structure of the biota. A sound understanding of the processes taking place in the REI is essential to recommend the correct amount of river water required for an acceptable estuary ecological status and, therefore, the correct ecological category in the reserve/freshwater requirement assessment.

The possible application of the criteria identified above for the determination of the REI in TOCEs was explored during a CERM/WRC meeting held in Durban in October 2001. This aimed at identifying research procedures to determine the importance of the REI zone in TOCEs. However the predominantly perched nature of TOCEs in KwaZulu Natal, prevents a distinct REI (*sensu* Bate *et al.* 2002) from forming within these systems. Because of the seasonal rainfall, there is more of a temporal than a spatial response to freshwater input and mouth condition in these estuaries, by comparison with their counterparts in the Cape. TOCEs seem to experience a period of biological rejuvenation some time after natural breaching, followed by a period of maximal productivity shortly after the re-closure of the mouth (Whitfield 1992, Nozais *et al.* 2001). Unnatural freshwater flow variations and artificial breaching may have a negative impact, by inducing changes in the frequency, timing and duration of closed mouth conditions, which in turn will interfere with the migration patterns of biota between the ocean and the estuary. The effect of such changes on the ecological functioning of the estuary needs to be determined in order to predict the changes in the ecological status of the estuary by comparison with the natural state.

The project title was therefore revised to reflect these differences, between permanently-open and TOCE systems on one side, and Cape versus KZN TOCEs on the other. The revised project title is **"Responses of the biological communities to flow variation and mouth state in temporarily open/closed estuaries."** The two temporarily open/closed estuaries chosen for this study were the Mdloti and Mhlanga estuaries, both situated on the KZN north coast, a mere 7.5 Km apart.

The aim of this project is to provide an improved understanding of the role and importance of freshwater flow, mouth condition and biological response, as well as the role and importance of the mouth as a migratory route in TOCEs. The overall objective of the project was to assess the role of river flow variation and mouth state (open/closed) in the structure and functioning of TOCEs on a temporal and spatial scale. The specific hypotheses to be tested can be summarized as follows.

1) Physico-chemical environment. With summer rainfall and increased river flow there is an input of nutrients and suspended solids, a decrease in salinity, an increase in water level, the mouth opens but later, due to wave action and sediment supply, the mouth closes.

2) Microphytobenthos (MPB). As a result of increased freshwater flow, MPB biomass increases due to an increase in nutrient availability, an increase in light and a decrease in grazing. This increase is at a maximum two weeks after mouth closure.

3) Phytoplankton. As a result of increased freshwater flow, phytoplankton biomass increases due to an increase in light and an increase in nutrient availability following freshwater and sea water mixing, to a maximum two weeks after mouth closure. Subsequently, grazing will reduce phytoplankton biomass 3 weeks after closure.

4) Zooplankton. As a result of increased freshwater flow, zooplankton biomass increases due to an increase in phytoplankton biomass to a maximum four weeks after mouth closure. Freshwater inflow and a sharp salinity drop stimulate hatching of resting copepod eggs after one week.

5) Macrobenthos. After the open state, a different macrobenthic community develops with filter feeders becoming dominant due to food availability by three months after mouth closure. Overtopping of the berm at the mouth and wind re-suspension of material within the estuary provides food for filter feeders. Overtopping of the berm is important for the recruitment of estuarine dependent fish and invertebrates.

6) Fish. The composition of estuarine-associated marine fish species in TOCE is directly linked to the seasonality and duration of the open mouth condition. The mouth must remain open for a minimum of 7 days during spring and summer in order to ensure optimum recruitment. The longer the mouth stays open the greater the diversity and abundance of fish composition. However, overtopping of the berm does provide a migration route for the recruitment of estuarine dependent fish.

7) Birds. The open mouth state will increase the diversity and abundance of bird species due to an increased availability of intertidal / exposed habitat.

MATERIALS & METHODS

Field operations started in March 2002 and continued at monthly intervals until March 2003 for the physico-chemical, microalgal and zooplankton easurements/collections. Macrobenthos, fish and bird surveys were carried out with quarterly frequency, i.e. March, June, September & December 2002 and finally March 2003. All activities were co-ordinated in a multidisciplinary approach, with each estuary being investigated by all members of the team during the same day on each survey. Day one of the survey was normally dedicated to the Mdloti Estuary and day two to the Mhlanga Estuary. A summary of the dates and estuarine conditions for each survey undertaken is presented in Table 1. In order to minimize tidal effects during the open phase, all surveys dates were predetermined to coincide with the middle of a neap tide. Measurements of hydrodynamic conditions continued at both estuaries until September 2003, while the recruitment via wave-overtopping study was conducted during each spring high tide from August to November 2003 (i.e. 2 surveys per month). Details on the approach employed in each specific collection/measurement are provided below.

PHYSICO-CHEMICAL ENVIRONMENT

The state of the mouth of both estuaries was monitored on a daily basis by field rangers of Ezemvelo KZN Wildlife (Durban) and independent residents in the area. Daily rainfall data were provided by the nearby South African Sugar Association Experiment Station, Mount Edgecombe and by the South African Weather Service, Durban International Airport.

Flows were obtained by "velocity-area" integration: A cross-section of the river was first measured at the measurement station, and then velocity measurements were made at selected positions and depths (usually at 20%, 80%, and/or 60% of the water depth) to get the average velocity at each section. The average velocities and cross-section data were then integrated numerically to determine the flow rate.

Velocities were measured using drogues (comprising a float & a drag vane) that were timed over a 10m fetch. The position of the drag vanes below the surface was adjusted to allow velocities at specific depths to be measured.

A disadvantage in the use of floats is possible interference from surface winds. A SWOFFER propeller-type flow measuring system became available in February 2003 and was used for all monitoring during 2003. This improved the accuracy of the flow measurements.

It is estimated that the drogue measurements had an accuracy ranging from 10% to 50%, with the least accurate measurements at the lower flow magnitudes. The measurements with the Swoffer are estimated to be accurate to within 10%.

Measurements were, whenever possible, made on relatively straight, narrow, reaches of the river with well-defined banks. It was usually necessary to locate the measurement stations 1 – 2 km upstream in order to obtain flow velocities that were large enough to be measurable. GPS co-ordinates of all measurement stations were recorded (Zietsman, 2003)

Water levels were recorded using digital pictures of the M4 road bridges at each estuary. The pile caps of the bridge piers were used as an interim datum. An accurate GPS survey (±10mm) was subsequently carried out to adjust the data to MSL as datum. The monthly measurements from the main field trips were supplemented (where possible) with additional weekly data. Continuous water level monitoring devices were requested from DWAF, who agreed to install them, but no action was taken. The lack of continuous water level monitoring was a setback for the project – the detailed temporal resolution of mouth breaching and closure was compromised as a result. In order to rectify this, we developed miniature submersible water level monitors based on pressure transducers and with digital data loggers. Since the sensors were not vented all measurements were corrected for atmospheric pressure variations. Estimated accuracy is ± 25mm. Details concerning the development of this device are given in Zietsman (2003). Prototypes were built and installed in each of the case study estuaries in March 2003 and provided continuous water levels (logged at hourly intervals) since that time. Note that although regular biological monitoring essentially ended in March 2003, it was felt that continued hydrodynamic monitoring was necessary in order to gain a better understanding of the mouth dynamics of the systems, particularly Mhlanga estuary.

Table 1. Monthly surveys undertaken during the period of the study, March 2002 - March 2003, with details of mouth conditions at and prior to the time of the survey.

MARCH 2002	Mdloti, 19.03.2002	Fully open, breached 5 days before survey
	Mhlanga, 20.03.2002	Closed, closed for previous 10 days
APRIL 2002	Mdloti, 18.04.2002	Open, open for previous 5 days
	Mhlanga, 19.04.2002	Open, open for previous 8 days
MAY 2002	Mdloti, 22.05.2002	Closed, closed for previous 12 days
	Mhlanga, 21.05.2002	Open, opened artificially the previous day (?)
JUNE 2002	Mdloti, 20.06.2002	Closed, closed for previous 40 days
	Mhlanga, 19.06.2002	Partly open, open for previous week
JULY 2002	Mdloti, 17.07.2002	Closed, closed for previous 67 days
	Mhlanga, 18.07.2002	Open, open for previous 15 days
AUGUST 2002	Mdloti, 15.08.2002	Partly open, open for previous 25 days
	Mhlanga, 16.08.2002	Open, open for previous 25 days
SEPTEMBER 2002	Mdloti, 12.09.2002	Partly open, open for previous 2 weeks
	Mhlanga, 13.09.2002	Just closed, partly open for previous 3 days
OCTOBER 2002	Mdloti, 16.10.2002	Just breached, closed for previous 10 days
	Mhlanga, 15.10.2002	Closed, closed for previous 9 days
NOVEMBER 2002	Mdloti, 12.11.2002	Closed & very full, closed previous 11 days
	Mhlanga, 13.11.2002	Partly open, open for previous 9 days
DECEMBER 2002	Mdloti, 09.12.2002	Closed, closed for previous12 days
	Mhlanga, 10.12.2002	Closed, partly open for previous 3 days
JANUARY 2003	Mdloti, 22.01.2003	Closed, closed for previous 8 days (?)
	Mhlanga, 23.01.2003	Open, open for previous 5 days
FEBRUARY 2003	Mdloti, 19.02.2003	Closed, closed for previous 10 days
	Mhlanga, 20.02.2003	Closed, closed for previous 11 days
MARCH 2003	Mdloti, 25.03.2003	Closed, closed for previous 44 days
	Mhlanga, 26.03.2003	Closed, open/partly open for previous 5 days

Every field survey included vertical profiles of depth, temperature, salinity, dissolved O_2 and pH made with a YSI 6920 Water Logger. Water Logger measurements were taken once a month at all three stations in each estuary, i.e. mouth, middle and head. At each station and on each occasion, a vertical profile of downwelling irradiance (PAR, 400 to 700 nm) was also measured with a LI–COR LI–189 underwater, spherical quantum sensor at the subsurface and every 50 cm to the bottom. The diffusive attenuation coefficient, K_d (m^{-1}) was then estimated using the relation: $K_d = -\ln (Iz_2 / Iz_1) / z_2 \; z_1$; where Iz_2 = irradiance ($\mu mol \; m^{-2}s^{-1}$) at depth z_2 (m), Iz_1 = irradiance at depth z_1.

Macronutrients were sampled at both estuaries with the same spatio-temporal resolution. In the water-column, samples for DIN (ammonia + nitrate) and DIP (orthophosphate) were taken at subsurface levels using a 1-liter pop-bottle. Similar samples of pore water were collected within the sediment (5-10 cm below sediment-water interface), using a hand-operated vacuum pump extractor connected to a stainless-steel, closed pipe inserted into the sediment. A series of tiny holes were drilled at the lower tip of the pipe, in order to ensure the inward flow of pore water while minimizing the passage of fine sand and silt particles. All water-column and pore water samples were immediately placed in 500 ml acid pre-washed polyethylene bottles and their macronutrient concentrations were later determined using a Technicon II Autoanalyser at the CSIR-Environmentek in Durban.

Sediment samples for granulometric analysis were taken during the quarterly surveys, from the three macrobenthic samples collected at each station (see also section on macrofauna below). The particle size composition was determined by standard wet sieving techniques using sieves in a geometric series. Cumulative curves were plotted and the median phi value determined from the plot. Organic content was determined by weight loss following incineration of a dried, pre-weighed sample at 600°C.

Microalgae

Microalgal collections at both estuaries were done at monthly intervals and at all three major stations, i.e. lower, middle & upper reaches. Size-fractionated (pico-, nano- & microplankton) measurements of phytoplankton biomass were obtained by sequential filtration of 200 ml samples onto 20 µm Nitex, 2 µm Isopore and GF/F glass-fibre filters. Samples were collected just below the water surface and just above the water-sediment interface. Chlorophyll-a and phaeopigments extraction was done in 90% acetone (10 ml).

Triplicate samples of benthic microalgae were collected with a 20 mm internal diameter twin-corer and only the upper 1 cm layer of sediment was retained for extraction of chlorophyll-a and phaeopigments, again in 90% acetone but using a volume of 30 ml (Rodriguez 1993). In all cases, pigment concentrations were eventually measured fluorometrically (Turner Designs 10-AU model), using the non-acidification narrow-band technique (Welshmeyer 1994). This system allows for precise measurements (max 10% error) of chl *a* without interference from other photosynthetic pigments or their degradation products (Welschmeyer, 1994). To obtain chl-*a* and phaeopigment concentrations in the sediment, the fluorescence readings were converted using the following equation:

$$[\text{mg Chl-}a\ \text{m}^{-2}] = \frac{\text{fluorescence reading}}{1000} \times \frac{\text{Volume acetone (ml)}}{\text{Core area (m}^2)} \times \text{dilution}$$

For the calculation of the same in the water-column (phytoplankton), the following was applied:

$$[\text{mg Chl-}a\ \text{m}^{-3}] = \text{fluorescence reading} \times \frac{\text{Volume acetone (ml)}}{\text{Water filtered (ml)}}$$

The statistical package Statistica was used to analyse spatial and temporal differences of microalgal chl *a* concentrations in the two estuaries. Prior to analysis, variables were $\log_{10}$ transformed to comply with the assumptions of the various parametric tests employed.

ZOOPLANKTON

Daytime midwater and suprabenthic samples were taken at monthly intervals, at all three stations and in both estuaries using a WP-2 (90 µm mesh) and a hyperbenthic sled (200 µm mesh), respectively. The sled was pulled over a fixed distance of 50 m. The volume of water filtered by the sled (V) was derived from the equation:

$$V = d\,\frac{(\pi r^2)}{2}$$

where: d = distance traveled by the sled and r = the mouth radius of the sled. Samples collected were fixed in 5% formalin for later laboratory analysis.

In the laboratory, samples were re-suspended in 200-5000 ml of solution, depending on their concentrations. Three 20 ml sub-samples were then drawn from each sample after vigorous stirring to prevent settlement (Perissinotto & Wooldridge 1989, Kibirige 2002). In the subsamples, zooplankton taxa were identified and enumerated under a dissecting microscope. Counts were standardized to number of individuals per cubic meter (ind.m^{-3}). Biomass was estimated by measuring the dry weight of half of each zooplankton samples, after removal of detrital particles under a dissecting microscope. Dry weight was obtained after oven-drying samples at 60 °C for a standard period of 24 hours (Kibirige & Perissinotto 2003).

In order to equalize variance and normalize distribution, all data used in the analysis of variance (ANOVA) were $Log_{10}(x+1)$ transformed. A 1-way ANOVA was run to test for spatial and temporal differences in zooplankton abundance and biomass at each estuary. A 2-way ANOVA was also applied to test for differences between the estuaries and between the closed and open phases of each estuary. A Spearman rank correlation analysis was performed between rainfall and estuarine physico-chemical parameters and zooplankton abundance/biomass data.

MACROBENTHOS

These and the following two groups of organisms were collected/monitored every three months because of their relatively slow turnover time. Macrobenthic samples were collected in March, June, September and December 2002 and again in March 2003. Samples were taken using a Zabalocki-type Ekman grab, which sampled an area of 15 cm^2 to a depth of 10-15 cm, depending on sediment type. Estimates of numerical abundance and community structure were again obtained through dissecting microscope analysis. Triplicate grabs taken *ca.* one metre apart were combined into one sample and three such samples were collected in each of the three regions described above. Each sample was suspended in water in a bucket by stirring and the supernatant liquid decanted through a 500 μm mesh. This was done five times. Any remaining sediment was sieved through a 2 mm diameter mesh to check for heavier organisms such as bivalves and gastropods. All organisms collected were preserved in a 10% formalin-phloxine mix, which fixed and stained the animal material. Identification and counting were done in the laboratory. Identifications followed Day (1967a, 1967b, 1969).

FISH

As with the macrobenthos above, the ichthyofauna of each estuary was sampled at quarterly intervals, starting in March 2002 and ending in March 2003. Samples were collected using a combination of seine netting and gill nets. The use of two sampling gears was adopted to ensure that representatives of all species and size classes were adequately sampled. Seine netting was conducted during daylight hours, using a 30 m x 1.7 m x 15 mm bar mesh seine net fitted with a 5 mm bar mesh purse. Seine netting was limited to shallow (<1.5 m deep), unobstructed areas with gently sloping banks. Where possible, specimens collected by seine netting were identified, counted and measured (mm SL) in the field, before being released again. Those specimens that could not be identified in the field were preserved in formaldehyde for later processing in the laboratory. Gill netting was generally carried out in deep (>1 m) open, mid-channel waters with the nets being deployed in the evening (18h00-19h00) and lifted the following morning (06h00-07h00). Each gill net comprised three 45 mm, 75 mm and 100 mm stretch mesh monofilament panels and was 10 m long and 1.7 m deep.

All fresh specimens were identified in the field, measured to the nearest mm (SL) using a measuring board and weighed to the nearest gram, using a Bonso kitchen scale. Fish brought back to the laboratory were identified by reference to Smith & Heemstra (1986) and measured as above using a measuring board and weighed to the nearest 0.1 g (wet mass) using a Mettler PJ 3000 balance.

The total species composition, both by number and by mass, of the fish communities within each system was calculated. The relative mass contribution of each species, was calculated using actual recorded masses and masses derived from length – mass relationships presented in Harrison (2001). All abundance and biomass measurements of fish caught either with the seine or the gill nets were converted to catch per unit effort (CPUE), in order to standardise the data.

The recruitment via berm-overtopping study was conducted in August-November 2003. Eight sampling exercises, for the collection of fish larvae/juveniles and invertebrates, were undertaken at the Mhlanga and the Mdloti during the peak of each spring high tide, when chances of substantial wave overtopping were highest. In total, six collections were carried out at the Mhlanga (August, September, October), while three were undertaken at the Mdloti (November). Note that initially both nets were used at the Mhlanga estuary. When this proved fruitless, one net was used at the Illovo estuary, and the other was deployed on the same day at the Mhlanga. When the Illovo estuary breached and remained open, we decided to move to the adjacent Msimbazi estuary, and when the Mhlanga consistently failed to yield larvae, the last two sample occasions were comparing the Mdloti and Msimbazi estuaries.

In all the overtopping related sampling, two fry nets were used. These were made of a 5m x 1.3m length of 50% shade-cloth, with a pole sewn into each end, and lead weights sewn into the bottom edge. A few floats were added to the top edge, but proved unnecessary, although the header rope, to which they were attached, was critical. Collections were done by positioning the nets upright at the mouth of the estuary, and allowing overtopping waves to pass through the nets. After several waves had passed, the nets were carefully lifted to a horizontal position, and inspected for any fish/invertebrate retained by the mesh. On all occasions, the procedure was repeated for a period of

1-2 hours. All specimens collected were fixed in 5% formalin and returned to the laboratory for identification and measurement.

BIRDS

Species composition and relative numerical abundance of the bird communities of the two estuaries were recorded in March, June, September and December 2002 as well as in March 2003. This study involved only direct visual observations throughout each estuary during the day of the general survey (linked to all the other physical, chemical and biological parameters measured above). Although the full area of the estuarine basin was investigated on each occasion, the focus was on the intertidal/exposed part of the habitat, as this is the region where the largest changes are expected to occur during the open phase.

RESULTS & DISCUSSION

1. HYDRODYNAMICS: FLOW, RESIDENCE TIMES, WATER LEVELS, & MOUTH DYNAMICS

1.1 Background

It is well known that hydrological and hydrodynamic factors play a key role in sustaining the health and function of estuarine systems. In South Africa about 70% of estuaries do not have a permanently open link to the sea – i.e. they are temporally open/closed estuaries (TOCEs). The open/closed state of the mouth governs the water exchange between sea and estuary (and hence the salinity gradients) as well as biological exchanges between the two systems. The influence of flow on the open/closed state of an estuary is thus a critical factor in determining reserve estuarine water requirements (as regulated by the SA National Water Act of 1998), where the aim is to maintain their ecological functioning while also meeting other demands on the water resources. The hydrodynamics component of this project therefore focussed mainly on the relationship between flow and the open/closed state of TOCE systems.

1.2 Objectives

The terms of reference of this project required that the hydrodynamics component:

> *"Provide data to predict the change in the frequency, timing and duration of mouth closure for different river flow scenarios. For example, reduced annual river inflow may increase the frequency and duration of mouth closure in a temporarily open/closed estuary or it may even cause the mouth of a permanently open estuary to close, changing it into a temporarily open/closed system."*

The objectives of the hydrodynamic investigation were therefore to:

1. *monitor flow rates, water levels and mouth state*
2. *estimate residence times*
3. *understand the relationship between flows, residence times, and mouth state*

1.3 Conceptual framework

The hydrological functioning of a TOC estuary system can be conceptualised in terms of a dynamic storage system with variable inputs and outputs. Freshwater inputs are dominated by the catchment's runoff (with relatively smaller direct rainfall input). Groundwater flow directly into the estuaries was not considered as part of this study but could also be significant. Outputs comprise losses due to evaporation/transpiration and seepage outflows through the sand berm separating the estuary from the sea when the inlet is closed. When the mouth is open, tidally driven exchange flows can occur. The magnitudes of the tidal prisms at the case study estuaries were unknown at the start of the project, but were expected to be small due to the perched nature of these estuaries.

A closed estuary can naturally develop an opening to the sea when water levels rise to cause breaching of the frontal berm that separates it from the sea. The maintenance of an open state depends on a balance between sediment removal by scouring and sediment deposition by wave action. At the case study sites, longshore sediment transport is estimated to exceed 1000 m^3/day (based on data from the sand trap at Durban harbour – see e.g. Schoonees, 2000). Given the small size of the estuaries, this provides a plentiful supply of sediment that, in combination with a relatively energetic wave climate and associated rapid cross-shore sediment redistribution can drive the re-closing of a small inlet over relatively short time scales (Huizinga, 2003, personal communication).

Storage "residence times" that characterise the water balance of an estuary may be defined as

$$T_S = \frac{S}{Q_{IN} - Q_{OUT}}$$

where, Q_{IN} and Q_{OUT} are lumped inflows and outflows to/from the system, and S is some reference storage e.g. the storage when the closed estuary is at maximum capacity just prior to breaching of the frontal berm. Defined in this way, T_S indicates the time scale for the estuary to fill to capacity under specific inflow/outflow conditions. Note that when inflows dominate outflows i.e. $Q_{IN} >> Q_{OUT}$, then $T_S \cong S/Q_{IN}$. Also when $Q_{IN} = Q_{OUT}$, storage remains constant (i.e. T_S is infinite).

Flow "residence times", defined simply as $T_Q = S/Q$, represent the time scales required for specific flows to replace the storage volume, and are a convenient way to compare flows (in terms of their role in the water balance) at estuaries of different sizes or storage volumes.

Residence times are key indicators of the hydrodynamic functioning of an estuary. For TOCEs, inflows with residence times that are short compared to the time scales for mouth closure due to cross-shore sediment redistribution, should be able to maintain an open inlet. Alternatively, inflows with much longer residence times would not be able to maintain an open mouth and may be expected to be associated with closed mouth conditions.

Resource Directed Measures (RDM) for estimating the reserve flow requirements of estuaries are currently based on an approach that specifies average (usually monthly) flow thresholds to define the different abiotic states of the systems. These include a threshold flow below which the mouth is closed, a threshold flow above which the mouth is open, and possibly intermediate values that define partially open conditions (see e.g. CSIR, 2002, 2003). This approach, based solely on flow rates, has the disadvantage that it cannot be directly linked to water levels since they are determined by the full water balance, and not just the inflows. It is only in situations where the water balance is dominated by inflows (or the lack thereof) such as in flood flows (or very low flows), that this approach can be expected to yield unambiguous results. This point will be clarified later when the results of this study are discussed.

1.4 Physical characteristics of the case study estuaries

1.4.1 Inflow hydrology and estimated changes from reference state

A summary of key hydrological data for the two case study estuaries is given in Table 1.1.

Table 1.1 Key hydrological data for the case study estuaries

Estuary name	Catchment Area (km^2)	MAR $10^6 m^3$ (m^3/s)	MAR/MAP	ΔFLOW	Approx % closed reference/present
Mdloti	484	83.2 (2.64)	17 %	- 20%	10% / 50%
Mhlanga	80	12.6 (0.4)	16 %	+ 57%	80% / 50%

In some cases the values in Table 1.1 differ significantly from those appearing in publications such as Begg (1984). The MAR figures are for simulated natural conditions, as given in WR90 (i.e. Midgley *et al*, 1994) for quaternary catchments U30A & U30B, but scaled to the appropriate catchment areas of each estuary. The U30A catchment contributes solely to the Mdloti river, while U30B contributes to both Mdloti and Mhlanga. The total area of U30A and U30B combined is given by WR90 as 597km^2, which includes a 35km^2 south-extending coastal section that does not contribute runoff to the estuaries. The combined natural MAR from U30A & B is given in WR90 as 101 Mm^3, which corrected for the non-contributing area gives a total for the two estuaries of approximately 96 Mm^3 (Mhlanga~13 Mm^3, Mdloti~83Mm^3). The net overall changes to the inflows (from natural reference conditions) are indicated separately in Table 1.1 as ΔFLOW. These estimated changes are due to the Hazelmere Dam (constructed in 1976/7) with associated abstractions and losses, discharges from wastewater treatment works, or run-of-river abstractions. Hazelmere dam, situated approximately 20km upstream of the Mdloti estuary at the outlet from U30A, has a storage capacity of 22Mm^3 (or about 30% of the MAR for U30A). A summary of the inflow changes from the reference (or natural) state to the present state is given in Table 1.2. Discharges from WWTW facilities were supplied by the operators and reflect average influent measurements. Influent flows have large diurnal variations: for example at Mhlanga WWTW the average influent flow is estimated as about 6Ml/day, but actual flows vary in the range 1 – 20 Ml/day depending on the time of day. Retention ponds that are provided before the effluent is discharged into the river smooth these short-term fluctuations.

Table 1.2 Summary of flow changes at the case study estuaries

FLOW CHANGE	MHLANGA	MDLOTI	Comments
Abstractions Mm^3/yr	-	13.2 0.7 5.1	Hazelmere Dam (WR90) Huletts (WR90) Umgeni Water (WR90)
Discharges: Mm^3/yr Mhlanga WWTW Phoenix WWTW Mhloti WWTW Verulam WWTW	 2.4 4.8 	 2.7 	Average values
NETT TOTALS Mm^3/yr	+ 7.2	- 16.3	Change from natural
% CHANGE	**+ 57%**	**- 20%**	Relative to natural

Streamflow data since 1978 from gauge U3H005 at Hazelmere dam have been analysed and give average annual flows that are within 10% of the natural MAR as given in WR90 and therefore do not reflect the expected abstractions noted in Table 1.2. For example the average annual flow between 1990-2001 was 60.4 Mm^3 which is 92% of the expected natural MAR from U30A.

Further, it is noted that CSIR (2002), in a report on a recent RDM workshop concerning Mdloti estuary, give the natural reference state MAR as 98.7 Mm^3 and the present-day flows as 72.3 Mm^3. The precise basis for those particular estimates is not clear, but they are significantly higher than the values given in Table 1.1.

To summarize, it is apparent that the estimates of the reference state MAR are subject to considerable uncertainty, and that values given in Tables 1.1 and 1.2 should therefore be treated with some caution.

1.4.2 Physical characteristics of the estuaries

A map showing the two estuaries and their catchments is given in Figure 1.1. The positions of the Hazelmere Dam and four waste-water treatment works (WWTW) are shown on the map. The spatial extents of the two estuaries as depicted in Figure 1.1(b) and (c) were obtained by locating the approximate position of the contour corresponding to the maximum observed water levels. Note the surface areas thus obtained reflect the maximum area impounded under closed mouth conditions, and are much larger than those previously reported (e.g. by Begg (1978) or Jezewski *et al* (1984)) and which seem to reflect only the area of the lagoon adjacent to the sandbar.

Storage capacities for the estuaries were estimated from the surface area and the maximum observed water level variations as $S = C_v \cdot Area \cdot \Delta H_{max}$, where C_v is an order unity coefficient (expected to be in the range $^1/_3 \leq C_v \leq 1$) that depends on the morphology of the volume. A value of $C_v = ½$ has been assumed for the present purposes since a detailed hydrographic survey has not been carried out.

The estuaries are located on an exposed coastline where the median significant wave height is approximately 1.8m and the dominant wave direction is from the South (Rossouw, 1984). Beach sediments are relatively coarse at this location ($d_{50} \sim 1$mm) and beach slopes are steep (typically 10% - 20%). The height of the berm is about 3m* above MSL. The permeability of the beach sediment was measured in laboratory tests as 1 to 7 mm/s, depending on compaction density. Longshore sediment transport in this region is estimated to be about 1400m^3/day in a northerly direction (e.g. Schoonees, 2000).

A compilation of physical data for the Mhlanga and Mdloti estuaries is given in Table 1.3.

Table 1.3 : Physical data for Mhlanga & Mdloti estuaries

Parameter	units	MHLANGA	MDLOTI	Source
Catchment area	km^2	80	484	WR90
Average slope	%	0.6	0.7	Jezewski *et al* (1984)
Length longest collector	km	25	75	Jezewski *et al* (1984)
Time concentration	hrs	6	12	Jezewski *et al* (1984)
MAP	mm	1000	1000	WR90
MAE	mm	1210	1210	WR90
Natural MAR	m^3/s	0.40	2.64	WR90
Present average inflows	m^3/s	0.63	2.06	present study
Estuary Area	ha	70	80	present study
% Closed	%	50%	50%	present study
Max water level (+MSL)	m	± 3200	± 3200	present study
Min water level (+MSL)	m	670	780	present study
Berm length	m	500	800	present study
Berm width	m	30 – 60	± 70	present study
Berm height (+MSL)	m	3 – 4	3 – 4	present study
Sandbar permeability	m/day	100 – 200	100 – 200	present study
Tidal amplitude (spring)	m	2.0	2.0	
Tidal amplitude (neap)	m	0.5	0.5	
Median significant wave ht	m	1.8	1.8	Rousouw (1984)
Typ beach slope	%	10% - 20%	10% - 20%	present study
Avg Long-shore transport	m^3/day	1400	1400	Schoonees (2000)
Avg cross-shore transport	m^3/day/m	15 – 20	15 – 20	present study
Live Storage in estuary	m^3	800 000	950 000	present study
Avg inflow residence time	days	15	5	present study
Breaching ht (+MSL)	mm	3000 - 3400	3000 - 3400	present study
Tidal prism % vol	%	5 - 30	?	present study
Mouth closing time scale	days	3 – 6	?	present study
Max Seepage outflows	m^3/s	0.25	0.50	present study

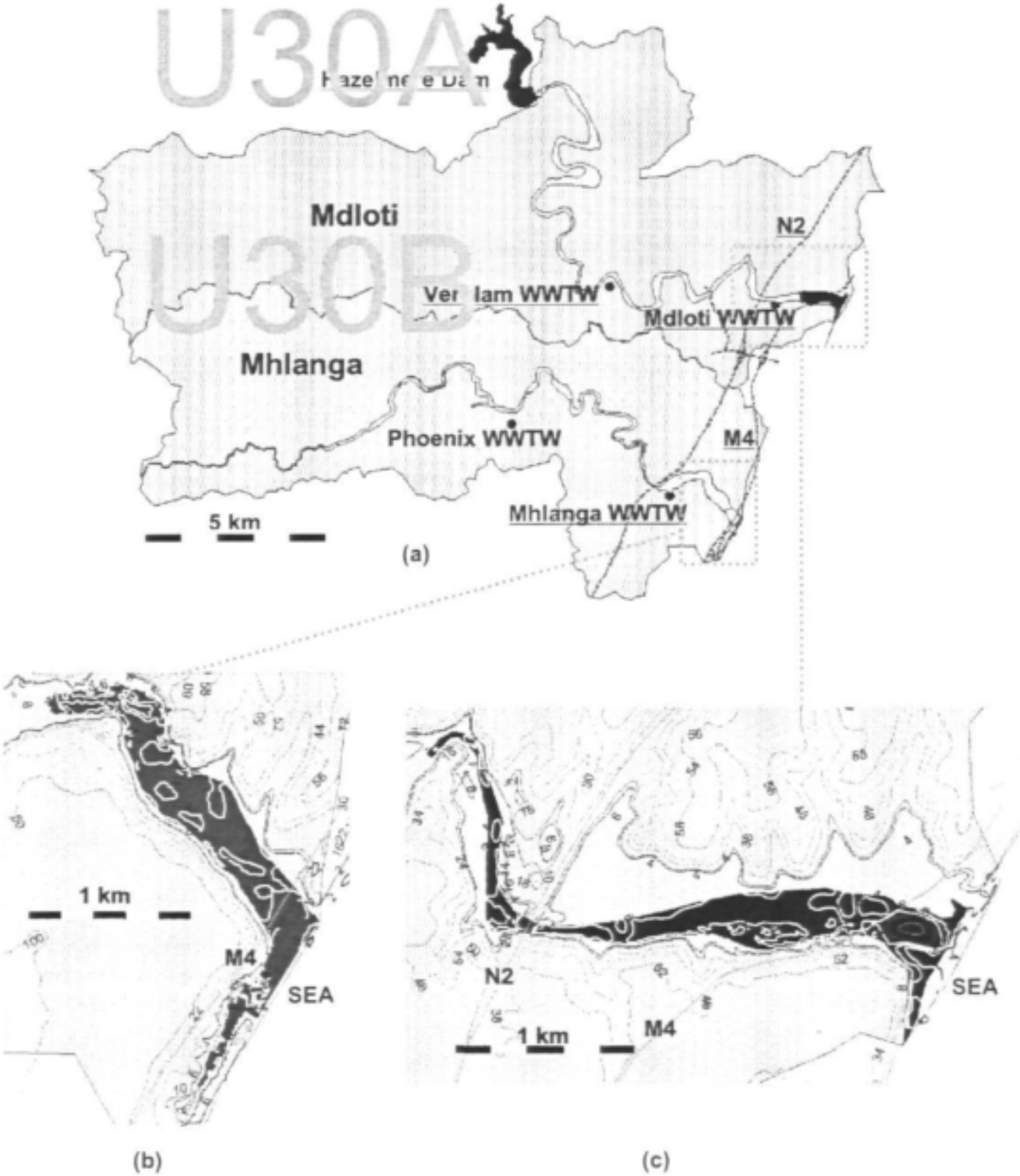


Figure 1.1 Maps showing the estuary catchments (a) and details of the maximum impounded water surface areas for (b) Mhlanga and (c) Mdloti respectively. Note that part of the Mdloti catchment upstream of the Hazelmere dam (quaternary U30A) is not shown. The scales shown are approximate indications only.

1.6 Results

This section starts with a general overview of the monitoring results, followed by detailed discussion of various factors concerning the flow and mouth dynamics. Further details and discussion of the results are reported in Zietsman (2003).

1.6.1 Overview of the monitoring results

Monitoring results for the two estuaries are shown in figures 1.2 – 1.5. The data presented includes daily rainfall, mouth state observations, water levels (relative to MSL), and measured (instantaneous) flow rates. Some general patterns discernible in the data may be summarized as follows:

(a) During the period March 2002 to March 2003 total rainfall was 69% of average and had an unusual seasonal distribution with high precipitation recorded during the winter. The rainfall in July accounted for 23% of the total rainfall for the period. From March 2003 to November 2003 rainfall was even lower at 55% of average, but had a typical seasonal distribution..

(b) Mouth state during the period Mar 2002 to Mar 2003 reflects the rainfall and also did not follow expected seasonal trends, with open conditions prevalent during the winter months. Mhlanga mouth was much more active than Mdloti with about double the number of breachings, although the overall mouth state statistics were similar. Mhlanga was closed 51% and open or partly open for 49% of the time (excluding missing data), while Mdloti was closed 58% of the time and open or partly open 42% . From March to December 2003 Mdloti remained continuously closed. In contrast Mhlanga breached at regular (quasi-periodic) intervals, but remained open for only short periods each time. Overall Mhlanga was closed for 80% of the time during the 2003 period.

(c) Flows also had an usual pattern during 2002 that reflects the rainfall distribution. After the summer rainfall at the start of 2002, flows gradually reduced during the autumn as expected. The rainfall during July reversed this trend and caused a sudden increase in flows during the winter months. These gradually reduced in spring before increasing again with the summer rains. By late February 2003 flows reduced again and remained low for the rest of the year due to the much lower than average rainfall.

(d) Water levels during the period March 2002 to March 2003 fluctuated erratically (particularly at Mhlanga) reflecting the frequent breaching events. Maximum recorded water levels (just prior to breaching) were 3000 – 3300mm above MSL with Mhlanga slightly lower than Mdloti. Minimum recorded water levels were 660mm and 790mm above MSL for Mhlanga and Mdloti respectively. The monthly photometric recordings of water levels did not have sufficient temporal resolution to provide details of breaching behaviour and tidal influence. Continuous water level measurements from March 2003 provided a much more detailed characterisation of mouth dynamics and the role of tidal exchange flows (see sections 1.6.5 – 1.6.7).

More detailed discussion of the hydrodynamic aspects is given in the sections following.

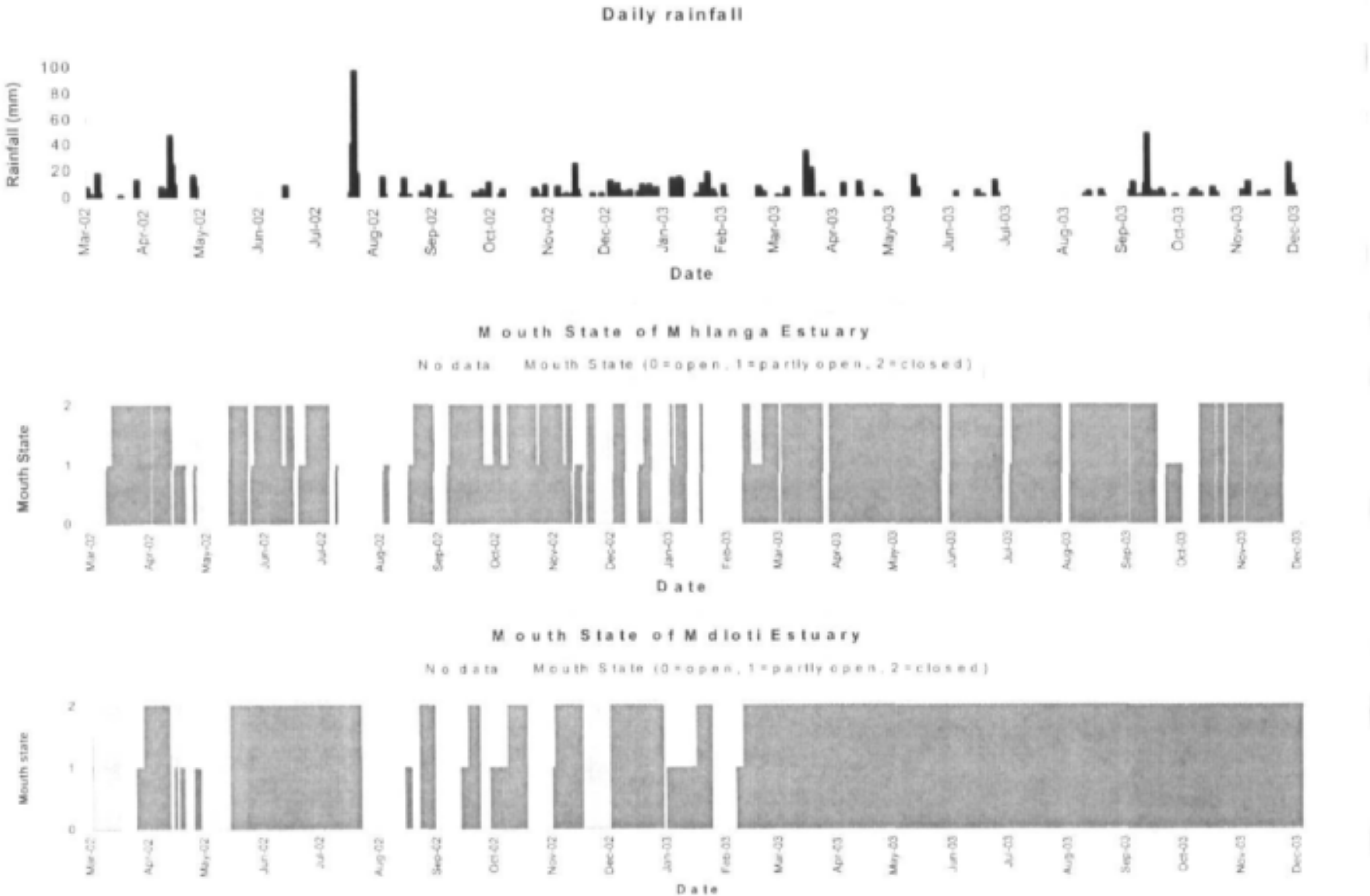
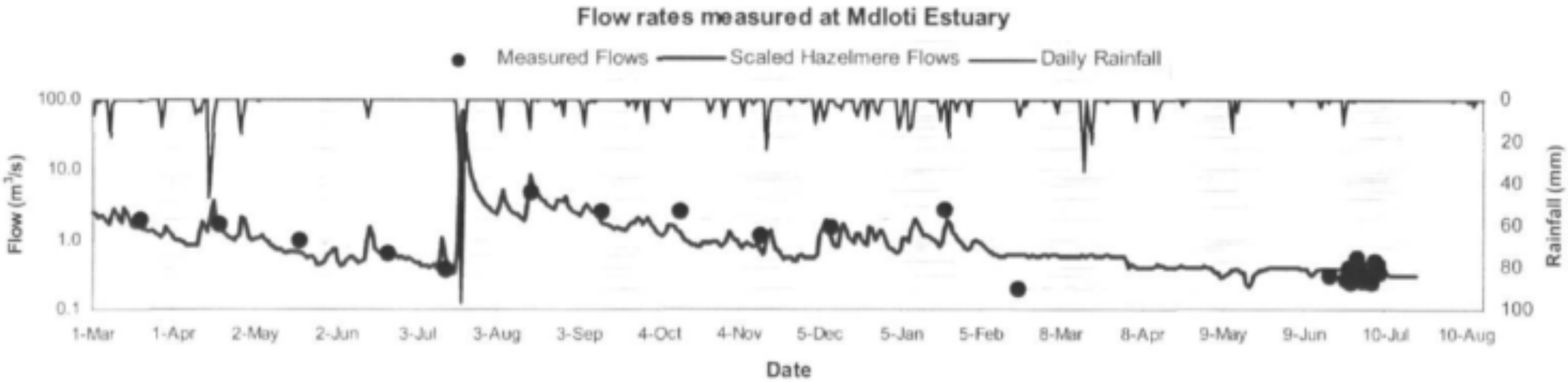


Figure 1.2 Daily Rainfall and mouth state for Mhlanga and Mdloti estuaries



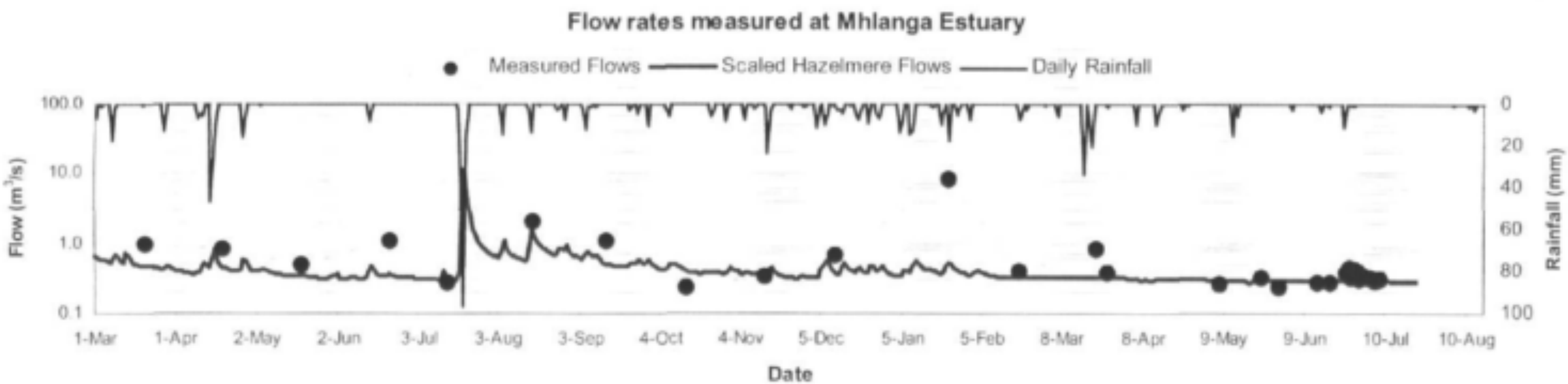
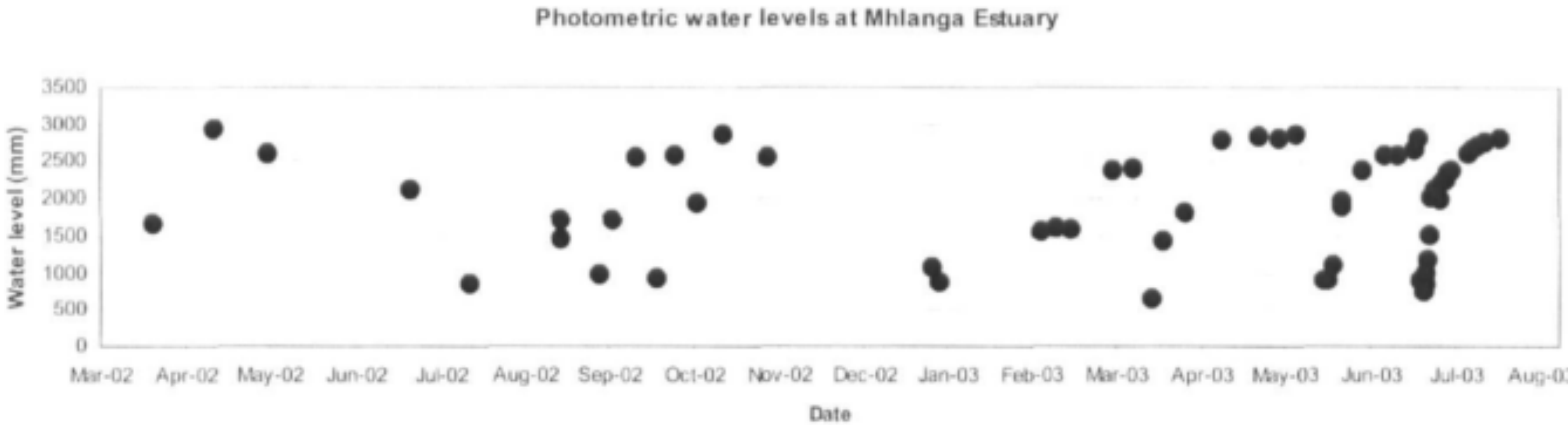


Figure 1.3 Measured flows at Mhlanga and Mdloti estuaries. Also shown are flow measurements from stream-gauge U3H005 located at Hazelmere dam. The stream-gauge data has been scaled by catchment areas to account for ungauged portions, and estimated WWTW discharges have been added.



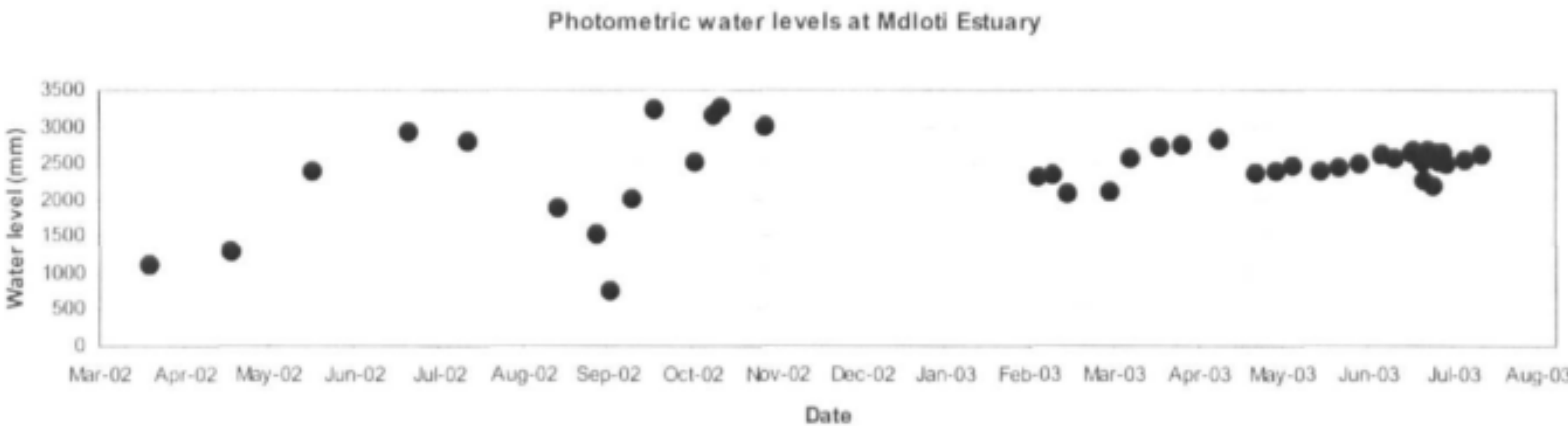


Figure 1.4 Water levels (relative to MSL) at Mhlanga and Mdloti measured using digital photographs

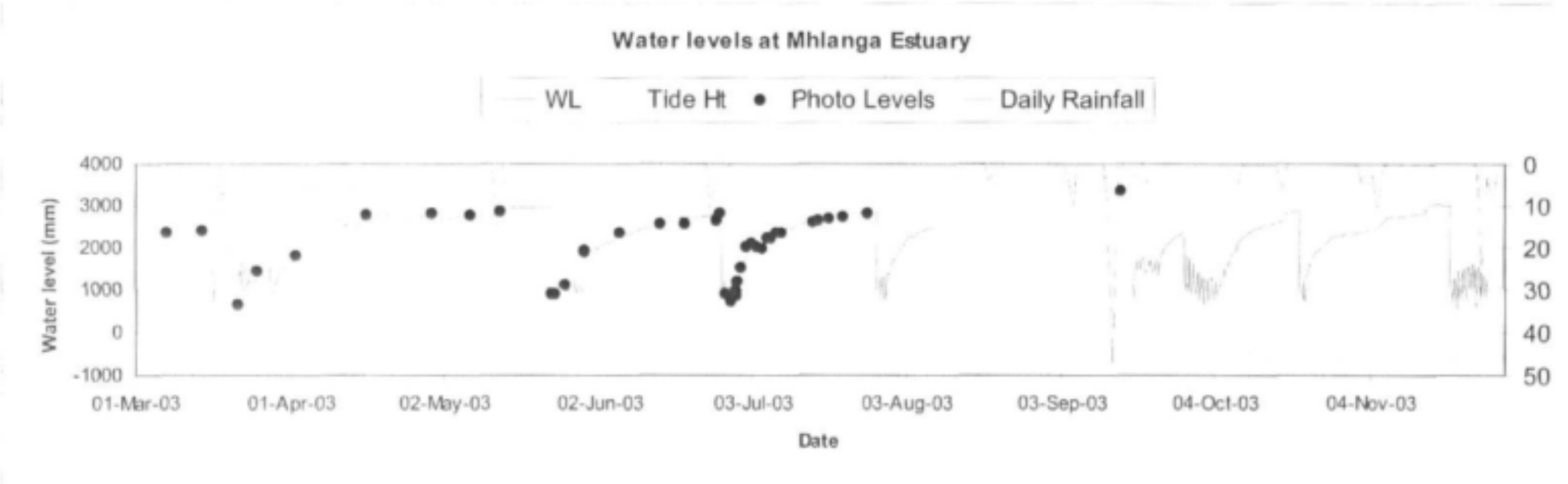


Figure 1.5 Water levels (relative to MSL), tide heights, and daily rainfall for Mhlanga estuary during the period March 2003 to December 2003. The water levels were logged hourly from a continuous water level sensor - photometric water levels are also shown for comparison.

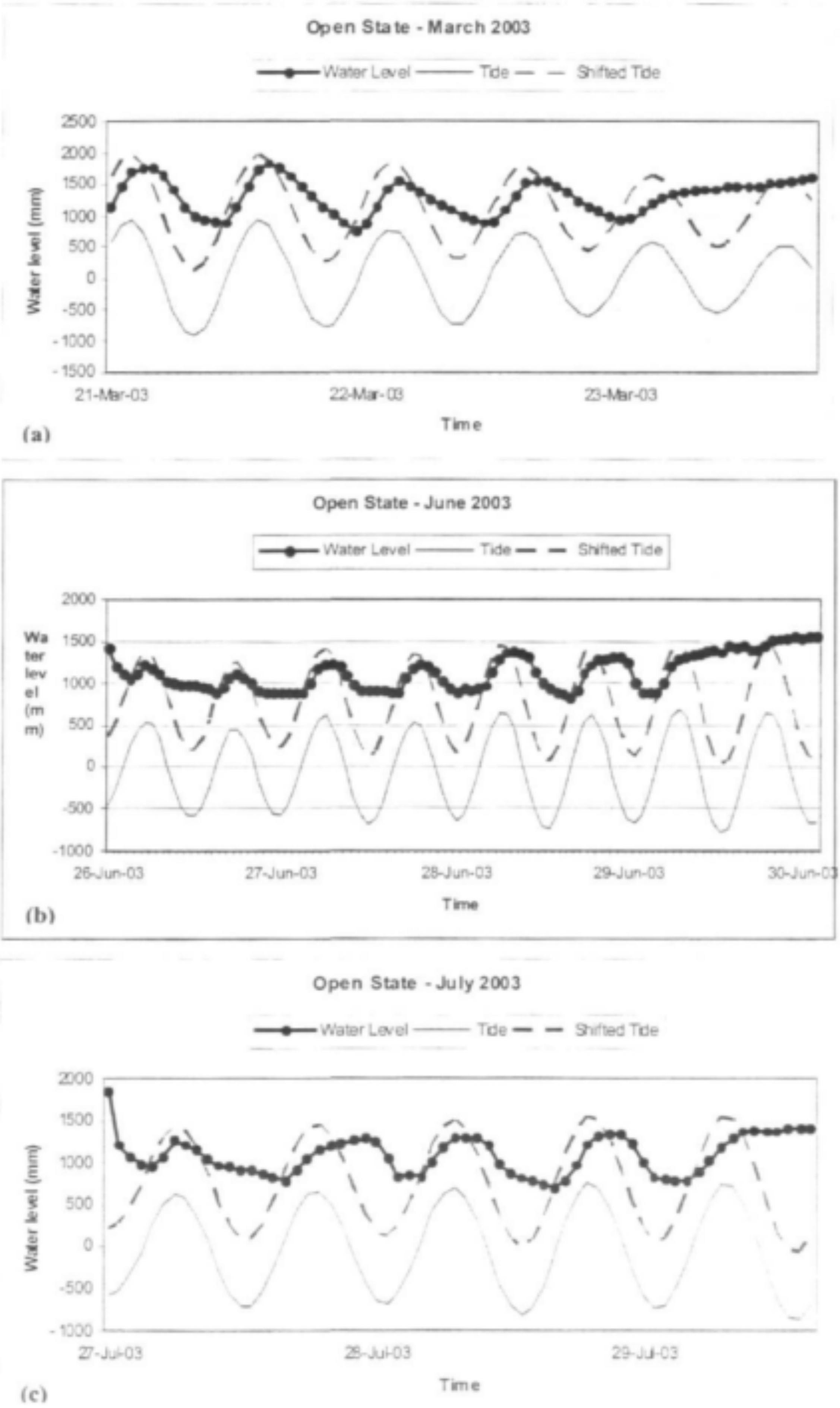


Figure 1.6 Examples of water levels and tide heights at Mhlanga when in an open state. Dashed lines are upward projections of tide heights to represent the effects of wave run-up.

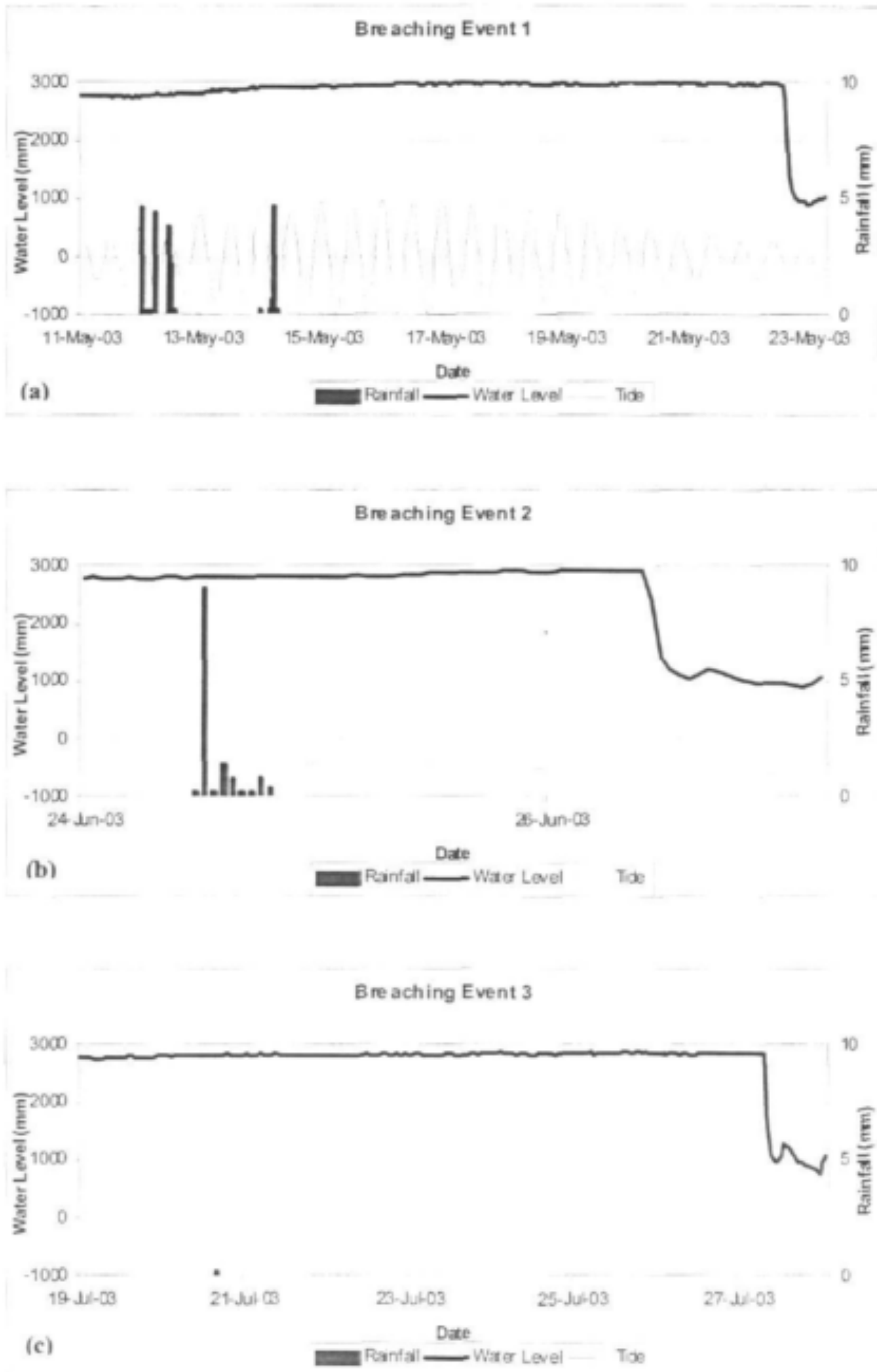


Figure 1.7 Three examples of breaching events at Mhlanga estuary, showing water levels, tide heights and hourly rainfall.

1.6.2 Rainfall

General patterns

Accumulated rainfall during the period March 2002 to March 2003 was 69% of average for that period. The daily rainfall data are shown in Figure 1.2. The unusual seasonal distribution of rainfall during 2002 is clearly evident, as is the low rainfall during 2003. About 160mm of rain fell in July 2002, representing the highest monthly record for the year and accounting for 23% of the total rainfall of the period (692mm). The second highest monthly rainfall occurred in April 2002 (124mm) and in January 2003 (100mm). The lowest figures were recorded in May (<1mm) and June 2002 (9mm), as well as in February 2003 (16mm).

Accumulated rainfall during the period March 2003 to November 2003 was 55% of average and had a typical seasonal distribution with very little rainfall during the winter months. No rainfall was recorded during July 2003. September 2003 was the only month in 2003 to receive above-average rainfall.

Recorded monthly rainfall totals are compared with average values in Figure 1.8 where the rainfall deficit during the case study period is clearly evident.

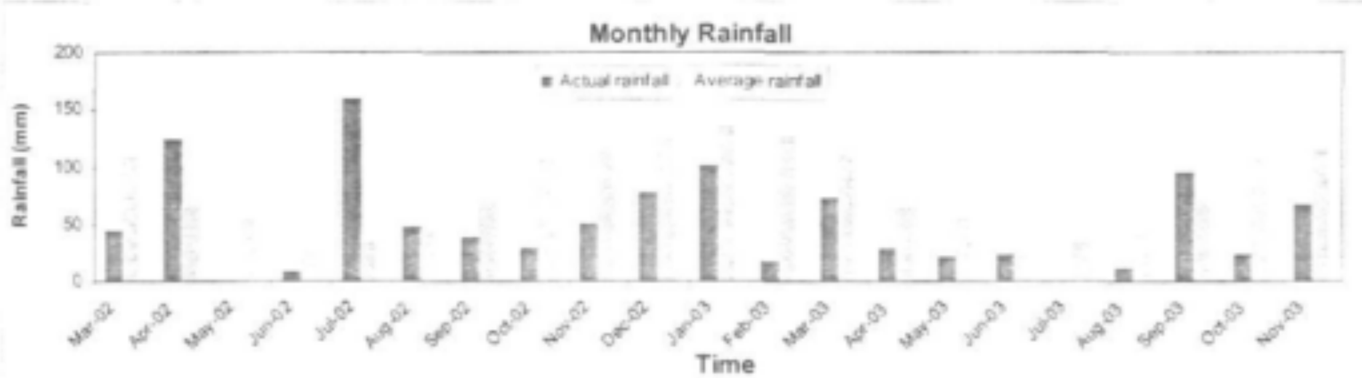


Figure 1.8 Monthly rainfall totals compared with average values

Relationship between rainfall and mouth dynamics

The relationship between rainfall events and mouth breaching is complex. High rainfall events with daily totals exceeding 25mm were usually associated with breaching events at Mhlanga. Significantly higher rainfall appears to be required to trigger breaching events at Mdloti (perhaps with totals of order 100mm), but there is insufficient data to establish any threshold. For lower daily rainfall totals, the situation is more complex and a direct and immediate link between daily rainfall and mouth state is not apparent.

In an average sense, as expected, the data shows a general trend with increased rainfall correlated with increased open mouth conditions. For example a Spearman's rank correlation analysis between monthly rainfall totals and the proportion of closed mouth conditions was performed for the period March 2002 to July 2003. For Mdloti the correlation was –0.4 (significant at the 85 percent confidence level), while at Mhlanga the correlation was –0.6 (significant at the 95% confidence level). Furthermore, the overall reduction in rainfall during 2003 clearly changed the breaching patterns : Mdloti remained closed continuously after February 2003, and while Mhlanga continued to breach at regularly intervals, it remained open only for short periods so that the overall proportion of time it was closed increased (see section 1.6.7).

1.6.3 Flows

Flow measurements made during the case study are shown in Figure 1.3. Also shown are rainfall data, and scaled daily flows from the U3H005 stream-gauge located at Hazelmere dam[2]. The U3H005 stream-gauge monitors the outflow from the dam (i.e. catchment U30A) but for comparison with the estuary inflows the ungauged part of the estuary's catchment ($108km^2$ out of a total of $484km^2$) should be accounted for. This was done by simply scaling the U3H005 data in proportion to the increased catchment area. Area scaling was also used to provide a rough comparison for the Mhlanga estuary, although in that case there is no common catchment area, although the catchments are adjacent to one another.

It can be seen that the scaled U3H005 data agrees very well with the instantaneous flow measurements made at Mdloti. As expected, the agreement for Mhlanga is not as good – the much smaller size of the Mhlanga catchment means that it typically responds much quicker to rainfall events (i.e. the time of concentration is much shorter at 6 versus 12 hours - see Table 1.3)

Flow patterns

The general pattern of the flows during the monitoring period reflects the unusual rainfall distribution. From March 2002 to mid July 2002 there was a gradual reduction in the flows. The large unseasonal rainfall event on the 20^{th} July dramatically increased the flows, although the flood peak itself was not captured by the discrete monthly measurements. The higher flows were sustained for 2 months or more although gradually reducing towards the end of the year. Flows at Mhlanga were more variable, perhaps reflecting the smaller catchment. The summer rains in December – January, although lower than average, again increased the flows. The high

[2] The streamflow data was kindly supplied by DWAF

flow ($8.35m^3/s$) measured at Mhlanga on 23rd January, was made within hours of an overnight rain-storm, and was probably very short-lived. The very low rainfall in February led to rapid decrease in the flows. Apart from a notable rainfall event in mid-March following which a slightly increased flow was measured at Mhlanga, all flow measurements made after February 2003 were low at both estuaries.

Note that flow measurements were not made for Mdloti between March and June due problems gaining access to the head of the estuary.

Relationship between flow and mouth state

Combining the data shown in figures 1.2 and 1.3 it can be seen that a relationship between the instantaneous flow measurements and the mouth state is difficult to discern. The flow data is compiled in Table 1.4 together with mouth state information and flow residence times S/Q.

The highest recorded flows, $8.35m^3/s$ and $4.77m^3/s$ at Mhlanga and Mdloti respectively, were both associated with open conditions. At Mhlanga, in particular, the situation can appear confusing since open mouth conditions also occurred for flows that were near the minimum values recorded ($0.23m^3/s$). This is explained later, but it is apparent that an unambiguous classification of the mouth state based on instantaneous flow magnitudes is not possible, particularly at Mhlanga. Mdloti remained closed from February 2003 onwards and the flows were clearly low during this time (although actual measurements were not made from March to June).

Residence times are a useful way to "normalize" flow magnitudes for estuaries of different sizes so that they may be compared and related to mouth dynamics.

It can be seen that flows with residence times shorter than about 5 days were usually associated with open conditions. Conversely, flows with residence times longer than about a month were mostly associated with closed mouth conditions although intermittent breaching still occurred at Mhlanga. An explanation is provided in section 1.7 where the relationship between flows and mouth state is revisited in more detail. .

Table 1.4 Measured flows, residence times, and mouth state for Mhlanga and Mdloti. The notation. "Open 2/6" indicates that it was the 2nd day out of 6 days spent in the open state, etc

Month	MDLOTI			MHLANGA		
	Flow m^3/s	$T_Q = S/Q$ days	Mouth State	Flow m^3/s	$T_Q = S/Q$ days	Mouth State
19-Mar-02	2.00	5	Open 6/10 days	1.00	9	Closed 9/31 days
18-Apr-02	1.68	7	Open 1/6 day	0.86	11	Open 2/6 days
19-May-02	1.00	11	Closed 7/19 days*	0.50	19	Closed 8/9 days
22-Jun-02	0.66	17	Closed 21/49 days	1.08	9	Closed 2/12 days
14-Jul-02	0.39	28	Closed 43/49 days	0.27	34	Open 12/43 days
16-Aug-02	4.77	2	Open 1/5 day	2.03	5	Partly open 2/2 days
12-Sep-02	2.54	4	Partly open 1/4 day	1.09	9	Closed 3/9 days
12-Oct-02	2.50	4	Closed 6/9 days	0.25	40	Closed 7/15 days
12-Nov-02	1.14	10	Closed 12/14 days*	0.34	27	Partly open 3/6 days
9-Dec-02	1.53	7	Closed 10/28*	0.68	14	Open 4/8 days
22-Jan-03	2.64	4	Closed 8/8 days	8.35	1	Open 7/22 days
19-Feb-03	0.20	55	Closed 11 days	0.40	23	Closed 2/9 days
21-Mar-03	-	-	-	0.86	11	Closed 21/21 days
25-Mar-03	-	-	-	0.37	25	Open 4/4 days
7-May-03	-	-	-	0.25	36	Closed 40/56 days
23-May-03	-	-	-	0.33	28	Closed 56/56 days
30-May-03	-	-	-	0.24	39	Closed 2/28 days
14-Jun-03	-	-	-	0.28	33	Closed 17/28 days
19-Jun-03	0.30	37	Closed 131 days	0.28	33	Closed 22/28 days
25-Jun-03	0.27	41	Closed 137 days	0.38	24	Closed 28/28 days
26-Jun-03	0.40	27	Closed 138 days	0.45	21	Open 1/4 day
27-Jun-03	0.24	45	Closed 139 days	0.33	28	Open 2/4 days
28-Jun-03	0.32	34	Closed 140 days	0.42	22	Open 3/4 days
29-Jun-03	0.56	20	Closed 141 days	0.40	23	Open 4/4 days
30-Jun-03	0.41	27	Closed 142 days	0.31	30	Partly open 1 day
1-Jul-03	0.26	42	Closed 143 days	0.33	28	Closed 1/26 day
2-Jul-03	0.29	38	Closed 144 days	0.33	28	Closed 2/26 days
3-Jul-03	0.26	42	Closed 145 days	0.32	29	Closed 3/26 days
4-Jul-03	0.30	36	Closed 146 days	0.32	29	Closed 4/26 days
5-Jul-03	0.25	44	Closed 147 days	0.31	30	Closed 5/26 days
6-Jul-03	0.48	23	Closed 148 days	0.28	33	Closed 6/26 days
7-Jul-03	0.43	25	Closed 149 days	0.30	31	Closed 7/26 days
8-Jul-03	0.34	32	Closed 150 days	0.31	30	Closed 8/26 days

* missing data renders the start and/or end of the mouth state period uncertain

Verification of the flow data

An important issue concerning the inflows to Mhlanga arose from the flow monitoring. The measured flows were generally considerably higher than expected, given the estimated MAR (see Table 1.1) and WWTW discharges (Table 1.2). For example the average of the flows measured between March 2002 and March 2003, after subtracting the estimated WWTW discharges (0.23 m^3/s) was 1.25m^3/s. This correspond to about 300% of the reference state MAR (0.4 m^3/s) obtained from WR90. The measured flows include one high flow (8.35m^3/s) measured just after a heavy rainfall event, and if this measurement is excluded, the mean value drops to

$0.63m^3/s$ or about 150% of estimated MAR. A similar calculation for the Mdloti estuary yields an average measured flow of $1.80m^3/s$ or 68% of the estimated reference state MAR ($2.64m^3/s$). The rainfall that fell on the catchment during that period was 69% of average. Therefore while the Mdloti flows are consistent with what may be expected, the flows at Mhlanga are about a factor of two higher than expected. Two possible reasons for this discrepancy are (1) underestimation of the natural MAR, or (2) underestimation of the WWW discharges.

Flow measurements upstream and downstream of the Mhlanga WWTW were made to check the estimated discharges. The measured increase in flow was 0.085 m^3/s (the equivalent of 7.4 Mlitres/day) which corroborated a discharge estimate of 0.083 m^3/s provided by the operator of the WWTW facility. A similar check of the discharges from the Phoenix WWTW further upstream was not done. However, assuming that they are also consistent with the estimates, it would appear that the natural runoff from the Mhlanga catchment could be up to a factor of two higher than estimated (Table 1.1) but the issue remains to be fully resolved. It should be noted that higher MAR values for Mhlanga have been previously published and are more consistent with the measurements made during this study - e.g. Jezewski *et al* (1984) report an MAR of $23Mm^3$ ($0.73m^3/s$).

The above issue is important because an accurate estimate of the reference state MAR is required in order to establish the magnitude of the changes that have occurred and hence the sensitivity of the system to flow changes. For example, if the reference state flows have indeed been underestimated by a factor of two, then the present WWTW discharges represent an increase of about 30%, and not 57% as indicated in tables 1.1 & 1.2. This could have implications for the outcome of the RDM for this estuary (CSIR, 2002).

1.6.4 Water levels

Photo-metric water level data for the monitoring period are shown plotted in Figure 1.4. The perched nature of the two estuaries is evident. Maximum water levels at the estuaries were typically in the range 3000 – 3400mm above MSL (with Mhlanga at the lower end of the range) and occurred just prior to breaching events. Minimum water levels were 660mm and 780mm above MSL at Mhlanga and Mdloti respectively, and were recorded during open mouth conditions.

While the monthly water level recordings were useful for determining the range of values that occur, the temporal resolution was not sufficient for detailed analysis of mouth dynamics and tidal exchange flows. The development and installation of continuous water level sensors (WLS) added a new dimension to the monitoring program where detailed aspects of the mouth dynamics, and tidal exchange flows could be studied. An 8-month period of continuous (hourly) water level data for Mhlanga is shown in Figure 1.5. Also shown on the plot are rainfall data and tide heights. Tide heights were computed for Durban using a calibrated harmonic model (Hopper, 2003). Photo-metric water levels are also shown since they were measured to corroborate the continuous water level monitor.

In a 1-week period when the estuary was open in late May, the WLS that was installed developed a leak and the data is therefore in error – it can be seen in Figure 1.5 that there is a jump in water level at the end of the open phase. This is simply a result of replacing the faulty WLS. Secondly there was a 1-month period from 8th August to 9th September when a non-functional WLS was inadvertently installed in the estuary and data from this period was lost. In-filling of the missing data (see Figure 1.5) was done using a simple model that mimics the observed behaviour from previous months.

The continuous water level monitor shows a characteristic pattern during the 2003 period. Between breaching events there is an exponential-like rise in water levels. Rainfall events generate local lifts in the levels. After reaching maximum levels of about 3000mm above MSL, breaching took place, typically at or near the neap tide phase. During the open phases, which tended to be short, tide driven fluctuations can be seen and water levels reached minimum values of 600 – 700mm, and maximum values of about 1800mm above MSL.

It can be seen from Figure 1.5 that breaching events occurred quasi-periodically during the 2003 period, typically every 30 – 40 days. This is a result of the low rainfall and nearly constant flows.

The breaching event on 15th September was unusual in several respects. Firstly the water levels prior to breaching reached the highest levels (3400mm) observed since the start of the monitoring in March 2002. This data was obtained from a photo of the M4 bridge since the WLS was not functioning (unknown to us at the time). The breaching event was associated in this case with a significant rainfall event comprising 62mm over 3 days. Water levels during the actual breach, which occurred on the night of the 15th, and for 2 days during the open phase, were lost. The dysfunctional WLS was replaced on 17th September so that the mouth re-closure process was captured. It can be seen that there were several small and erratic water level oscillations during the mouth closure process in the 10-days to 27th September. These are apparently due to repeated partial breaching of the sand bar, and are also present in the water level records following the open phase in March. The estuary breached again (from a much lower water level of 2340mm) at spring tide on the 27th September, after which a 1-week open phase with large tidal influence occurred. This breaching event may have been triggered by wave action. The next breach on 20th October followed a similar pattern to others earlier in the year – it occurred at the neap tide and from a water level of 2856mm (perhaps slightly lower than usual).

1.6.5 Tidal exchange flows during open mouth conditions

A major benefit of the continuous water level monitoring was that it enabled tidal exchange flows at Mhlanga to be observed and analysed. As noted previously, both estuaries are "perched" above MSL and therefore only small (if any) tidal influence was anticipated. However the steep beach slope apparently leads to considerable wave run-up that in turn drives a flow into the estuary when it is open.

The first examples of tidally driven water level fluctuations within the Mhlanga Estuary were captured in March 2003 (immediately after the installation of a continuous water level sensor). The recorded water levels are shown plotted in Figure 1.9 on the same scale as the corresponding tide (see also figures 1.4, 1.5).

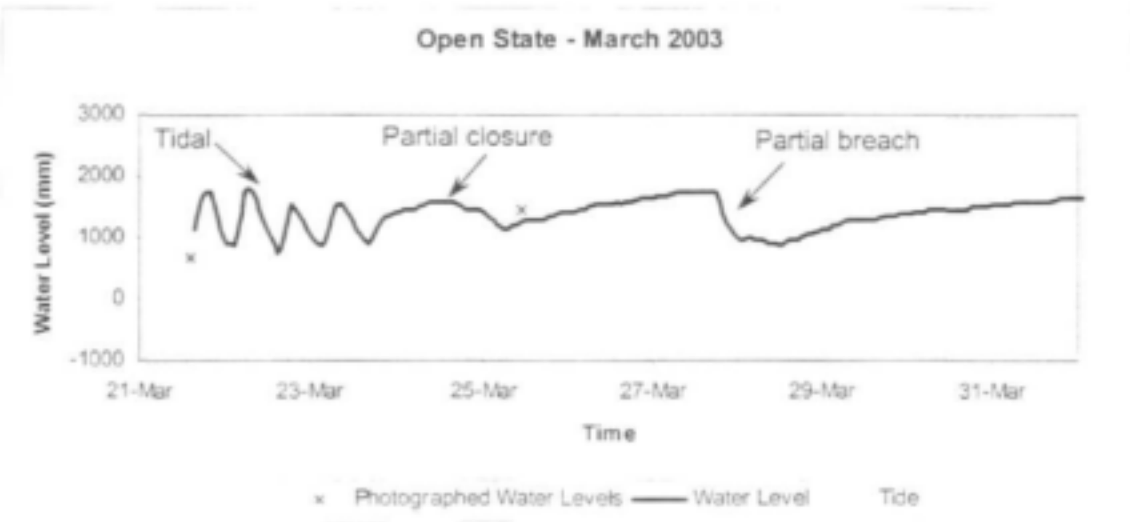


Figure 1.9 The first recorded water levels & tide heights obtained from the continuous water level sensor at Mhlanga during March 2003

During high tide the water level in the estuary reached up to 1800 mm above MSL, with the lowest recorded water level in the estuary, at low tide, approximately 740 mm above MSL. i.e. the spring tide sea level fluctuations of 2 m produced changes in the estuary water level of

1.06m. The tidal prism may be estimated if an assumption is made to relate changes in volume to changes in water level e.g. $\Delta Vol \sim \Delta H^k$, with $1 \le k \le 3$ depending on the morphology of the estuary. The observed water level fluctuations represent about half the water level range occurring in the estuary, which corresponds to a tidal prism of between 12% and 50% (k=1 & k=3 respectively) of the estuary's storage capacity.

Figure 1.6(a) is a detailed view of the water level fluctuations in the Mhlanga Estuary during an open phase in March 2003. The tidal level has been projected upwards, by 1 m (approximately 55% of the median significant wave height for Durban, see Table 1.3) to represent the effects of wave run-up.

The water level fluctuations seen in Figure 1.6 do not follow the same symmetric temporal variations as the tide, but rather exhibit asymmetry, with the increases in water levels occurring more rapidly than the decreases. Constriction of the mouth results in tidal asymmetry and in this case the estuary is flood dominated, as the flood tide is more intense than the ebb tide. The constriction of the mouth also results in the estuary water levels lagging behind the tidal fluctuation (e.g. Schumann, Largier and Slinger, 1999).

From the plots of water level against time it can be seen that the change in estuary water levels are smaller than the changes in sea level. Tidally driven water level changes in the estuary are inhibited by its perched nature. The difference between the estuary water levels and sea levels are far greater at low tide than at high tide.

Figure 1.10 is a detailed plot of recorded water levels during a typical tidal cycle. The cycle starts at point A, where the tidal level (increased to account for wave run-up) exceeds the water level within the estuary causing a rise in the water level within the estuary. The rise in tidal level is larger than the water level rise within the estuary. The water level continues to rise until point C despite the fact that the tide began dropping at point B some time before point C. Point C represents the point at which the tidal level and estuary water level are once again equal. As the tide continues to drop from C to D, the water level in the estuary also drops, but not as quickly. The estuary water level continues to drop after point D, despite the tide having turned, until the estuary water level matches that of the tidal level at A to restart the cycle.

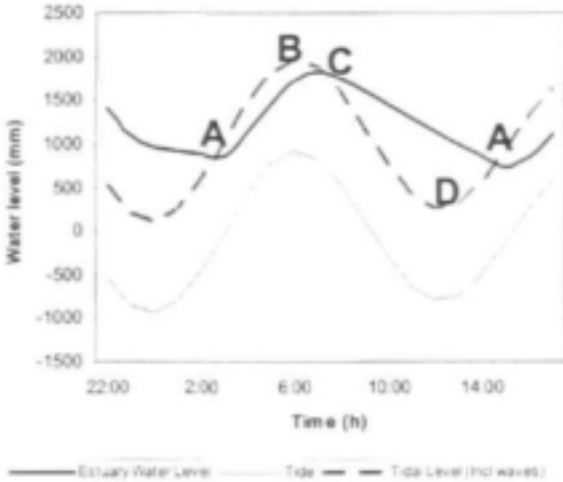


Figure1.10 Water levels and tide heights during an open phase at Mhlanga showing the effects of tidal-driven exchange flows

Figure 1.11 Photographs of the open mouth showing the narrow ebb channel and much wider flood channel.

Direct visual observations have provided an explanation for the flood dominated tidal exchanges at Mhlanga Estuary. At high tide, flood-water washing into the estuary under wave action uses the entire width of the mouth, as seen in Figure 1.11. However when the water leaves the estuary, it flows out of a small ebb-channel in the middle of the mouth thus causing the estuary to empty slower than it filled.

Figure 1.6 (b) shows the Mhlanga estuary under open conditions during June 2003. The tidal level in this case has been projected upwards, by 0.8 m (approximately 44% of the median significant wave height) to represent the effects of wave run-up. An interesting feature of this data is that the water level data does not follow the same pattern as for the case discussed above. As the tide shifted towards low tide the water level stopped dropping indicating that there was a sandbar at the mouth preventing the estuary from emptying any further.

The tidal fluctuations in the sea during the open phase were between 850 and 1350 mm, causing a 200 to 500 mm change in the water level within the estuary. The tidal influence seen here is smaller than that for the open state during March 2003. This may be attributed to different prevailing tidal range or wave conditions, as well as the mouth not having scoured as deeply.

Six high tide cycles are visible before the estuary closed on the 30 June, i.e. the estuary was open for about three days. At present it is not possible to determine whether the estuary is fully or only partially closed from the water level monitors i.e when the berm was fully restored. This information is best obtained from direct visual observations.

During July the estuary was open for two and a half days (Figure 1.6(c)), closing on the fifth high tide. As with the previous cases significant tidal exchanges occurred under open conditions. In Figure 1.6(c), the tidal level has been projected upwards, by 0.8 m (approximately 44% of the median wave height) to represent the effects of wave run-up. The tidal range at the time varied from 1350 to 1500 mm, while the water levels within the estuary varied by between 470 and 580 mm.

The mouth always closed at high tide thereby trapping a significant volume of saline water within the estuary. During mouth closure at the end of July 2003, the water level upon closure was about 1400 mm above MSL. The volume of saline water trapped in the estuary is estimated to be 10 – 30% of the estuary's storage capacity (depending on morphology).

A summary of the information pertaining to tidal exchange flows from the examples discussed above is given in Table 1.5.

Table 1.5 Summary of information from a sample of three open phases

	March '03	**June '03**	**July '03**
Tide range during open period - mm	1000 – 2000	850 – 1350	1350 – 1500
Water level range – mm	600 – 1000	200 – 500	470 – 580
Estimated tidal prism - % storage (k=2)	20%	5%	6%
No. high tides leading to closure	8	8	6
No of days open	4	4	3

1.6.6 Outflows during closed state

The two main outflows (or losses) from the estuaries when the mouths are closed are seepage and evaporation. This is probably typical of perched estuaries with narrow sand bars. Both seepage and evaporation losses are storage dependant, with seepage increasing as the water level within the estuary rises and evaporation increasing as surface area increases. When estuaries are perched the seepage losses can be substantial, and tend to dominate the evaporation losses. The seepage losses can be estimated by combining water level data and flow readings. When the water level remains constant in the estuary, there is an equilibrium with inflows equal to the outflows. Therefore flow readings taken at constant water levels are equal to the seepage losses at that water level. To determine the maximum seepage from the system, the relationship between flow rate and water level needs to be projected to the maximum water level.

During May 2003 the water level monitor at Mhlanga Estuary captured two different stages at which the water level remained constant for a period of time. Flow measurements were made during each of these two occasions. The water levels and flows recorded are tabulated in Table 1.6. Fortunately at Mhlanga Estuary a flow rate was captured the morning of a breach after the estuary had remained closed at the high water level for several days. This flow rate of $0.25m^3/s$ indicates the maximum seepage from the estuary. The average evaporation from the system was estimated by multiplying the MAE by the surface area. For Mhlanga Estuary the estimated evaporation from the system was calculated as $0.02m^3/s$, which is relatively small (<10%) compared to the estimated seepage and can therefore be ignored.

Table 1.6 Mhlanga seepage outflows

Date	Water level (m +MSL)	Flow rate (m^3/s)	Comment
7 May 2003	2.79	0.19	
23 May 2003	2.96	0.25	Morning of breach

Two data points, shown in Table 1.7, were similarly obtained for Mdloti Estuary, neither of which corresponded to the highest water level of the estuary.

Table 1.7 Mdloti seepage outflows

Date	Water level (m+MSL)	Flow rate (m^3/s)	Comment
19 Feb 2003	2.09	0.20	
25 Jun – 8 Jul '03	2.57	0.34	2-week average

There is insufficient data to accurately determine the relationship between water level and seepage, although it is clear that seepage increases with water level and that the seepage at Mdloti is significantly higher than at Mhlanga. It can be shown that a power law dependence of seepage on water level is expected, and was therefore used to extrapolate the data in order to estimate the maximum seepage flows at breaching water levels. The result (see Figure 1.12) for Mdloti is an estimate of $0.5m^3/s$ at a water level of 3.2m above MSL. Evaporation losses from the system when the estuary is full and the maximum surface area exposed, is approximately $0.02m^3/s$ or 5% of the maximum seepage.

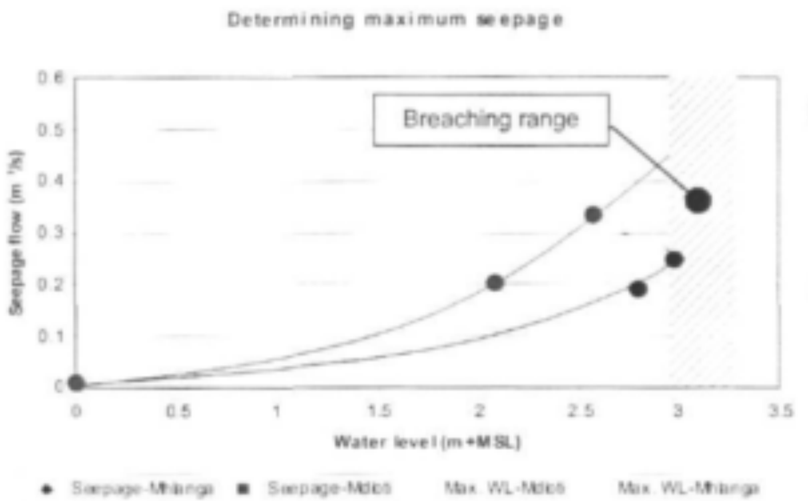


Figure 1.12 Extrapolation of seepage flows to determine the values at breaching water levels.

The significance of the above analysis of the outflows during closed conditions is related to the important role that these outflows have in the water balance of the estuaries under low inflow conditions. ***Only inflows that equal or exceed the maximum outflows, can raise the water levels to those at which breaching occurs. Conversely, any inflows that are less than the maximum outflows will have an equilibrium water level that is lower than the breaching level. The maximum seepage outflow therefore defines a critical flow rate Q_{CRIT} below which breaching will not occur and the estuary will remain closed.*** Note however that this does not imply that inflows exceeding Q_{CRIT} will result in immediate breaching of the estuary because there is a characteristic time delay, or residence time $T = S/Q$, associated with each flow that represents the time it takes for that flow to fill the storage in the estuary and raise the water levels to breaching levels.

A summary of the results ot the outflow analyses is given in Table 1.7.

Table 1.7 Summary of maximum outflows under closed conditions.

Parameter	MHLANGA	MDLOTI
Max outflow when closed Q_{CRIT} (m^3/s)	0.25	0.50
Evaporation (m^3/s)	0.02	0.02
Critical time scale $T_{CRIT} = S/Q_{CRIT}$	~ 35 days	~ 20 days

1.6.7 Mouth dynamics

Breaching mechanisms

An understanding of breaching mechanisms is important for developing models to predict the process. The present study provided an opportunity to gain some insight since Mhlanga estuary breached frequently and in a fairly predictable, quasi-periodic fashion during 2003. This was due to the low rainfall that gave rise to approximately constant (low) inflows. In fact flows were consistently at, or near, the critical flow rate as defined above.

A simplistic view of the breaching process is that it involves overtopping of the sand-bar due to rising water levels, and subsequent scouring of a channel to the sea. However, while this may occur under high flow (flood) conditions, it is not an accurate depiction of the breaching events that took place at Mhlanga during 2003 under low flow conditions. Observations have shown that high seepage flows through the sand-bar, and associated erosion, play a key role in the breaching process in this case. The process is depicted schematically in Figure 1.13 : strong seepage flows cause "sand-piping" erosion of the beach face and appear to eventually trigger slumping failure of the sand-bar, much like can occur in the failure of earth dams. The seepage-driven failure mechanism appears to require a time scale longer than a typical tidal ½ -cycle (~ 6hrs) to develop, with the result that this mode of breaching favours the neap tide phase where hydraulic gradients are more uniform over the tide cycle. Breaching consistently began at (or near) low tide when hydraulic gradients are highest, which is further indication of the important role played by seepage in the process. Unfortunately, an actual breaching event has not yet been observed since they occurred at night.

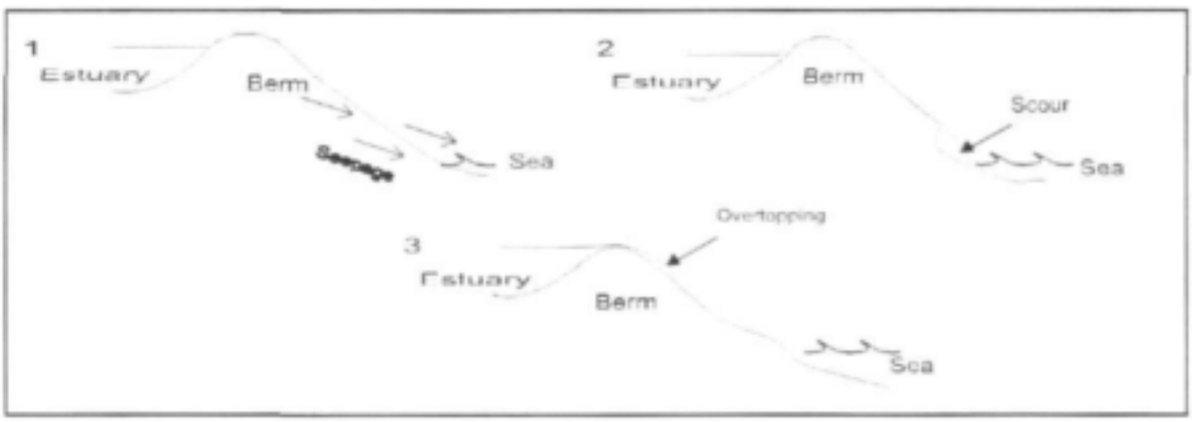


Figure 1.13 Schematic depiction of the role played by seepage in breaching

Figure1.14 shows an example of observed seepage erosion of the beach face just a few hours before breaching took place on 15 November 2003. The breaching water levels on this occasion were slightly higher than during previous recorded events, and the observed erosion was somewhat more pronounced as a result.

Wave over-topping during spring high tide is a common occurrence at the case study estuaries and appears to have a significant effect on the morphology of the sand-bar. The swash tends to widen and reduce the height of the berm by pushing sediment across-shore into the lagoon. The

Figure 1.14 Observed seepage erosion of the beach face prior to breaching

widening of the sand-bar is clearly illustrated in Figure 1.15. It is interesting to note that the effects are less pronounced towards the southern end of the berm, which is the current breaching location. This could be due to some wave-sheltering provided by a near-shore reef at that location and may explain why breaching occurs there. Widening of the berm by this process reduces the local hydraulic gradient and could therefore inhibit seepage failure of the sand-bar. However it also reduces the berm height which could promote overflowing and associated scour.

Breaching water levels

Water level monitoring has shown that breaching occurs at water levels 3000 – 3400mm above MSL for both estuaries. Before September 2003, Mhlanga seemed to breach consistently at water

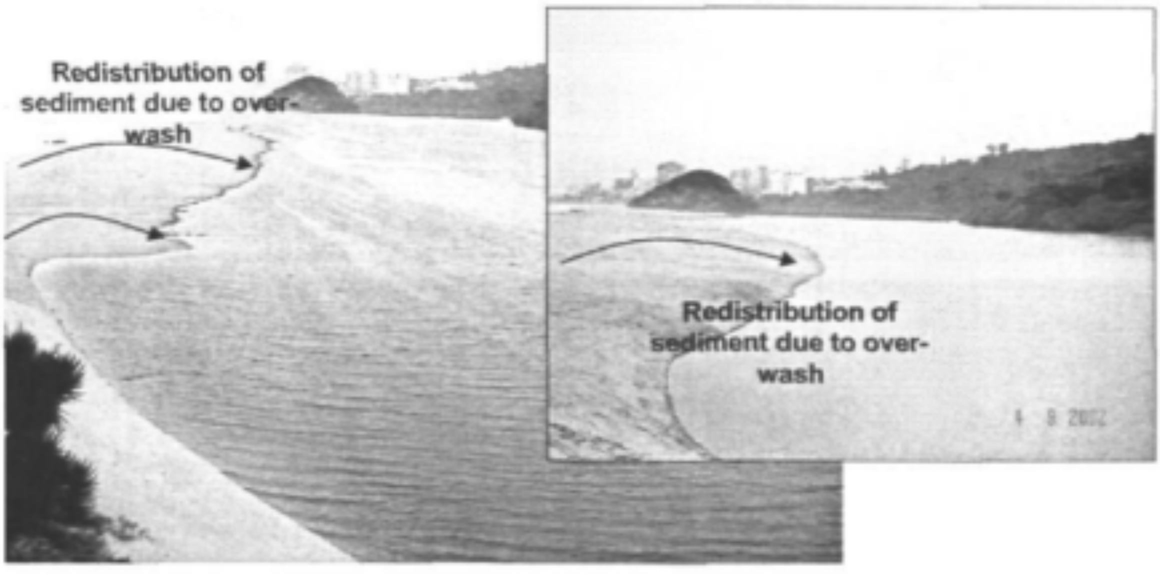


Figure 1.15 Observed wave over-topping effects on the sand-bar (looking South)

levels of 3000mm (or slightly less), however on 15 September 2003 the water levels reached 3390mm MSL just before breaching. This change could be linked to the high rainfall that occurred just before and during that breaching event.

Watermarks on the M4 road-bridges at the estuaries provide a clear indication of the typical breaching levels (Figure 1.16)

Breaching water levels also give an indication of the height of the wave-built berm at the breaching location. It is expected that the berm height is determined by wave heights, beach slope and sediment characteristics (e.g. Bagnold, 1940). The observed breaching levels are about $1.8H_s$ above MSL, where $H_s = 1.8$m is the median significant wave height at this location. Alternatively they are about $1.2H_s$ above spring high tide levels. This is consistent with Bagnold.

Breaching frequency

From March 2002 to March 2003 there were 23 breaching events recorded at Mhlanga estuary (including changes from "closed" to "partly open" states). During the same period there were 10 breaching events recorded at Mdloti estuary. During 2003, the mouth dynamics changed significantly for both estuaries. Mdloti remained closed for the entire period from 9th February through to December 2003. In contrast, at Mhlanga estuary there was a quasi-cyclic breaching pattern, with the mouth breaching 8 times between March 2003 and December 2003 i.e. a breaching frequency of once every 30 – 40 days.

The change in breaching patterns during 2003 can be simply explained in terms of the critical flows defined in section 1.6.6. Flows at Mdloti during that period were consistently less than its critical value $Q_{CRIT} = 0.5m^3/s$ which means that water levels remained below breaching levels and the estuary therefore remained closed. In contrast, flows at Mhlanga were consistently at (or slightly above) the critical flow $Q_{CRIT} = 0.25m^3/s$ due to WWTW discharges ($0.23m^3/s$). The near constant flow regime led to a quasi-periodic filling and breaching pattern on a time scale of order $S/Q_{CRIT} = 37$ days

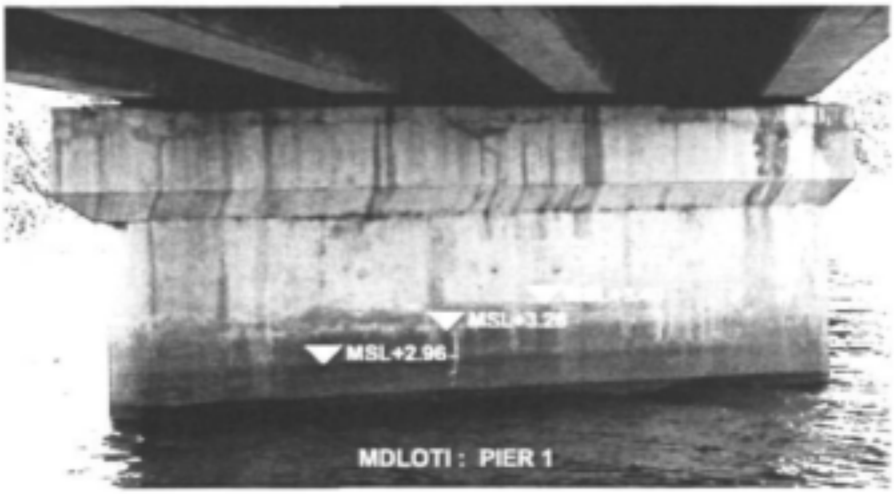


Figure 1.16 Watermarks on the M4 road bridges showing normal breaching levels.

A summary of mouth breaching statistics for March 2002 – July 2003 is given in Table 1.8.

Table 1.8 Summary of the mouth states March 2002 – July 2003.

	Parameters	Quantity	Comment
MDLOTI	No. of breaches	9	8 at neap tide ± 3 days
	No. of partial breaches	4	3 at neap tide ± 3 days
	No. of days fully open	103	21%
	No. of days partly open	47	10%
	No. of days closed	328	69%
	Average open duration	12	(Open days)/(# breaches)
MHLANGA	No. of breaches	18	12 atneap tide ± 3 days
	No. of partial breaches	14	7 at neap tide ± 3 days, 1 at spring
	No. of days fully open	146	27%
	No. of days partly open	97	18%
	No. of days closed	296	55%
	Average open duration	7	(Open days)/(# breaches)

Breaching time scales

The continuous water level sensor in Mhlanga has provided a detailed record of the change in water levels during the breaching process. Three examples of the process from May, June and July 2003 are shown in Figure 1.7 and a summary of information from these breaching events is provided in Table 1.9.

The data in Table 1.9 shows that breaching consistently began at low tide, with water levels dropping for 5 hours before tidally driven flows began to enter the open mouth. Water levels changed most rapidly during the 1st two hours after breaching. The average change in water level during the 1st five hours of breaching was approximately 1.9m. Given an estimated storage volume of 800000m^3 (Table 1.3), this translates into a flow of 44m^3/s. For comparison, this is slightly greater than the peak discharge of 36m^3/s estimated by Jezewski *et al* (1984) for a 2-year return period flood event at this location.

Table 1.9 Summary of information from three breaching events during 2003.

		May	June	July
Tidal Situation		Neap	Neap + 4 days	Neap + 4 days
Tide		Low	Low	Low
Water Level before breach		2955	2915	2810
Rainfall during closed period (mm)		50 (~7wks)	22 (~4wks)	< 1 (~4wks)
Reduction in water level during the hours following breaching	**Hour 1**	285	516	998
	Hour 2	1240	980	622
	Hour 3	255	219	131
	Hour 4	106	85	86
	Hour 5	43	61	38
	Hour 5-10	123	n/a	n/a

Mouth re-closure time scales

The continuous water level sensor in Mhlanga provided a detailed record of the mouth re-closure process following breaching events. Furthermore, the regularity of the breaching at Mhlanga made a detailed 2-week monitoring program of a specific breaching and re-closure cycle possible at the end of June 2003.

Mouth re-closure was observed to take place over 3 – 6 days under the low flow conditions (of order 0.3 m^3/s - see figure 1.3) that prevailed during the monitoring period.

The water level monitors indicate that the estuary consistently closes at high tide. In June and July 2003 the estuary took 4 and 3 days to close respectively i.e. 6 – 8 tidal cycles. During the two-week period at the end June 2003, detailed observations of the mouth re-closure process were made (Zietsman, 2003), including daily photometric surveys of the berm as it was rebuilt by wave action. The approximate volume of sediment lost from the sand berm when the mouth opened on the 26 June 2003, was estimated as $1400m^3$. Over the 4 days it took for the mouth to re-close, the sand bar within the channel built up at a rate of approximately $200m^3$ during each high tide ($400m^3$/day or about $15m^3$/day per meter of coastline since the breach was ~ 25m wide). This may be compared with the estimated long-shore sediment transport rate for this location of about $1400m^3$/day (see Table 4.3). Wave parameters were not measured[3].

The available data does not allow the effects of flow on mouth re-closure times to be accurately estimated because the flows during 2003 did not vary substantially. It seems reasonable to assume that higher flows inhibit the re-closure process by their scouring action and by reducing the tidal prism and associated sediment deposition in the mouth. An example of this occurred during the breaching recorded in November 2003. The mouth breached 2-hours after low tide at 23h00 on 19th November. (1-day after neap). The estuary was still (partially) open two weeks later on 3rd December when a field visit was made to retrieve the water level sensor. During the open phase, about 25mm of rainfall fell on the 26th November, which would have increased the flows and probably contributed to the extended open phase. A flow measurement during the field visit showed that the flow had reduced to $0.36m^3/s$, which is low enough not to inhibit closure.

Using the mouth state data shown in Figure 1.2 the average time that the estuaries remain open after a breaching event can be estimated by dividing the number of open days by the number of breaching events. Splitting the data into two periods, March 2002 to March 2003, and March 2003 to November 2003, yields an estimate of 6 days for Mhlanga during both periods, even although the flows during 2002 were generally higher than in 2003. Mdloti did not breach during the second period, but the average open time between March 2002 and March 2003 was 13 days i.e about twice the value for Mhlanga. However the sample is too small to establish an accurate estimate and it is tentatively concluded that typical mouth re-closure times for Mhlanga and Mdloti may both be assumed to be T_{CLOSE} ~ 6 days and that they do not vary strongly with flow rates.

[3] Wave data is available from a monitoring station at the port of Durban but was not obtained for this study.

1.7 Modelling the relationship between flow and mouth dynamics

1.7.1 Background

A key objective of this work was to provide data and a methodology for predicting changes in the frequency, timing and duration of mouth closure for various flow scenarios. In this section the monitoring results are reviewed in the context of this objective.

One method of analysing the effects of flow variations on mouth dynamics is to link a statistical flow-duration analysis with a model that describes the relationship between mouth state and flow. This is current practice in implementing the RDM process for determining ecological reserve requirements for estuaries. Typically the model for linking flow to mouth state consists of empirically specified flow thresholds that delineate different states (open, closed, partly open/closed). The monitoring results of the present study have highlighted the limitations of this approach (section 6.4). Mouth breaching is directly related to water levels, and only indirectly to inflows through their role in the water balance of the systems. As already noted, except for flows at the extremes of the spectrum, there is no clear and direct link between instantaneous flow rates and mouth state. Observations concerning the mouth dynamics of Mhlanga during the period from March 2003, has led to considerable insight into breaching patterns and to a somewhat different conceptual model of the process. This is based more directly on the water balance and associated water level variations, rather than purely on inflow magnitudes.

1.7.2 Conceptual model

The results of the present study may be assimilated into a simplified conceptual model to relate mouth dynamics to flows. The key features of the flow - mouth state relationship that have emerged are as follows:

(1) Firstly, there is a critical flow rate Q_{CRIT} (based on the maximum outflows that occur when the estuary is in a closed state) below which water levels do not increase to breaching levels i.e. the estuary will be closed 100% of the time for flows $Q/Q_{CRIT} < 1$. This flow regime will be denoted the closed (**C**) regime. Note that the residence time associated with the critical flow rate, $T_{CRIT} = S/Q_{CRIT}$ indicates the time interval between periodic breaching events under constant (critical) inflow conditions.

(2) Secondly, there is a characteristic time-scale T_{CLOSE} required for mouth closure by wave action and associated sediment transport. Flows that have residence times S/Q similar to, or shorter than, the closure time scale should be able to maintain open mouth conditions. A flow

regime where the mouth is open 100% of the time can therefore be defined by $Q/Q_{CRIT} > T_{CRIT}/T_{CLOSE}$. This flow regime will be denoted the open (**O**) regime.

(3) Thirdly, for flows in the range $1 < Q/Q_{CRIT} < T_{CRIT}/T_{CLOSE}$ the estuary may be expected to breach at regular intervals $T \sim S/Q$ that decrease as the flow increases so that the proportion of time that the estuary is closed reduces from 100% to 0% in this regime. For simplicity a linear variation with flow is assumed here. This flow regime will be denoted the intermittently open/closed (**O/C**) regime.

A schematic representation of this conceptual model for the relationship between flow and mouth state is given in Figure 1.17. Note that the **O/C** regime as envisaged here is **not** the same as the "partly open state" defined in CSIR (2002, 2003) which was described as a quasi-stable state with outflows through a perched outlet channel, but with no tidal exchanges. Observations at Mhlanga and Mdloti during this study suggest that, in practice, such a state is difficult to define unambiguously, and seems to occur mainly as a transitional state between open and closed conditions. Nevertheless, the **O/C** regime could incorporate such a state if it occurs. In the present model, the **O/C** regime is better described as **intermittently** open/closed i.e. given a sustained flow rate in this regime, mouth breaching would occur quasi-periodically at intervals of order S/Q. However a key feature of estuaries that are functioning in this regime is that **there is no direct link between instantaneous flow rates and mouth state at a specific time**, although breaching should occur within a time-scale of order S/Q or less if that flow is maintained. In contrast, flows in the **C** or **O** regime should always be associated with closed or open mouth conditions respectively provided they persist for times ~ T_{CLOSE} or longer.

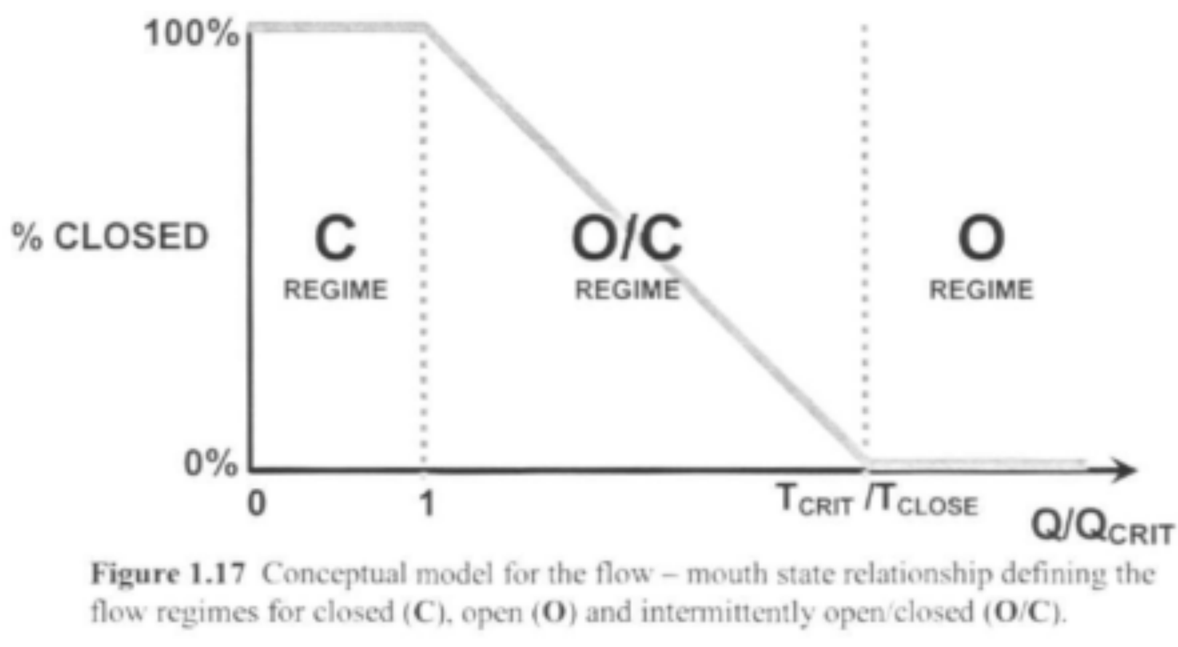


Figure 1.17 Conceptual model for the flow – mouth state relationship defining the flow regimes for closed (**C**), open (**O**) and intermittently open/closed (**O/C**).

A key simplifying assumption in this model is that the mouth closure time scale T_{CLOSE} is independent of the flow rate. This is probably not accurate, and may need to be refined. Such effects could be incorporated by allowing for a non-linear relationship between flow and mouth state in Figure 1.17.

It is common to consider river flows as comprising a "base-flow" component, and a "flood" or "quick-flow" component. Base-flows have low magnitudes (typically $Q/MAR < 1$) and vary slowly, with time scales of order weeks to months. The flood or quick-flow component may have high peak magnitudes (typically $Q/MAR >> 1$) but are generally short-lived episodic events where the flows vary over time scales characterised by the time of concentration for the catchment. For example, time of concentration estimates for Mhlanga and Mdloti are 6 and 12 hours respectively (Jesewski *et al*, 1984, and Table 1.3).

It is anticipated that most flood events with return periods of 1-year or longer, would have flow magnitudes and associated residence times that place them in the **O** regime of Figure 1.17. For example 2-year flood peaks for Mhlanga and Mdloti were estimated by Jezewski *et al* (1984) as $76m^3/s$ and $36m^3/s$ respectively, which have corresponding residence times of about 4-hrs and 6-hrs respectively. Such flows would clearly cause the mouth to open within a few hours. These episodic flood events are unpredictable and could have major impacts on the morphology of estuaries by redistributing or removing sediments etc (which can in turn severely impact the biota of the system). However the high magnitude of these flows makes them relatively insensitive to human induced changes (except for major developments such as dams) which limits their relevance to the management of estuaries.

In contrast base-flows are more persistent and respond more predictably to changes in average climatic conditions (e.g. seasonal variations). For TOCEs base-flows are expected to be in the **C** or **O/C** regimes defined above (perhaps varying seasonally) and would constitute the most common flow scenario. The magnitude of base-flows is probably best represented by the median (50^{th} percentile) flow, which for small catchments is typically much smaller than the mean flow (MAR). For example, statistical flow-duration analyses for the Mhlanga and Mdloti catchments (see section 1.7.2 below) give the median flows as about 40% of MAR. The lower magnitude of base-flows can make them more sensitive to human induced impacts and they are therefore more relevant from a management perspective. Changes to median flows e.g. by WWTW discharges or abstractions, can dramatically alter the normal functioning of an estuary by changing its typical state from the **C** regime to the **O/C** regime or vice versa.

The conceptual model discussed above provides a framework to consistently delineate flow regimes for the purposes of analysing mouth dynamics. Although based mainly on observations at Mhlanga, it has deliberately been formulated in non-dimensional terms (using time scale ratios) in the hope that it may have some general applicability to other perched temporary open/closed estuaries, but this remains to be tested.

1.7.3 Application of the model

In this section, the conceptual model discussed above will be applied to the two case study estuaries as an illustration.

Normalised flow-duration curves (FDCs) for quaternary catchments U30A and U30B, based on 70 years of simulated, naturalized monthly flow data from WR90 are shown in Figure 1.18 (a). The differences between the normalised FDCs are small, as expected, since they are adjacent catchments. For small catchments (typical for TOCEs in South Africa), monthly averaged flows provide a degree of averaging that may be unsuitable for establishing links to mouth dynamics. Daily averaged flows seem more appropriate for this application. However, simulated monthly flow data is available for all drainage regions in South Africa from WR90 (i.e.Midgley *et al*, 1994), while daily flow data is generally not readily available. Smakhtin (2000) gives a simple approximate method for deriving annual FDCs for daily flows from annual FDCs for monthly flows. The annual FDCs for daily flows, derived using this approximate method are given in Figure 1.18 (b). The differences between the FDCs for monthly and daily flows are small for the range of flows between the 5^{th} and the 95^{th} percentiles. Since this is the range of flows that is most important for assessing the effects of flow changes on the mouth dynamics of TOCEs, this result suggests that monthly averaged flows may be adequate for this application. Nevertheless, the daily flow FDCs will be used here for illustrative purposes.

Normalized FDCs derived for present state conditions at Mhlanga and Mdloti estuaries are shown in Figure 1.19. The Mhlanga FDC was derived by adding estimated WWTW discharges (assumed constant) to the reference state FDC for U30B. In the case of Mdloti, the present state FDC was obtained by analysing data from stream-gauge U3H005, combining this with the reference state FDC of U30B (area weighted), and then incorporating abstractions and WWTW discharges (both assumed constant). Note that the present state FDC for Mdloti shows a severe drop at low flows due to the assumed abstractions. This may not be realistic, but since the flows above the 95th percentile are of little relevance to the present application, no attempt has been made to improve the FDC.

Combining the model described in section 1.7.1 with the FDCs yields the results shown in tables 1.10 and 1.11 for Mhlanga and Mdloti respectively. Results using the flow thresholds defined by the recent RDM studies (CSIR, 2002, 2003) are also shown for comparison[4]. The reference and present state MAR figures for Mdloti are those given by CSIR (2002) rather than those estimated for the present study.

In order to compare the results from the RDM study with the present model, some assumption is required to relate the two **O/C** regimes. Recall that in the present model, mouth statistics vary from 100% – 0% closed as flows vary from the lower to upper end of the range that defines this regime. The nearly exponential form of the FDCs suggests that statistically a greater proportion of time is spent in the lower part of the flow range, therefore it will be assumed for the present purposes that $^2/_3$ of the **O/C** time represents closed conditions, and $^1/_3$ open conditions.

Table 1.10 Model analysis results for the effects of flow on the mouth state of Mhlanga estuary for reference and present conditions. Results from CSIR (2002) are also shown.

MHLANGA REFERENCE			**MHLANGA PRESENT**		
MAR =	12.6	Mm^3/yr	MAR =	12.60	Mm^3/yr
	0.40	m^3/s		0.40	m^3/s
WWTW	0.00	Mm^3/yr	WWTW	7.20	Mm^3/yr
Abstractions	0.00	Mm^3/yr	Abstractions	0.00	Mm^3/yr
TOTAL FLOW	12.60	Mm^3/yr	TOTAL FLOW	19.80	Mm^3/yr
	0.40	m^3/s		0.63	m^3/s
STORAGE	750000	m^3	STORAGE	750000	m^3
Q_{CRIT}	0.25	m^3/s	Q_{CRIT}	0.25	m^3/s
Q_{CRIT}/MAR	0.63		Q_{CRIT}/MAR	0.63	
T_{CRIT}	35	Days	T_{CRIT}	35	Days
T_{CLOSE}	6	Days	T_{CLOSE}	6	Days
Q_{OPEN} =	5.8	Q_{CRIT}	Q_{OPEN} =	6	Q_{CRIT}
Q_{OPEN} =	1.4	m^3/s	Q_{OPEN} =	1.4	m^3/s
Q_{OPEN}/MAR	3.6		Q_{OPEN}/MAR	3.6	
RDM (CSIR, 2003)					
C	**O/C**	**O**	**C**	**O/C**	**O**
< 0.4	0.4 - 0.5	> 0.5	< 0.4	0.4 - 0.5	> 0.5
80%	4%	16%	60%	15%	25%
PROPOSED MODEL					
C	**O/C**	**O**	**C**	**O/C**	**O**
< 0.25	0.25 - 1.4	>1.4	< 0.25	0.25 - 1.4	>1.4
75%	20%	5%	1%	92%	7%

[4] Note however that the results given here may differ slightly from those in the CSIR reports because the FDC analysis has been done independently and may reflect different assumptions and estimates e.g. for discharges and abstractions.

Referring to Table 1.10 it can be seen that under reference conditions Mhlanga functioned mainly in the **C** regime. The two models are consistent (after distributing the **O/C** classification of the present model to **C** and **O**). Under present conditions the present model indicates that the WWTW discharges have caused the estuary to shift almost entirely to regime **O/C** at the expense of **C**. i.e prolonged periods of closed mouth conditions under present flow conditions will be very rare. The RDM model does not pick this up, and continues to allocate 60% of the time to the **C** regime. The observations made during this research project are clearly consistent with the conclusion that the estuary is now functioning in the **O/C** regime[5].

Under reference state conditions the median flow magnitude at Mhlanga is 40%MAR or $0.16m^3/s$. This is lower than the critical flow of $0.25m^3/s$ which indicates that typical base-flows at this estuary were originally in the **C** regime so that mouth breaching was dependant on episodic flood events.

Under present conditions the median flow magnitude at Mhlanga is estimated as $0.4m^3/s$ (Figure 1.19). This is higher than the critical flow of $0.25m^3/s$ which indicates that typical base-flows at this estuary have now changed to the **O/C** regime so that regular mouth breaching will occur independently of those due to episodic flood events.

According to the model, proposed future increases in the WWTW discharges by 75% (CSIR, 2003) will not change the main functional regime from **O/C**, but will have the effect of increasing the average breaching frequency.

From Table 1.11 it can be seen that under reference conditions the present model suggests that Mdloti functioned mainly within the **O/C** regime and is again in good agreement with the RDM model. Both models also indicate only a small percentage of time in the **C** regime under reference conditions. Historical observations discussed in CSIR (2002) suggest that this estuary was originally nearly permanently open.

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[5] This is not surprising since those observations were used to help formulate the model !

Table 1.11 Model analysis results for the effects of flow on the mouth state of Mdloti estuary for reference and present conditions. Results from CSIR (2002) are also shown.

MDLOTI REFERENCE			MDLOTI PRESENT		
MAR =	98.70	Mm^3/yr	MAR =	72.3	Mm^3/yr
	3.13	m^3/s		2.29	m^3/s
WWTW	0.00	Mm^3/yr	WWTW	2.70	Mm^3/yr
Abstractions	0.00	Mm^3/yr	Abstractions	5.90	Mm^3/yr
TOTAL	98.70	Mm^3/yr	TOTAL	69.1	Mm^3/yr
FLOW	3.13	m^3/s	FLOW	2.19	m^3/s
STORAGE	1000000	m^3	STORAGE	1000000	m^3
Q_{CRIT}	0.5	m^3/s	Q_{CRIT}	0.5	m^3/s
Q_{CRIT}/MAR	0.16		Q_{CRIT}/MAR	0.22	
T_{CRIT}	23	days	T_{CRIT}	23	Days
T_{CLOSE}	6	days	T_{CLOSE}	6	Days
Q_{OPEN} =	3.9	Q_{CRIT}	Q_{OPEN} =	3.9	Q_{CRIT}
Q_{OPEN} =	1.9	m^3/s	Q_{OPEN} =	1.9	m^3/s
Q_{OPEN}/MAR	0.6		Q_{OPEN}/MAR	0.8	
RDM (CSIR, 2002)					
C	**O/C**	**O**	**C**	**O/C**	**O**
< 0.3	0.3 - 2.0	> 2.0	< 0.3	0.3 - 2.0	>2.0
5%	70%	25%	20%	55%	25%
PROPOSED MODEL					
C	**O/C**	**O**	**C**	**O/C**	**O**
< 0.5	0.5 - 1.9	> 1.9	< 0.5	0.5 - 1.9	> 1.9
10%	65%	25%	30%	50%	20%

The main change from reference to present conditions is a significant shift towards the **C** regime i.e. a greater proportion of closed conditions due to the overall reduction in flows. This is again consistent with historical observations (CSIR, 2002).

The median flow at Mdloti under reference conditions is estimated as about $1.1m^3/s$ which is above the critical flow of $0.50m^3/s$. This indicates that base-flows at Mdloti were originally in the **O/C** regime so that regular breaching occurred in addition to those caused by episodic flood flows.

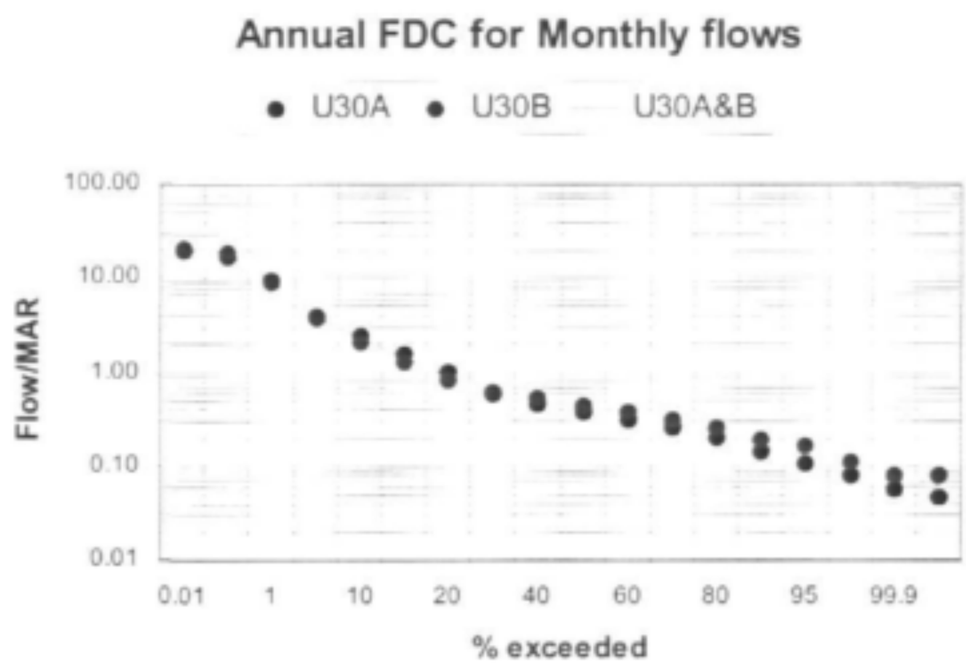
The median flow at Mdloti under present conditions is estimated as about $0.66m^3/s$ (Figure 7.3) which is closer to, but still higher than the critical flow of $0.50m^3/s$. Average base-flows at Mdloti therefore remain in the **O/C** regime generating regular breaching (albeit at lower average frequency) in addition to those caused by episodic flood flows.

A direct comparison of the model predictions and observational data from the present study is difficult because the rainfall during the monitoring period was well below average (69% and

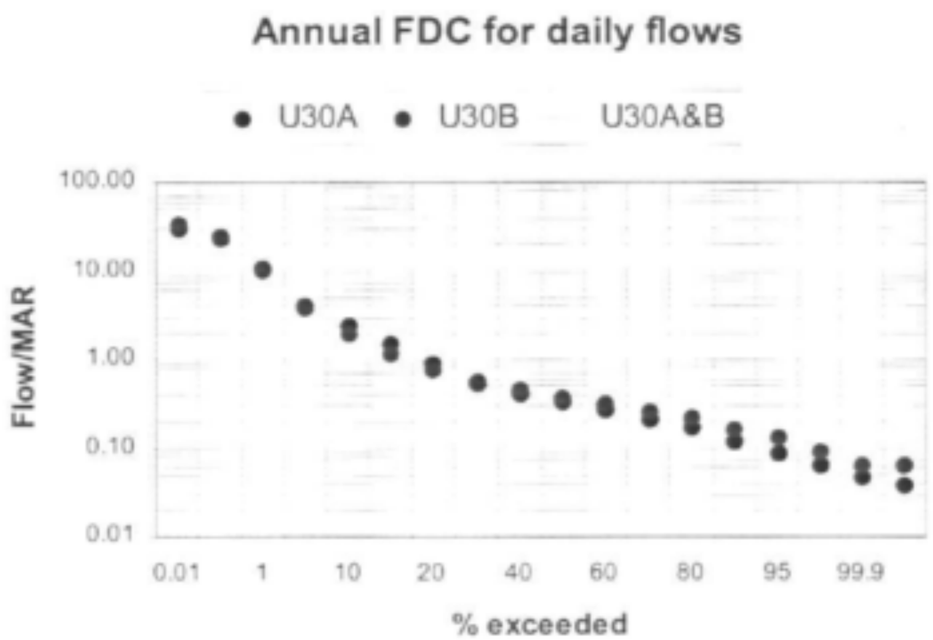
55% for 2002 and 2003 respectively), and the inflow monitoring did not have sufficient temporal resolution for flow-duration analysis. However if the area-adjusted daily flow data from U3H005 are used for Mdloti estuary (see Figure 1.3), then the model can be used to calculate the precise open/closed percentages. These can then be compared with observations from the same period. Results for March 2002 – March 2003 are shown in Table 1.12 and show good agreement. Slight adjustment of the T_{CLOSE} parameter from 6 days to 7 days would give exact agreement. However a longer monitoring period incorporating greater climatic variability is required for any sensible validation of the model.

Table 1.12 Comparison of observations with model results for Mdloti during the period March 2002 – March 2003 using adjusted U3H005 flow data. Observations classified as "partially open" are included in the "open" total (and the percentage given in brackets).

ESTUARY	Rainfall % average	MODEL RESULTS		OBSERVED	
		% Closed	% Open	% Closed	% Open
MDLOTI	69%	62%	38%	58%	42% (12%)

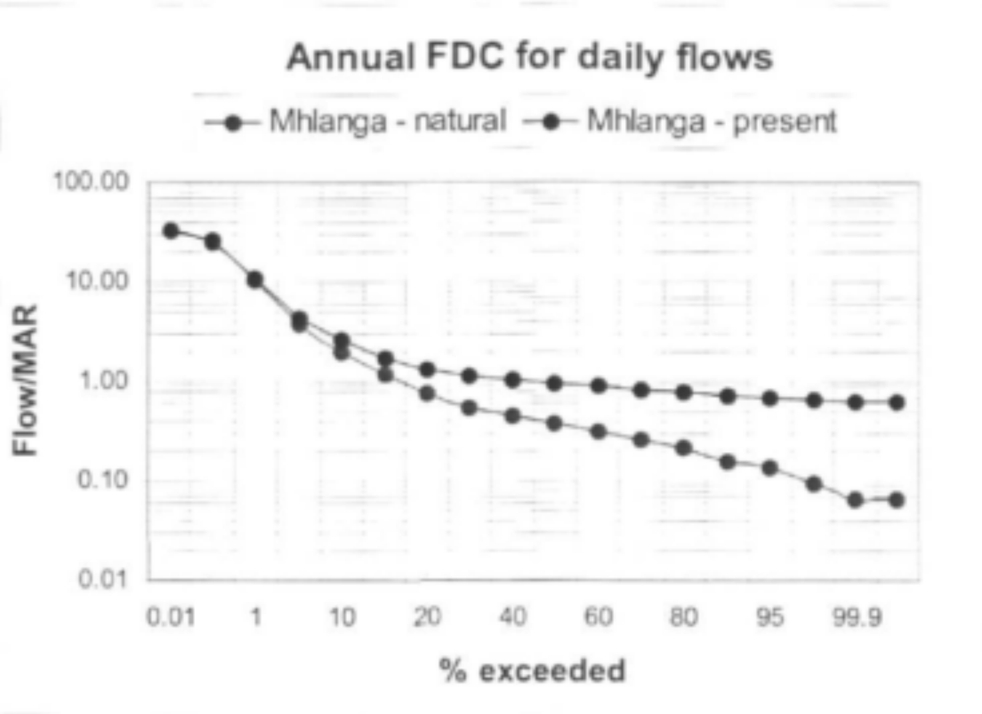


(a)

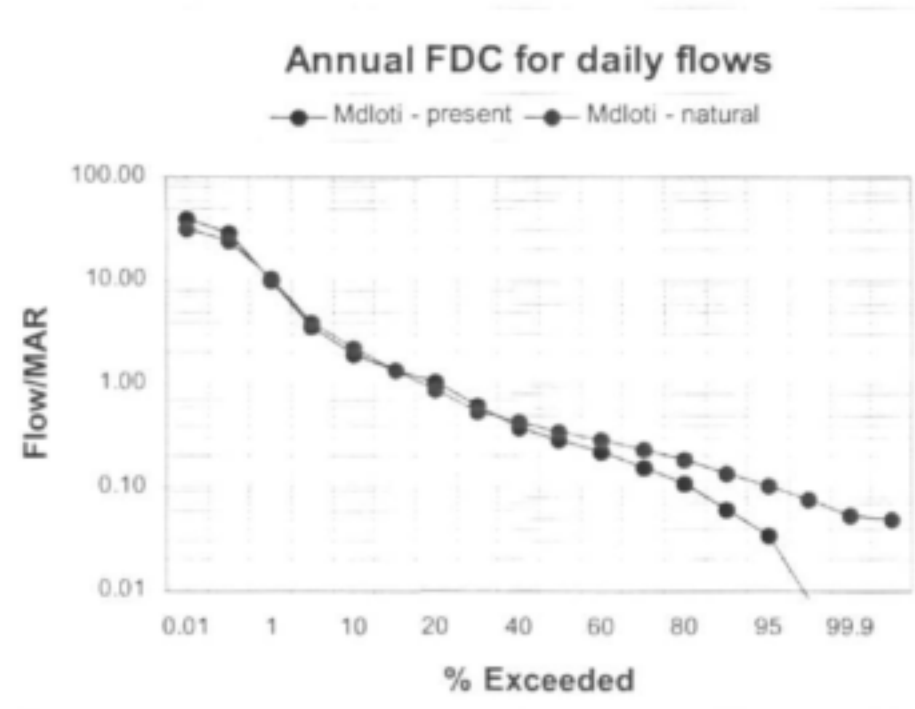


(b)

Figure 1.18 Normalised annual flow duration curves for monthy averaged flows (a) and daily averaged flows (b) for reference (natural) conditions. Monthly simulated flows from WR90 were used to obtain (a) and the method of Smakhtin (2000) was used to derive (b). The curves are normalised by the natural (reference state) MAR.



(a)



(b)

Figure 1.19 Normalised annual flow duration curves for daily averaged flows under present conditions at (a) Mhlanga and (b) Mdloti. The curves were obtained by incorporating the changes induced by abstractions and WWTW discharges and are normalised by the natural (reference state) MAR.

2. PHYSICO-CHEMICAL PARAMETERS

2.1 Temperature

Water temperatures followed a typical subtropical coastal pattern, with very small differences observed between summer and winter averages (Figs. 2.1 & 2.2). In the Mhlanga Estuary water temperature ranged from 13.7 °C at the surface in July 2002 (upper reaches) to 30 °C at the bottom during February 2003 (middle reaches). Summer values varied from 21°C to 30°C and winter values from 14°C to 24°C (Fig. 2.1). At the Mdloti Estuary, the highest water temperature of 30°C was again recorded during February 2003, at the surface (middle reaches) and the lowest, 14.5°C, during July 2002 at the bottom (lower reaches). Summer values ranged from 19°C to 30°C and winter values from 14.5°C to 24°C (Fig. 2.2).

2.2 Salinity

At the Mhlanga Estuary, salinity values ranged from 0.1 ‰ in the upper reaches, during January 2003, to 32 ‰ in the lower reaches during May 2002. Due to the generally narrow and low berm separating its waters from the ocean, the Mhlanga appeared to have received a steady supply of seawater via overtopping. This estuary also experienced more frequent breaching than the Mdloti. As a result, salinity in the lower reaches never dropped below 4-5 ‰ (Fig. 2.3). At the Mdloti Estuary, on the other hand, seawater input was much less regular and, as a result of this, the salinity maxima recorded were substantially lower, about 24 ‰ in October 2002. Also, freshwater predominated in the upper and middle reaches for much of the year, and occasionally even in the lower reaches of the estuary (Fig. 2.4).

2.3 Dissolved Oxygen

Several situations of hypoxic/anoxic conditions were observed in both estuaries, particularly during prolonged periods of mouth closure, predictably mainly in bottom waters and in the upper and middle reaches (Figs. 2.5 & 2.6). The highest dissolved oxygen (DO) value recorded in the Mhlanga Estuary was 12.2 mg l^{-1}, in the surface waters of the middle reaches during April 2002. The lowest was 0.08 mg l^{-1} and was recorded at the bottom of the upper reaches during the same month (Fig. 2.5). In the Mdloti, both highest (13.21 mg l^{-1}) and lowest DO values (1.8 mg l^{-1} mg l^{-1}) were recorded in the lower reaches of the estuary, during September and April 2002, respectively (Fig. 2.6).

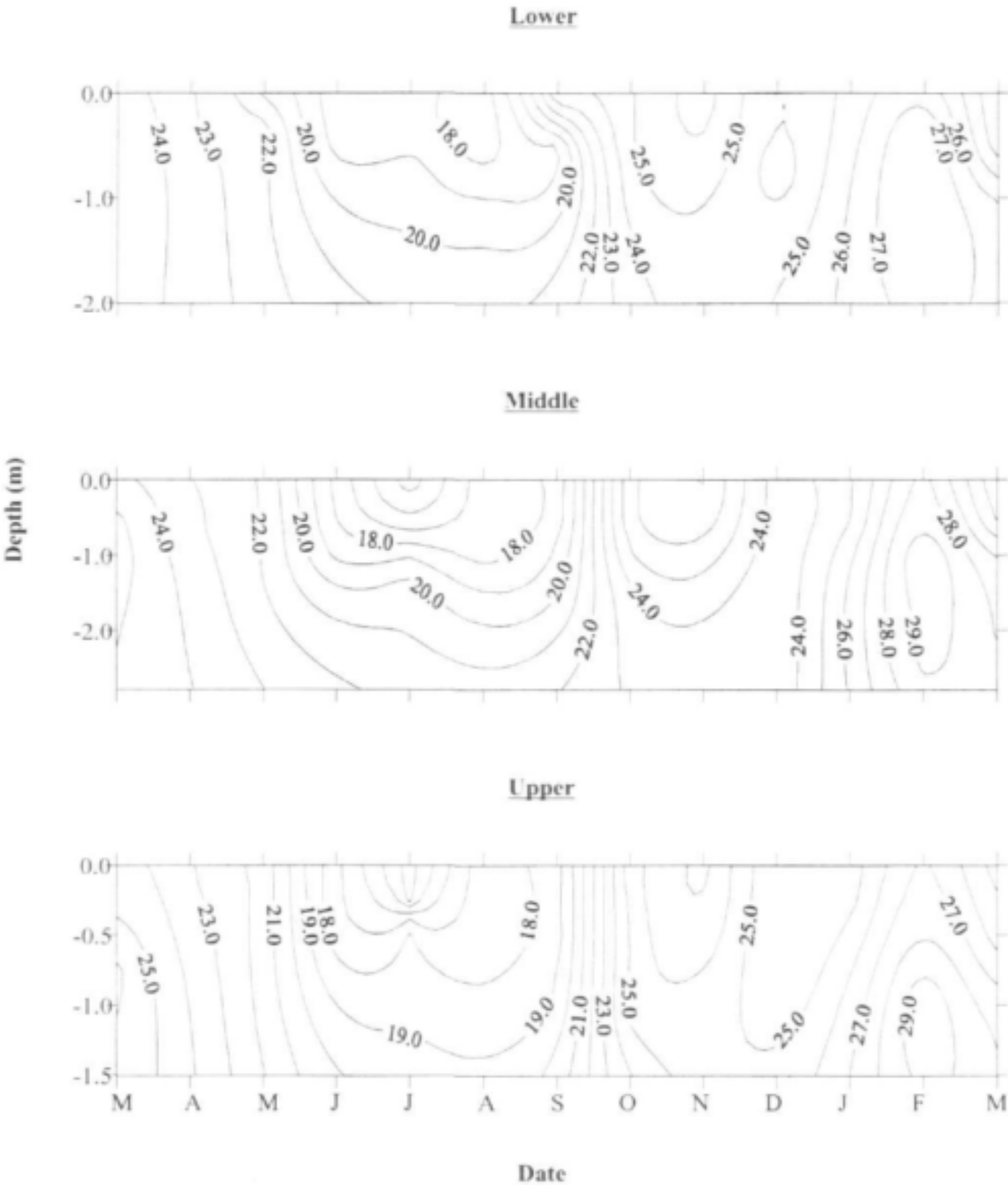


Figure 2.1. Annual variations of temperature (°C) with depth in the three reaches of the Mhlanga Estuary (March 2002- March 2003, monthly measurements).

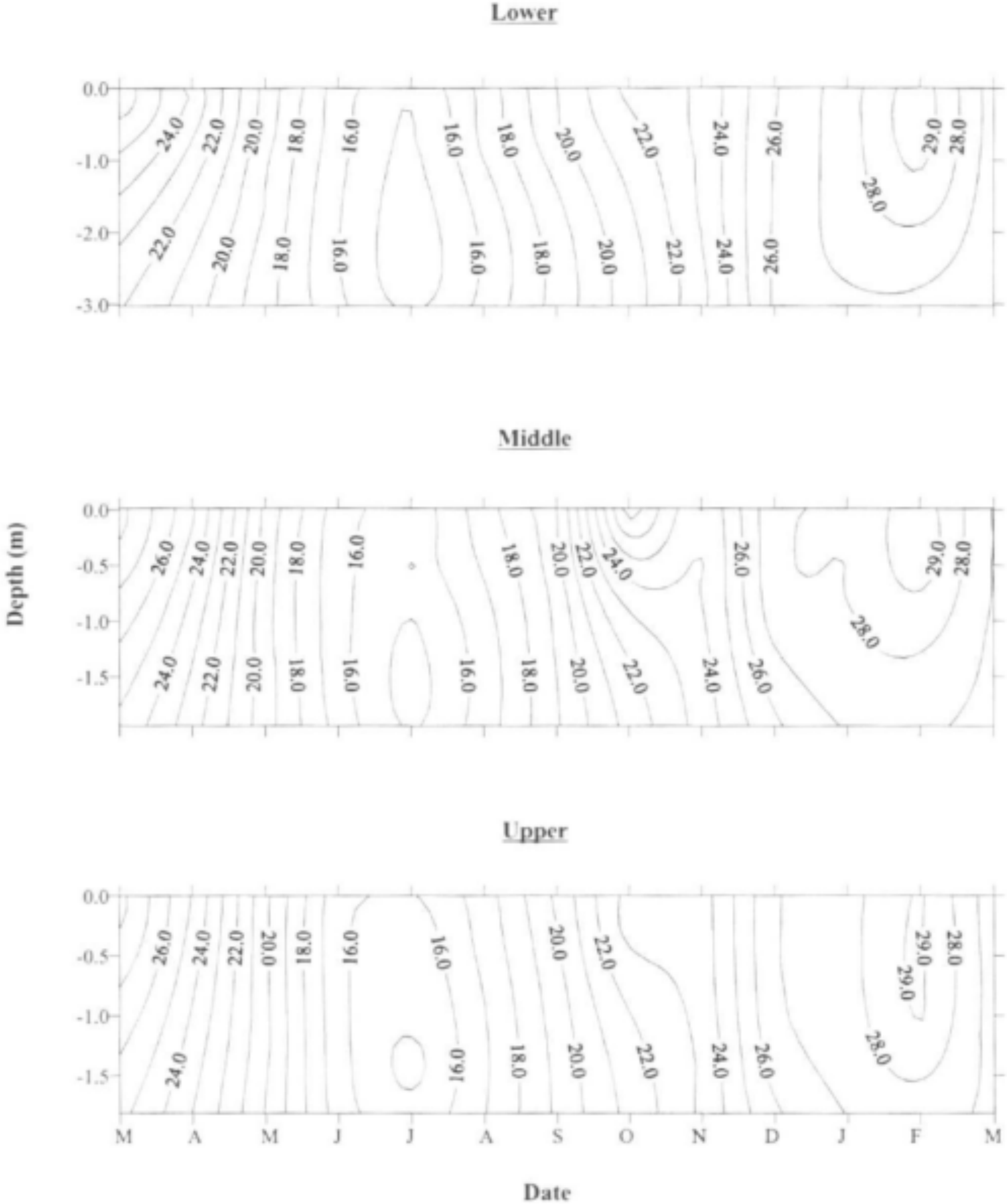


Figure 2.2. Annual variations of temperature (°C) with depth in the three reaches of the Mdloti Estuary (March 2002- March 2003, monthly measurements).

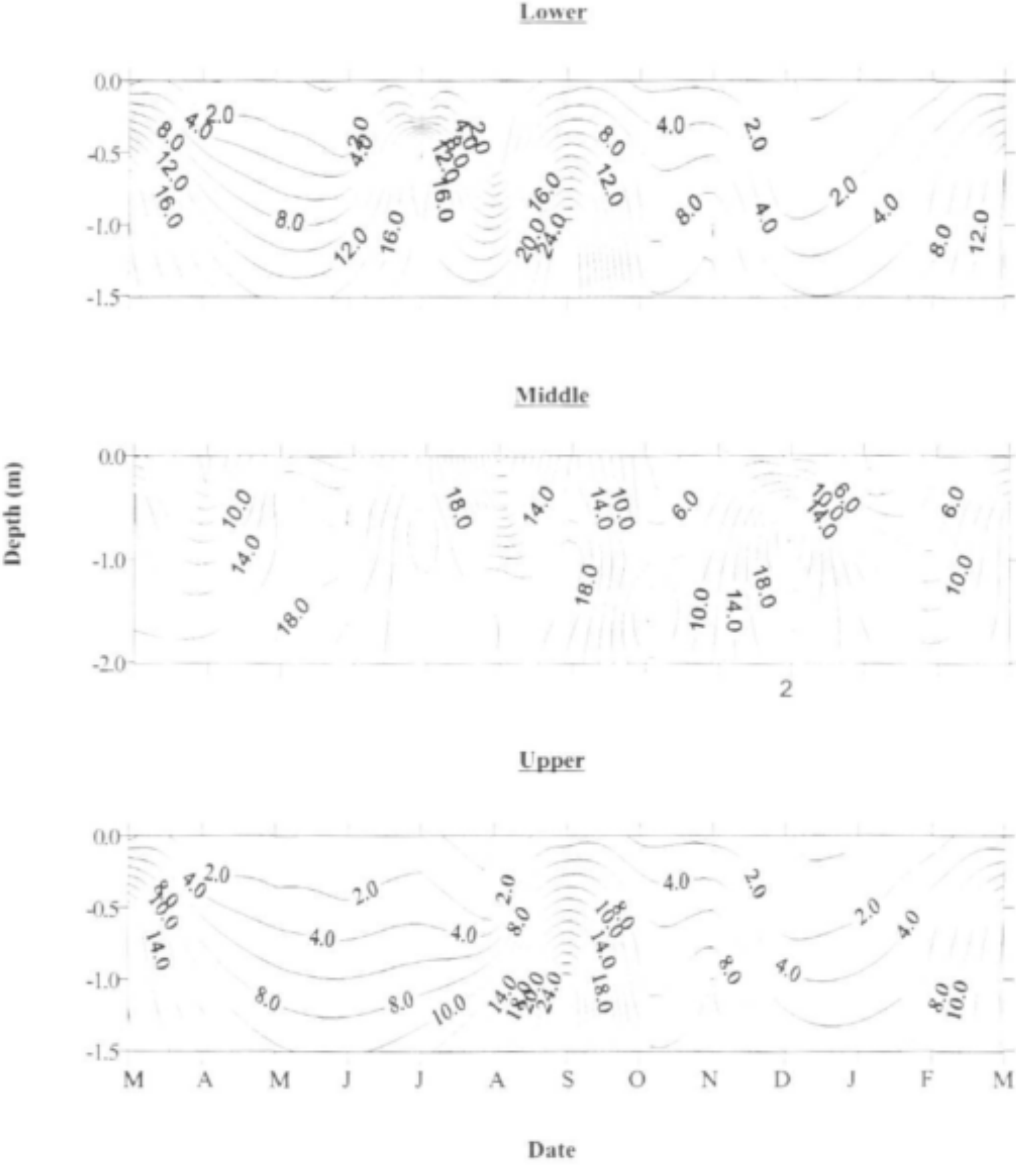


Figure 2.3 Annual variations of salinity (ppt) with depth in the three reaches of the Mhlanga Estuary (March 2002- March 2003, monthly measurements).

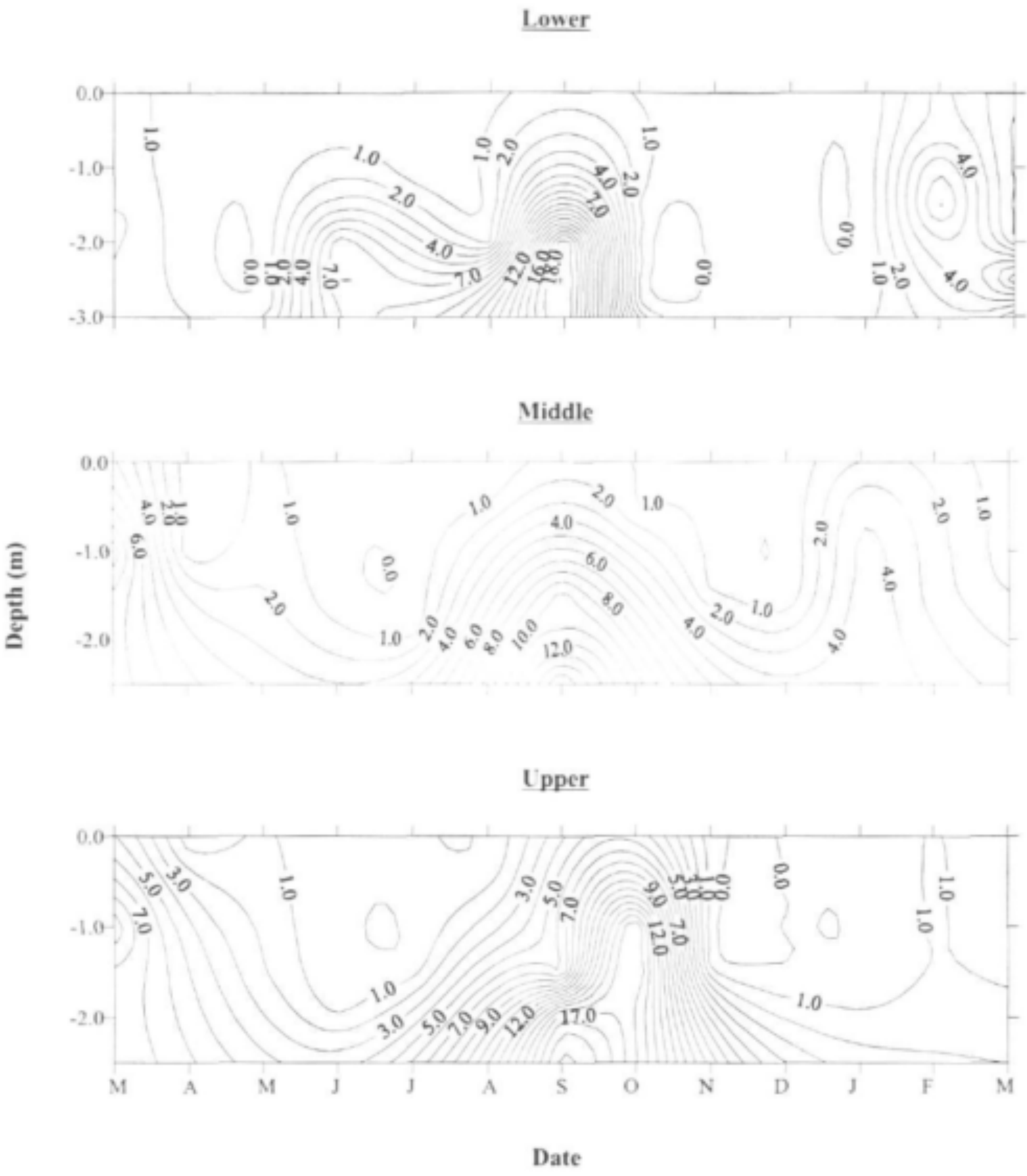


Figure 2.4 Annual variations of salinity (ppt) with depth in the three reaches of the Mdloti Estuary (March 2002- March 2003, monthly measurements).

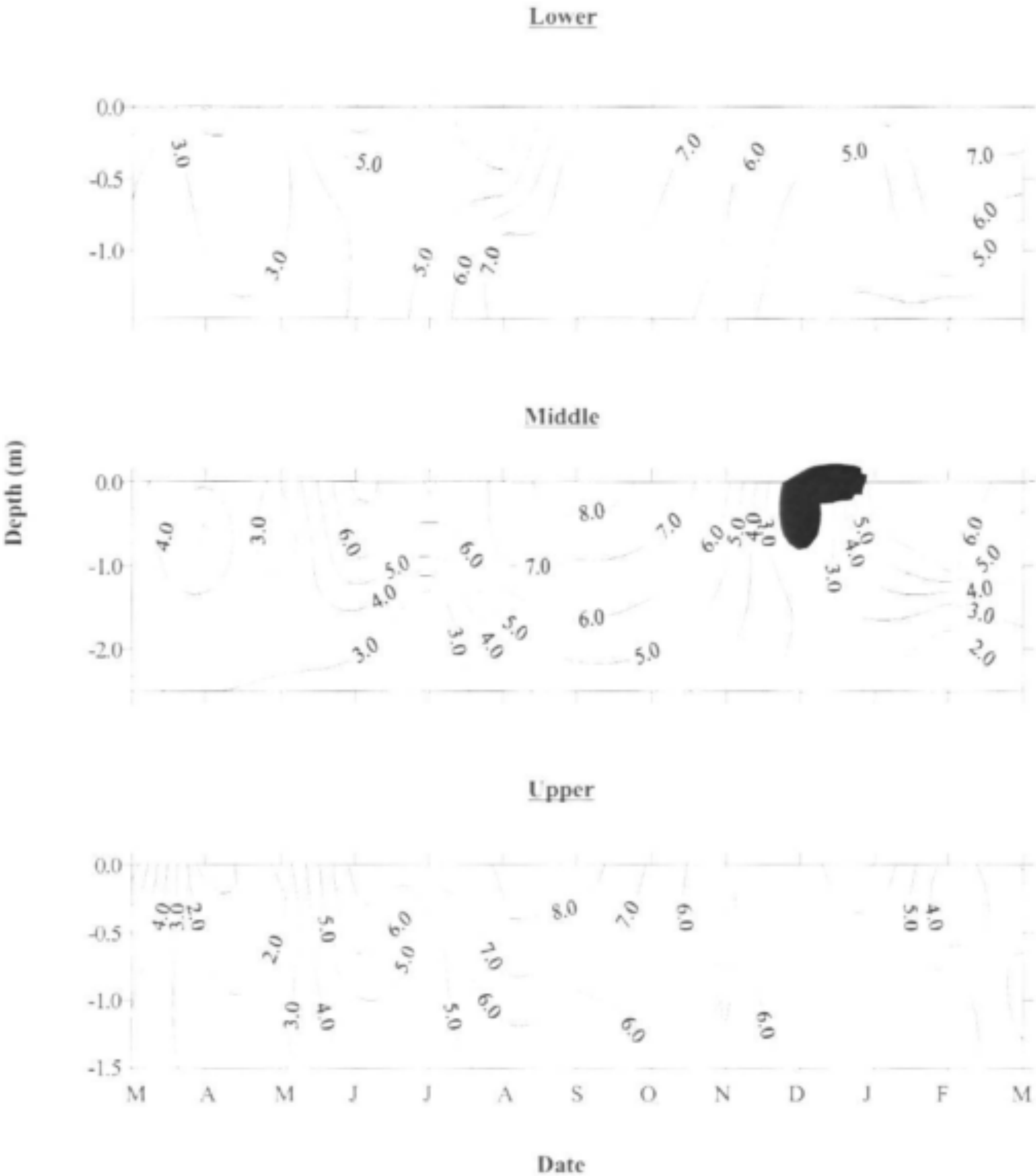


Figure 2.5 Annual variations of dissolved oxygen (mg/l) with depth in the three reaches of the Mhlanga Estuary (March 2002- March 2003, monthly measurements).

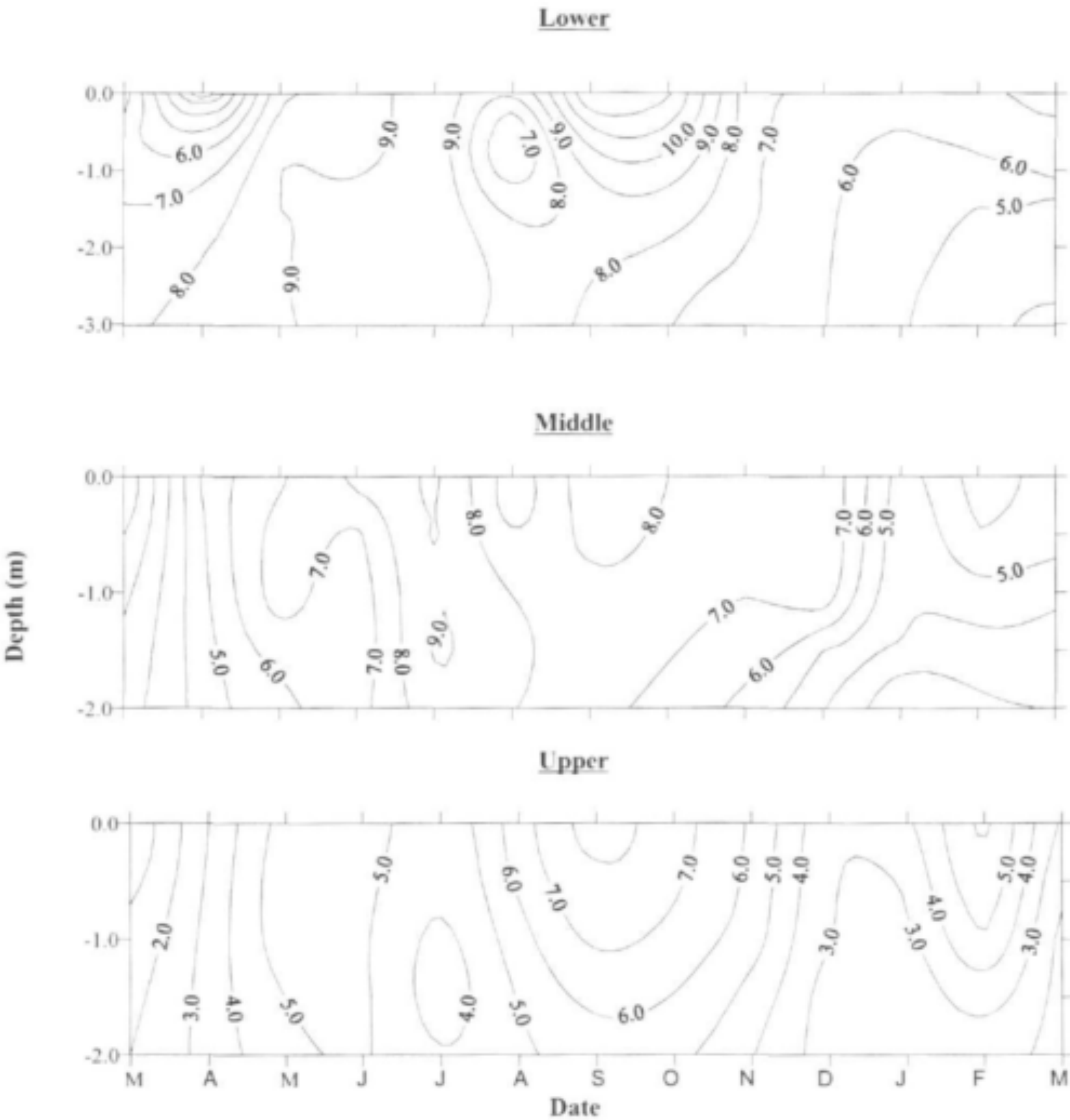


Figure 2.6 Annual variations of dissolved oxygen (mg/l) with depth in the three reaches of the Mdloti Estuary (March 2002- March 2003, monthly measurements).

2.4 Irradiance.

The maximum and minimum light extinction coefficients (K_d) at the Mhlanga were 14 m^{-1} (November 2002, middle reaches) and 0.21 m^{-1} (January 2003, middle reaches), respectively. In comparison, at the Mdloti these were 3.92 m^{-1} (September 2002, upper reaches) and 0.08 m^{-1} (August 2002, middle reaches). In all cases, the lowest values always coincided with a closed phase, while the highest values were recorded during the open phase of the Mhlanga and towards the end of the open phase (i.e. at the onset of a new closed phase) in the Mdloti. During the open phase of the Mhlanga, K_d values varied between 0.21 m^{-1} and 10.02 m^{-1}. During its closed phase, K_d values ranged from 0.84 m^{-1} to 14 m^{-1} (Fig. 2.7). In the Mdloti, the open phase exhibited K_d values between 0.08 m^{-1} and 3.92 m^{-1} and the closed phase between 1.14 m^{-1} and 3.42 m^{-1} (Fig. 2.8).

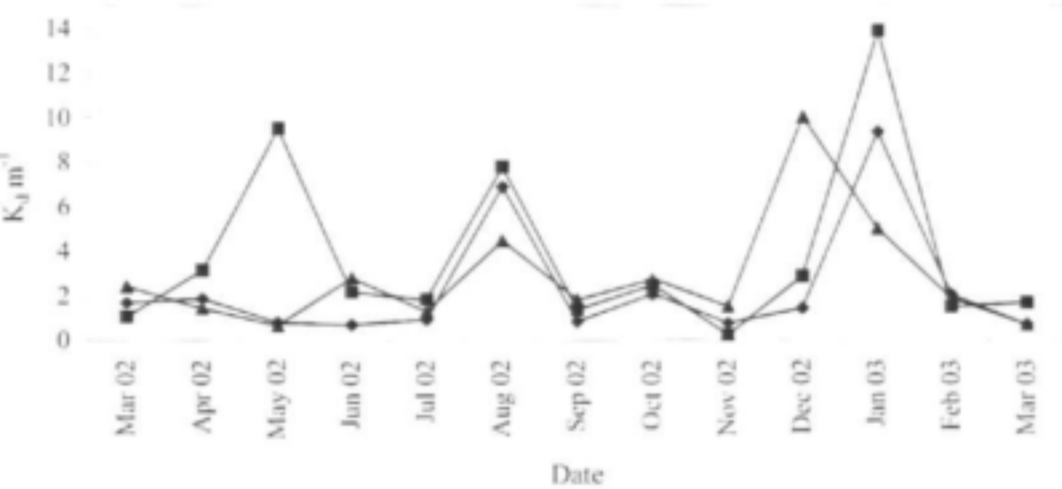


Figure 2.7 Temporal distribution of the light attenuation coefficient in the Mhlanga Estuary.

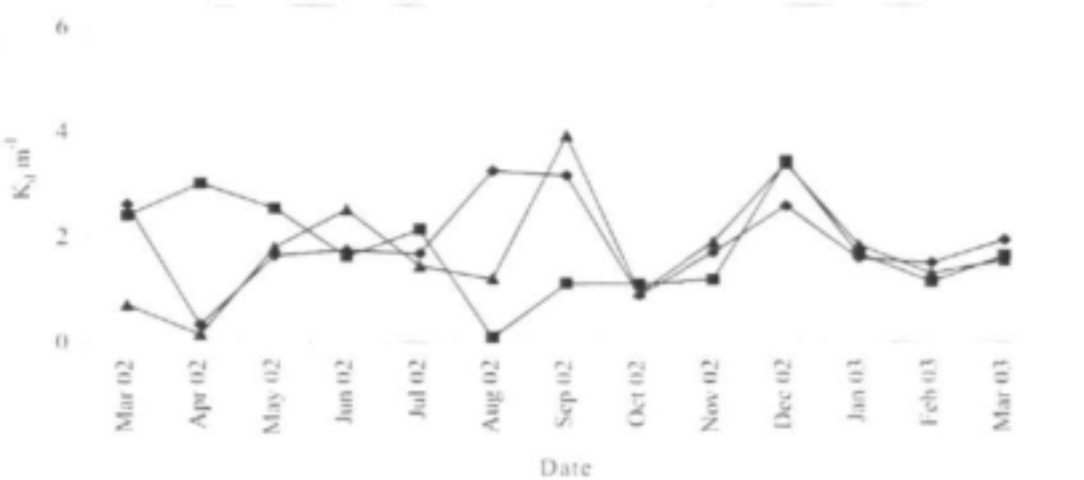


Figure 2.8 Temporal distribution of the light attenuation coefficient in the Mdloti Estuary.

2.5 Macronutrients

During the Mhlanga open phase, dissolved inorganic nitrogen concentrations (DIN) in the water column varied from 17.1 µM in August 2002 (middle reaches) to 418 µM in July 2002 (upper reaches) (Fig. 2.9). Dissolved inorganic phosphorous levels (DIP) varied between 7.81μM during January 2003 and 81.7 μM in July 2002 (Fig. 2.10). Both values were recorded in the lower reaches of the estuary. The DIN:DIP ratio during the open phase of the Mhlanga ranged between 0.50 in August 2002, i.e. well below the critical value of 16, to 27 in January 2003 (Fig. 2.9). During the closed phase, DIN levels varied between 39.3 μM (lower reaches) during October 2002 and 366 μM (middle reaches) in March 2002. DIP varied between 11.7 μM (upper reaches) during September 2002 and 73 μM (middle reaches) during March 2002. The DIN:DIP ratio varied from 1.09 in October 2002 (again well below the critical value of 16) to 24.49 during September 2002 (Fig. 2.9).

In the Mdloti Estuary, during the open phase DIN ranged from 29.6 μM (upper reaches) in September 2002 to 1236 μM in March 2002 (lower reaches) (Fig. 3.10). DIP ranged from 0.097 μM (lower reaches) during September 2002 to 13. 5 μM (middle reaches) during October 2002 (Fig. 2.10). The DIN:DIP ratio varied between 2.24 μM in October 2002 (middle reaches) and 6383 μM during March 2002 (upper reaches) (Fig. 2.10). When the estuary was closed, water column DIN varied from 0.143 μM during January and February 2003 (middle & upper reaches) to 65.7μM in July 2002 (lower reaches). DIP ranged between 0.16 μM in February 2003 (upper reaches) and 4.2 μM in December 2002 (lower reaches). The DIN:DIP ratio varied between 0.296 in January 2003 (middle reaches) and 133 in July 2002 (upper reaches) (Fig. 2.10).

In the pore water of the Mhlanga, DIN concentrations varied between 11.5 μM in November 2002 (upper reaches) to 218 μM in June 2002 (lower reaches) during the open phase and between 32.1 μM in January 2003 (middle reaches) and 212 μM in September 2002 (lower reaches) during the closed phase (Fig. 3.9). Pore DIP concentrations varied between 0.67 μM in July 2002 (upper reaches) to 45.1 μM in August 2002 (lower reaches), during the open phase, and between 1.38 μM in September 2002 (upper reaches) and 35.6 μM in October 2002 (middle reaches), during the closed phase (Fig. 2.9). In the Mdloti, pore DIN concentrations varied between 24.3 μM in March 2002 (middle reaches) to 310 μM in October 2002 (middle reaches), during the open phase, and between 2.9 μM in November 2002 (lower reaches) and 98.2 μM in December 2002 and in January 2003 (middle reaches), during the closed phase (Fig. 2.10). Pore DIP concentrations varied between 0.13 μM in March 2002 (middle reaches) and 3.3 μM in August 2002 (upper reaches), during the open phase, and between 0.1 μM in May 2002 (lower reaches) and 17.3 μM in December 2002 (middle reaches), during the closed phase (Fig. 2.10).

Overall, pore DIN:DIP ratios were much lower in the Mhlanga than in the Mdloti Estuary. In the Mhlanga, ratios ranged from a minimum of 1.6 (lower reaches, March 2002 & January 2003) during the closed phase to a maximum of 104 (upper reaches, July 2002) during the open phase (Fig. 2.9). In the Mdloti, pore DIN:DIP ratios were usually below 16 in all reaches during the closed phase. During the open phase, values ranged from 8 (upper, September 2002) to 490 (middle, September 2002). This suggests a potential nitrogen limitation of productivity within the sediment of both estuaries not only during part of their closed phase, but occasionally also during the open phase (Fig. 2.10).

In summary, three main patterns appear to emerge from the nutrient availability in the two estuaries.

1) Concentrations of both DIN & DIP are normally much higher (2-5 times) in the Mhlanga than in the Mdloti. The extremely high values observed consistently at the Mhlanga can only be explained in terms of the high volume of treated sewage waters (20 $Ml.d^{-1}$) that this estuary receives regularly. By all means the estuary can, therefore, be regarded as artificially eutrophic in its current state.
2) There are so far no significant differences in DIN or DIP concentrations between the open and the closed phase of the Mhlanga. But in contrast to this, the differences between the two phases are very marked in the Mdloti, with values normally 1.5 – 10 times higher during the open than during the closed phase of the estuary. An exception to this is observed in the pore DIP, which seems substantially higher during the closed than the open phase of the Mdloti: an indication of the importance of sediments in the regeneration of inorganic phosphorous during periods of low inflow from the river.
3) Both DIN & DIP are consistently higher in the water-column than within the sediment (pore water) at the Mhlanga, but only so during the open phase at the Mdloti. Here, the situation appears to be reversed during the closed phase, with higher levels recorded in pore waters as a result of mineralization processes within the sediment becoming more important that freshwater inflow during this phase.

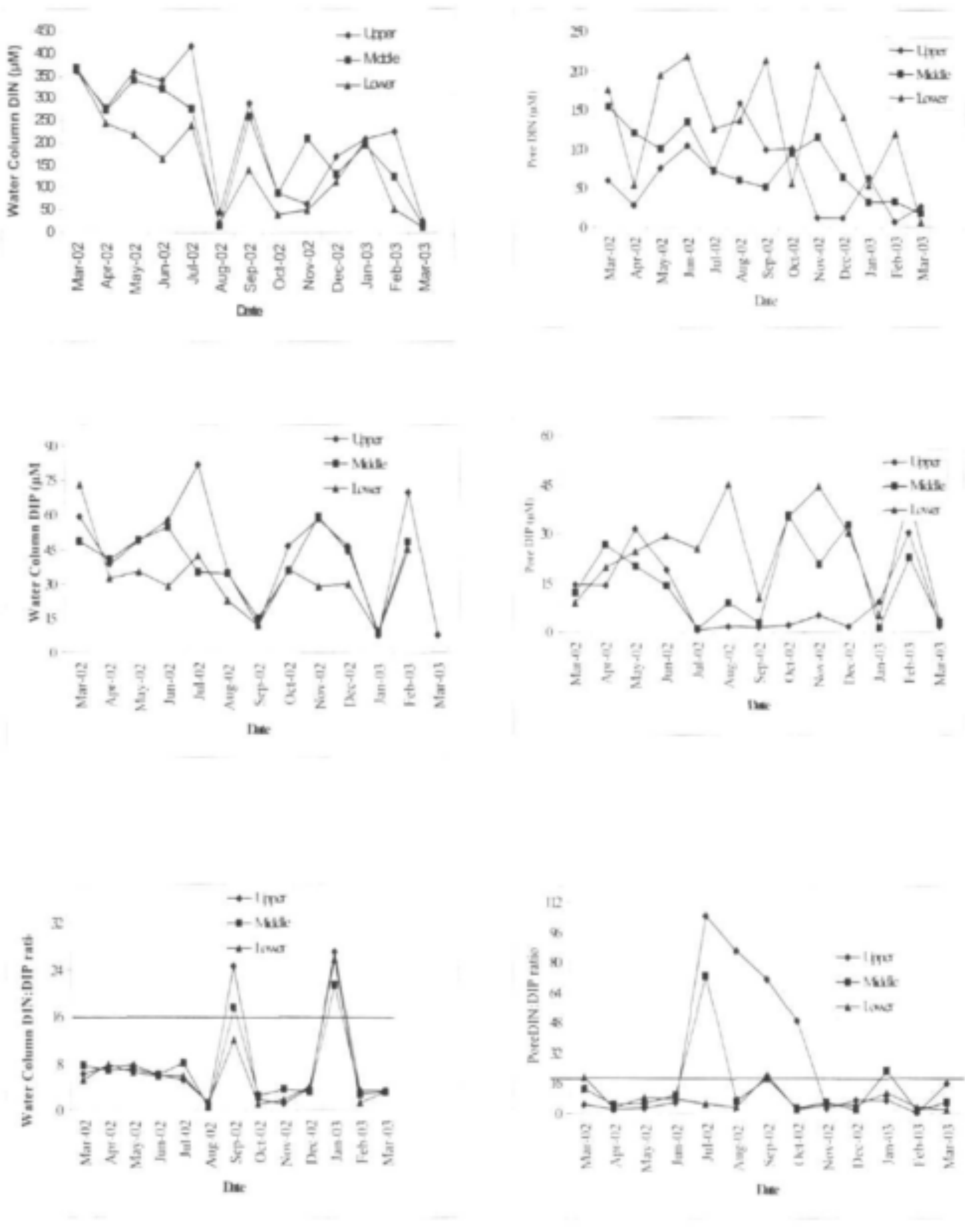


Figure 2.9 Dissolved inorganic nitrogen and phosphorous in the water column and pore water of the Mhlanga Estuary during March 2002 – March 2003.

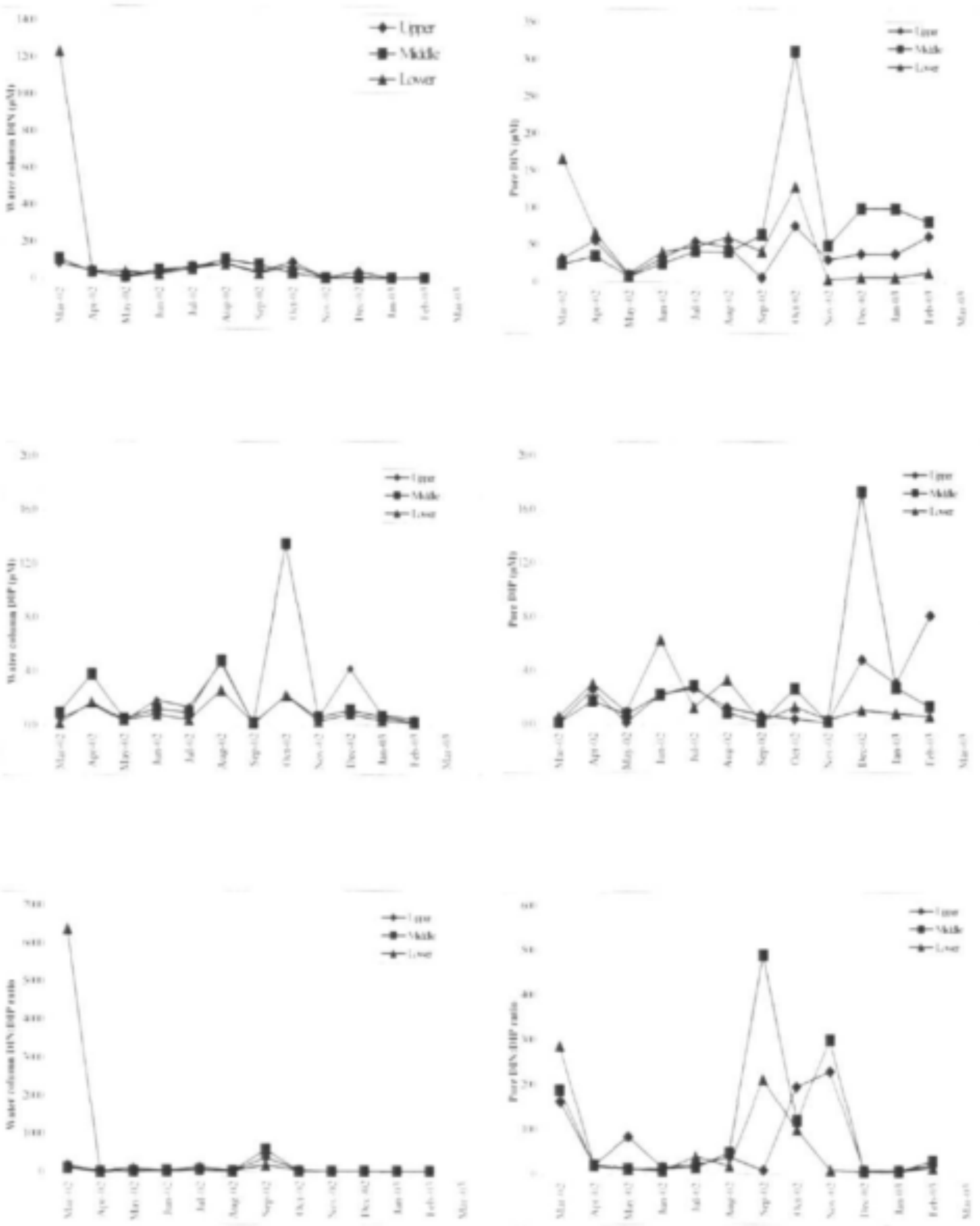


Figure 2.10 Dissolved inorganic nitrogen and phosphorous in the water column and pore water of the Mdloti Estuary during March 2002 – March 2003.

2.6 Sediment structure and organic content

The broad pattern of sediment composition and distribution in the Mhlanga in March, June and September 2002 (Figure 2.11) was of relatively coarse sand (0.5 – 1 mm) in the upper and lower reaches and medium (0.25 – 0.5 mm) in the mid reaches. The December 2002 and March 2003 samples indicated a trend towards slightly coarser conditions and a greater similarity at all stations (Figure 2.12). In the Mdloti (Figure 2.13, 2.14) there was a consistent gradient from medium (0.25 – 0.5 mm) sand at the lower stations to coarse (0.5 – 1 mm) in the mid and upper reaches. Samples showed little variation in particle size composition in any one area over the survey period.

Organic content was low, peaking at 4% in the mid-reaches of the Mdloti (Table 2.1). Sixty eight (76%) of the 89 samples taken registered less than 1 %. This was to be expected on the basis of the generally sandy nature of both systems.

Table 2.1 Percentage organic content of sediments in the Mhlanga and Mdloti estuaries. L: lower reaches, M: middle reaches, U: upper reaches.

SEDIMENT ORGANIC CONTENT (%)										
	MHLANGA					MDLOTI				
	2002				2003	2002				2003
Site	**Mar**	**Jun**	**Sep**	**Dec**	**Mar**	**Mar**	**Jun**	**Sep**	**Dec**	**Mar**
L1	3.4	0.9	1.4	3.6	0.2	1.9	0.3	0.6	0.5	0.5
L2	2.7	0.8	0.7	3.7	0.2	0.4	0.4	0.6	0.7	0.4
L3	2.4	0.7	0.9	3.7	0.2	0.3	0.4	0.4	0.4	0.4
M1	4.7	1.0	0.5	0.4	0.2	0.4	0.2	0.3	0.3	0.4
M2	1.1	1.6	0.4	0.3	0.1	0.3	0.2	0.3	0.3	0.4
M3	0.7	3.4	0.5	0.3	0.1	0.5	0.2	0.4	0.4	0.4
U1	0.5	0.2	1.0	0.4	0.5	0.3	0.3	0.3	0.5	0.3
U2	1.4	0.2	1.6	3.3	0.5	0.3	0.5	0.5	0.5	0.3
U3	0.4	0.3	2.2	0.5	n/d	0.3	2.6	1.0	0.4	0.3

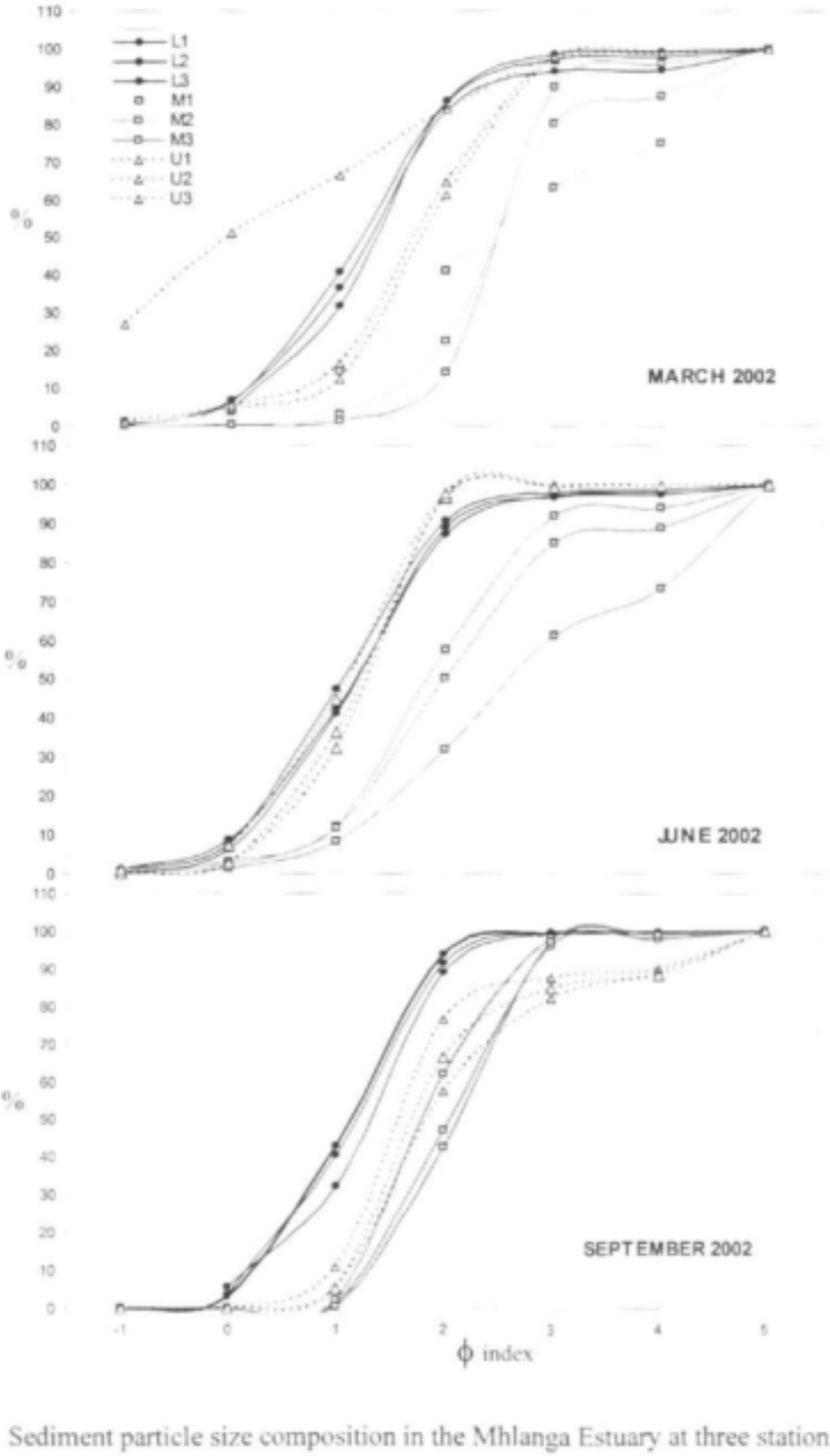


Figure 2.11 Sediment particle size composition in the Mhlanga Estuary at three stations (L = Lower, M= Mid, U=Upper) during March, June and September 2002

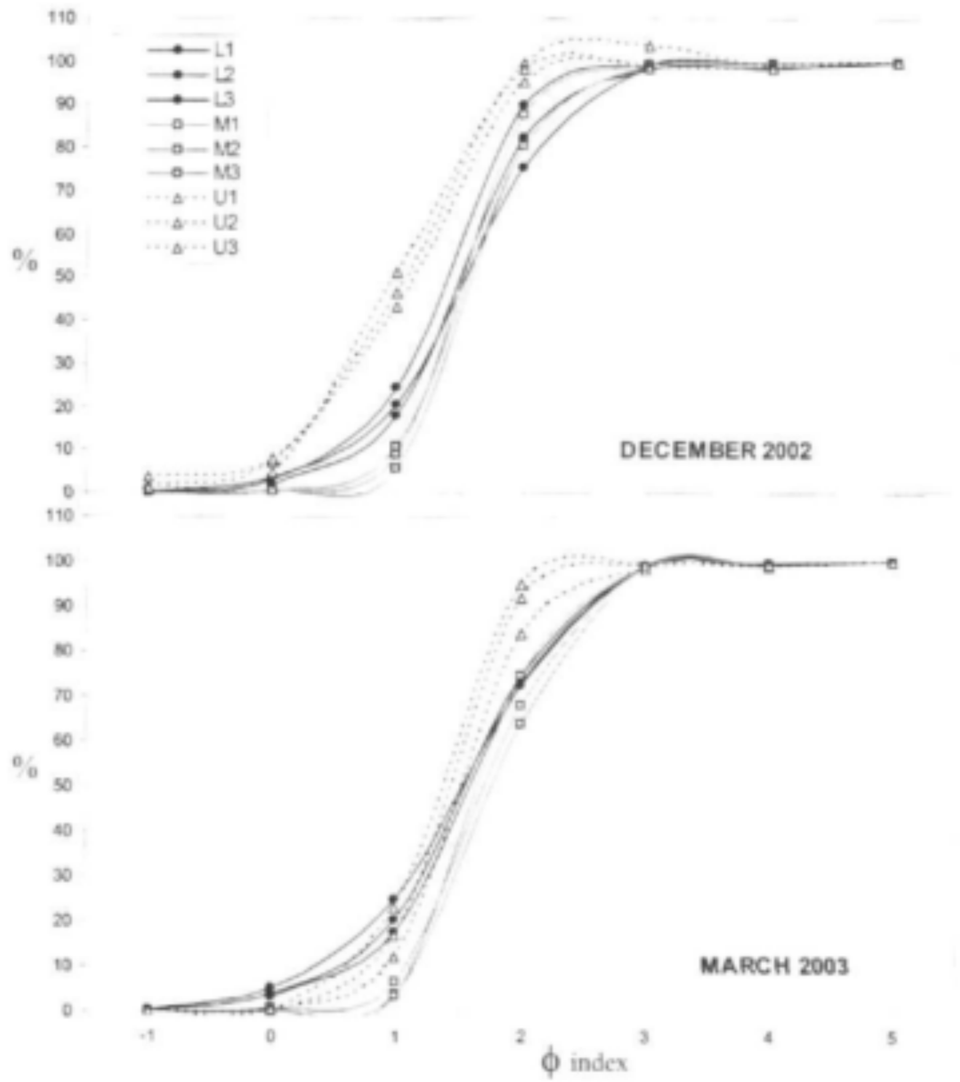


Figure 2.12 Sediment particle size composition in the Mhlanga Estuary at three stations (L = Lower, M= Mid, U=Upper) during December 2002 and March 2003

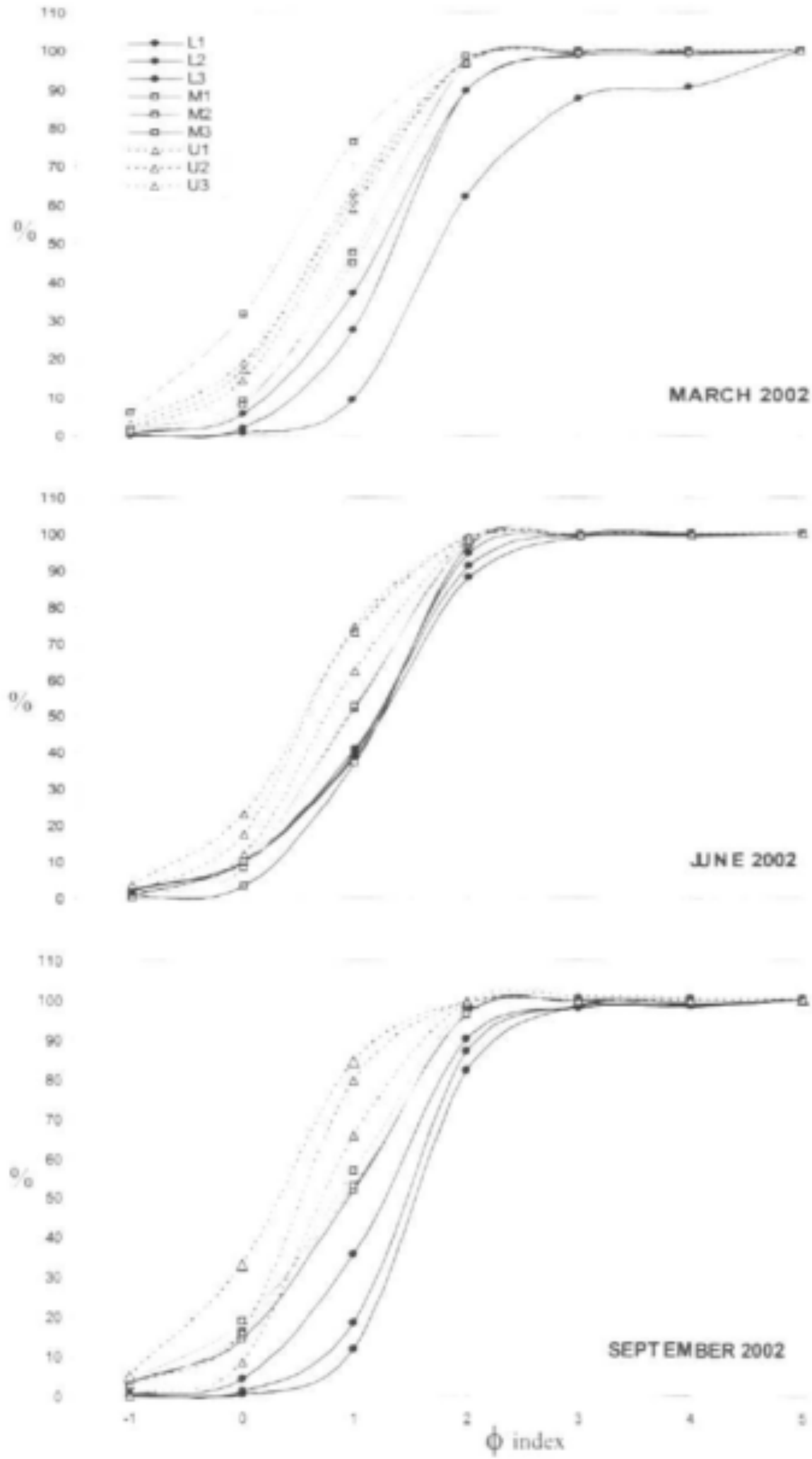


Figure 2.13 : Sediment particle size composition in the Mdloti Estuary at three stations (L = Lower, M= Mid, U=Upper) during March, June and September 2002

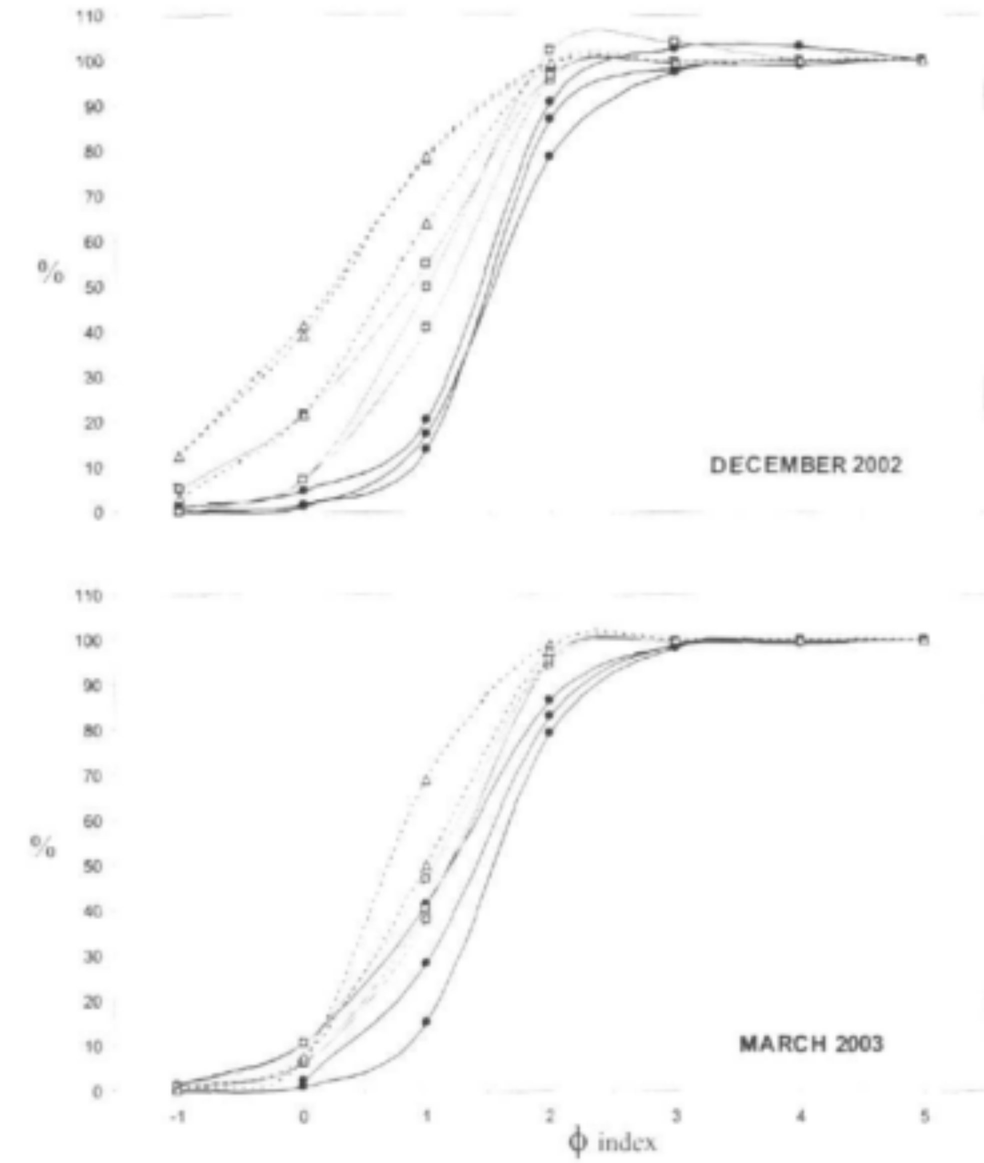


Figure 2.14 Sediment particle size composition in the Mdloti Estuary at three stations (L = Lower, M= Mid, U=Upper) during December 2002 and March 2003

3. PHYTOPLANKTON

The lowest phytoplankton chl-*a* concentrations occurred during the open phase in both the Mhlanga and the Mdloti estuaries (Figs 3.1 & 3.2). The minimum concentrations were 0.869 mg m^{-3} in the Mdloti (September 2002) and 0.732 mg m^{-3} in the Mhlanga (June 2002). These values were recorded at the surface of the upper and middle reaches, respectively. The highest phytoplankton chl-*a* concentrations occurred at the surface of the water-column during the closed phase in both estuaries. In the Mhlanga, 303 mg chl-*a* m^{-3} were recorded in October 2002 in the lower reaches (Fig. 3.1). In the Mdloti, the highest phytoplankton chl-*a* concentration recorded was 111 mg m^{-3}, in December 2002 in its middle reaches (Fig. 3.2). Dense phytoplankton blooms (> 100 mg chl-*a* m^{-3}, Adams and Bate 1999) also occurred during October 2002 and February 2003 at the surface of the water-column in the middle and upper reaches of the Mhlanga. For the entire period of the survey, high mean chl-*a* concentrations of 22.8 ± 42.4 mg m^{-3}, were recorded for the Mhlanga. The mean chl-*a* concentration for the Mdloti was 19.0 ± 22.9 mg m^{-3}.

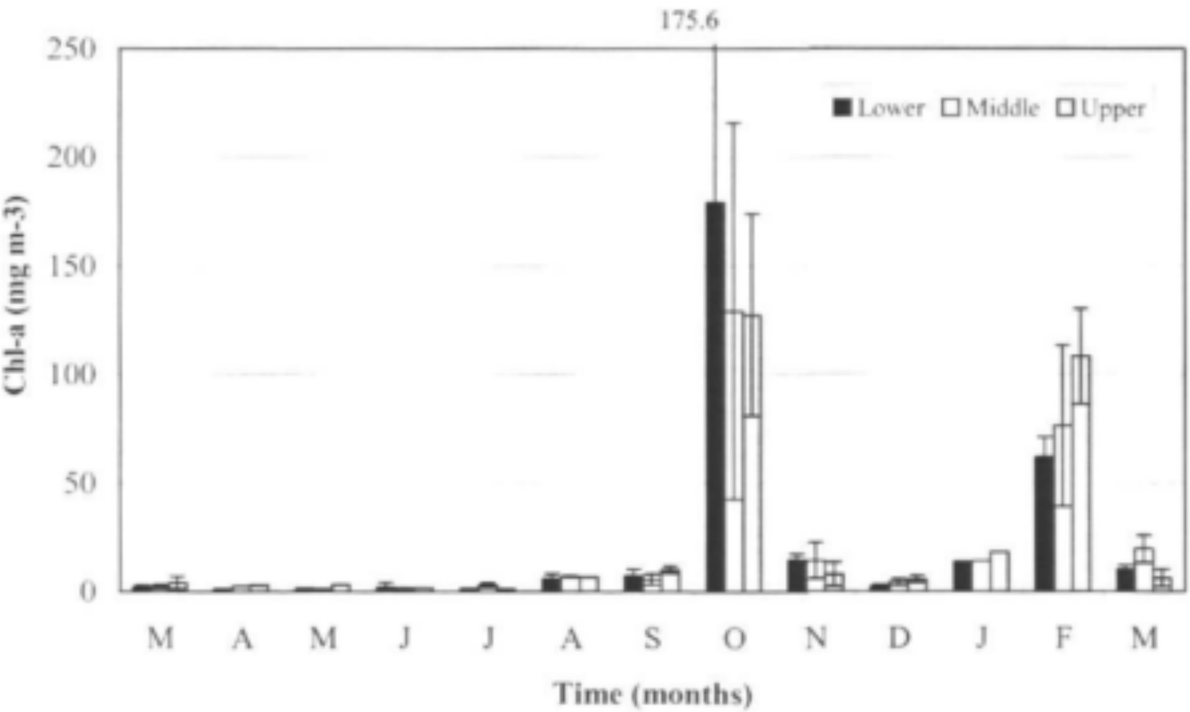


Figure 3.1 Spatio-temporal variation in phytoplankton chl-*a* biomass (mean ± SD) in the Mhlanga Estuary during the study period.

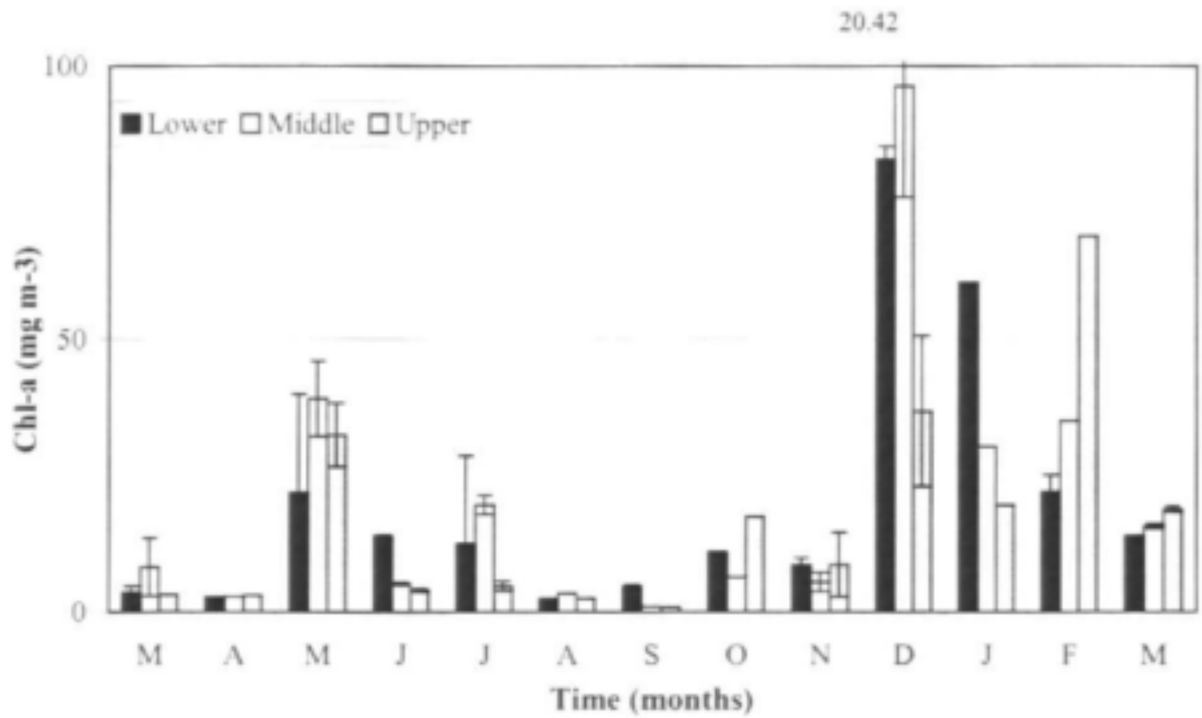


Figure 3.2 Spatio-temporal variation in phytoplankton chl-*a* biomass (mean ± SD) in the Mdloti Estuary during the study period.

Prior to these measurements, the Mhlanga had been closed for 9 days and the Mdloti for 12 days. The peak value for the Mhlanga (303 mg chl-*a* m^{-3}) is the highest ever recorded in any of the South African estuaries for which information is currently available (Adams *et al.* 1999, Nozais *et al.* 2001).

The minima and maxima of the different phytoplankton size classes, and their mean contribution to phytoplankton biomass during the survey period, are summarized in Table 3.1. The nanophytoplankton fraction virtually dominated phytoplankton biomass during the entire study period, during both open and closed phases of the two estuaries. This was followed by the picophytoplankton (< 2μm) and then the microphytoplankton (> 20 μm). The only exceptions to this were: a) in July 2002, in the surface waters of the middle reaches of the Mdloti, when total biomass was dominated by picophytoplankton (72%); and b) in January 2003, in the surface waters of the middle and upper reaches of the Mhlanga, when total biomass was dominated by microphytoplankton (53%).

Table 3.1 Range of micro-, nano- and picophytoplankton concentrations (mg m^{-3}) and their mean percentage contribution to total phytoplankton biomass in the Mdloti and the Mhlanga estuaries. Superscripts L, M, U = lower, middle and upper reaches, respectively.

	MDLOTI			**MHLANGA**		
Size class	**Min**	**Max**	**% total chl-*a***	**Min**	**Max**	**% total chl-*a***
Micro-	0.087^{L}	6.64^{M}	8.07	0.029^{L}	9.6^{U}	11.9
Nano-	0.560^{M}	86.4^{M}	79.3	0.847^{M}	171^{L}	76.7
Pico-	0.047^{U}	7.47^{M}	12.6	0.051^{L}	15.5^{U}	15.9

When the maximum phytoplankton biomass of 303 mg chl-*a* m^{-3} was recorded in the Mhlanga Estuary in October 2002, the nanophytoplankton comprised 96% of the total phytoplankton biomass, while the microphytoplankton and picophytoplankton contributed 1% and 3%, respectively. When the minimum phytoplankton biomass (0.732 mg m^{-3}) was recorded in the same estuary in June 2002, the contribution of micro-, nano- and picophytoplankton were 5%, 84%, and 11%, respectively. At the time of maximum phytoplankton biomass in the Mdloti Estuary (111 mg m^{-3} in December 2002), the nanophytoplankton comprised 88% of the total phytoplankton biomass, while the microphytoplankton and picophytoplankton contributed 3% and 9%, respectively. However, the contribution of micro-, nano- and picophytoplankton were 11%, 83%, and 6% respectively, when the minimum phytoplankton biomass (0.869 mg m^{-3}) was recorded in this estuary in September 2002.

The concentration of phytoplankton chl-*a* at the surface and bottom of the water-column did not differ significantly within each estuary [$U = 481$ (Mdloti), $U = 372$ (Mhlanga), $p > 0.05$]. Thus, the phytoplankton chl-*a* concentrations at the surface and bottom were averaged to provide the total phytoplankton chl-*a* concentration in the water-column. A 1-way ANOVA revealed no significant differences in water-column phytoplankton chl-*a* concentrations between the Mdloti and the Mhlanga estuaries ($F_{1, 76} = 0.886$, $p > 0.05$). Variations in water-column phytoplankton chl-*a* concentrations between the two estuaries were also not significant when their closed ($U = 169$, $p > 0.05$) and open phases ($U = 170$, $p > 0.05$) were considered separately. Within each estuary, there were significant temporal ($F_{12, 26} = p < 0.001$) but non-significant spatial ($F_{2, 36} = p > 0.05$) differences in water-column phytoplankton chl-*a* concentrations. In particular, water-column phytoplankton chl-*a* concentrations differed significantly between the open and closed mouth conditions of each estuary [$U = 26$, $p < 0.001$ (Mdloti), $U = 80$, $p < 0.01$ (Mhlanga)].

Results of the Spearman correlation analysis between rainfall and average phytoplankton biomass showed a significant inverse relationship during the closed phase of the Mhlanga ($r = -0.51$, $p < 0.05$). A similar relationship was found between rainfall and average phytoplankton biomass at the Mdloti. However, in this case the relationship was significant only for the open phase ($r = -0.64$, $p < 0.01$). With the exception of the open state of the Mhlanga Estuary, these analyses contradict the hypothesis that increased freshwater flow causes an increase in phytoplankton biomass but confirms the findings of Nozais *et al* (2001) in the Mdloti Estuary.

Spearman correlation analyses between K_d and average phytoplankton biomass were only significant during the closed phase of the Mdloti estuary ($r = 0.4$, $p < 0.05$). It is possible that the high phytoplankton biomass during this phase may have increased the attenuation of light to such an extent that by self-shading the phytoplankton cells limited their own population growth (Kirk 1983). This result is in contrast to the finding of Nozais *et al.* (2001) and, therefore, does not confirm the hypothesis that a decrease in the attenuation coefficient for irradiance is directly responsible for an increase in phytoplankton biomass (cf. Nozais *et al.* 2001).

The Spearman correlation analysis between phytoplankton biomass and DIN showed a significant inverse relationship during both closed ($r = -0.55$, $p < 0.01$) and open phase of the Mhlanga ($r = -0.64$, $p < 0.001$). DIP on the other hand, exhibited a negative correlation ($r = -0.48$, $p < 0.01$) with biomass only during the open phase. At the Mdloti, again DIN was negatively correlated with phytoplankton chl-a ($r = -0.52$, $p < 0.001$) for closed and open phases combined. The same relationship was also obtained between biomass and DIN:DIP ($r = -0.48$, $p < 0.001$;). It is likely that all these inverse relationships between macronutrient levels and microalgae biomass simply reflect nutrient uptake by microalgae (see also Nozais *et al.* 2001).

Correlation analyses between average phytoplankton and microphytobenthic biomass showed a significant positive relationship only for the combined phases of the Mdloti ($r = 0.54$, $p < 0.001$). Phytoplankton biomass was positively correlated with zooplankton abundance (i.e. the major potential grazers) during the closed phase [$r = 0.79$, $p < 0.001$ (sled zooplankton); $r = 0.77$, $p < 0.001$ (WP2 net zooplankton)] of the Mhlanga Estuary. During the closed phase of the Mdloti, a negative but non-significant relationship existed between phytoplankton and zooplankton. No significant correlations were also obtained between phytoplankton and their main grazers during the open phase of both estuaries. The greatest contributor to phytoplankton biomass in this study was the nanophytoplankton, which is the preferential food source of most of the zooplankton of temporarily-open estuaries (Kibirige & Perissinotto 2003). The inverse interaction between the zooplankton and average phytoplankton biomass during the closed phase of the Mdloti may be explained by the prolonged period of mouth closure experienced by this estuary between May and July 2002 (67 days). This finding, therefore, does not provide conclusive evidence for the hypothesis that grazing pressure reduces phytoplankton biomass 3 weeks after closure. The process may in fact involve a longer time-scale.

4. MICROPHYTOBENTHOS

Microphytobenthic chl-*a* concentrations ranged from 1.3 to 391 mg chl-*a* m^{-2} in the Mdloti (Fig. 4.1) and from 1.7 to 313 mg chl-*a* m^{-2} in the Mhlanga (Fig. 4.2). The maximum values attained during this study, with 391 and 313 mg chl-*a* m^{-2} at the Mdloti and the Mhlanga respectively, are consistent with those already reported for other South African TOCEs (Adams *et al.* 1999; Perissinotto *et al.*2000; Nozais *et al.* 2001; Perissinotto *et al.* 2002). They are amongst the highest reported in the literature (de Jonge & Colijn 1994; Brotas *et al.* 1995; McIntyre *et al* 1996; Light & Beardall 1998; Cahoon *et al*.1999).

At the Mdloti Estuary, microphytobenthic chl-*a* concentrations varied considerably, from 1.33 mg chl *a*-m^{-2} (lower reaches, September 2002) to 131 mg chl *a*-m^{-2} (upper reaches, September 2002) during the open phase, and from 18 mg chl *a*-m^{-2} (lower reaches, February 2003) to 391 mg chl *a*-m^{-2} (upper reaches December 2002) during the closed phase. During the Mhlanga open phase, microphytobenthic chl *a* concentrations ranged from 7 mg m^{-2} (lower reaches, December 2002) to 313 mg m^{-2} (lower reaches, June 2002). During its closed phase, microphytobenthic chl *a* concentrations ranged from 1.7 mg m^{-2} (lower reaches, September 2002) to 267 mg m^{-2} (middle reaches, February 2003). Along the Mdloti, the mean microphytobenthic biomass was lowest in the lower reaches (43.8 mg chl-*a* $m^{-2} \pm 66.3$ SD) and highest in the upper reaches of the estuary (96.1 mg chl-*a* $m^{-2} \pm 114$ SD). The middle reaches exhibited a mean microphytobenthic biomass of 84 ± 73 (SD) mg chl-*a* m^{-2} (Fig. 4.1). Similarly, at the Mhlanga, mean microphytobenthic biomass was lowest at the lower reaches of the estuary, with 68.1 ± 93.9 (SD) mg chl-*a* m^{-2}. The highest, however, was observed in the middle reaches with 105.2 ± 113.7 (SD) mg chl-*a* m^{-2}, while the upper reaches exhibited a mean microphytobenthic biomass of 96.4 ± 86 (SD) mg chl-*a* m^{-2} (Fig. 4.2).

Results of a Spearman rank correlation analysis on the Mhlanga data set showed a significant relationship between total microphytobenthic biomass, pore DIN concentration ($r = -0.43$, $p < 0.01$) and temperature ($r = 0.42$, $p < 0.01$). The Mdloti data set showed, however, significant correlations between microphytobenthic biomass and salinity ($r = -0.4$, $p < 0.01$). For the duration of this study, positive correlations between microphytobenthic biomass and zooplankton biomass occurred in both the Mdloti ($r = 0.40$, $p < 0.001$) and the Mhlanga ($r = 0.33$, $p < 0.05$).

The trend in TOCEs thus far seems to suggest an association of the highest sediment chl-*a* concentrations with the closed phase of the estuary (Nozais *et al.* 2001; Perissinotto *et al.* 2002; Perissinotto *et al.* 2003). A 2-way ANOVA, performed on $\log_{10}$-transformed chl-*a* values, revealed significant differences between the open and closed phase ($F = 160$, $p < 0.0001$), as well as between reaches ($F = 17.8$, $p < 0.0001$) at the Mdloti Estuary. Here, higher microphytobenthic biomass values generally coincided with the closed phase (18 to 391 mg chl-*a* m^{-2}) while the lowest biomass values typically occurred during the open phase (1.33 to 131 mg chl-*a* m^{-2}). In the Mhlanga, however, there were no significant differences between the open and the closed phase ($F = 1.40$, $p > 0.05$), but significant differences were observed between reaches ($F = 5.91$, $p < 0.05$) (Fig. 4.2). These results seem to indicate that this estuary is not closed long enough to establish a new benthic community between two subsequent breaching events. Unlike typical TOCEs, the higher biomass values at the Mhlanga Estuary were not always associated with the closed mouth state. For instance, high concentrations were also observed during the open mouth conditions of June 2002 and January 2003, particularly at the lower and middle reaches and at times when the estuary had been open for approximately a week. The high microphytobenthic biomass observed during the open phase suggest that although the biomass may decline temporarily during the transition from the closed to open phase, *in situ* growth and advective transport of "buried" benthic microalgae may be sufficient to re-supply the community within a relatively short period of time (Ray 1989; Lucas *et al.* 2000; Mundree *et al.* 2003).

A t-test for independent variables performed on $\log_{10}$-transformed data showed that the microphytobenthic chl-*a* concentration between the two estuaries did not differ significantly for the entire data set ($t = -1.73$, $p > 0.05$). However, when separated into open and closed phases, differences in microphytobenthic chl-*a* concentrations were significant between the two estuaries during their open phase ($t = -6.56$, $p < 0.0001$), but not during their closed phase ($t = 1.74$, $p > 0.05$).

Thus, the pattern seems to indicate that in the Mdloti, high values are generally associated with the closed phase, while in the Mhlanga the higher values are associated with the partly open phase. During the closed phase of the Mdloti, the dominance of favourable light conditions (K_d between 1.14 to 3.42 m^{-1}) increases the availability of light to the sediment surface and its penetration into deeper layers. Furthermore, during this period, microphytobenthic biomass was positively correlated with rainfall ($r = 0.52$, $p < 0.01$) and nutrients, particularly pore DIP concentration ($r = 0.49$, $p < 0.05$). The Mhlanga system, however, appears to be much more complicated primarily because the estuary has breached so often during the study period, with consequent regular interruptions of its closed phase.

Throughout the study, the mean microphytobenthic chl-*a* concentrations generally exceeded water column phytoplankton concentrations, in both estuaries. Studies previously carried out at the Mdloti, Mpenjati and Nyara estuaries have reported microalgal biomass in the sediment exceeding by one to three orders of magnitude that in the water column (Nozais *et al.* 2001; Perissinotto *et al.* 2002). These results have been attributed to the prevailing conditions in these systems, which include low turbidity, more stable sediment and a large nutrient pool available within the substratum (Adams and Bate 1999).

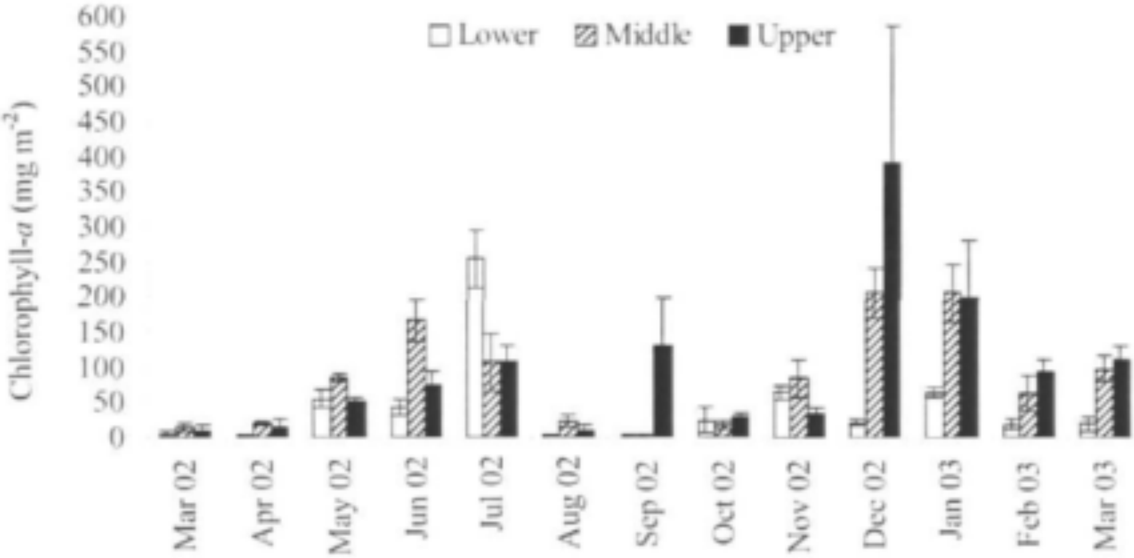


Figure 4.1 Temporal and spatial distribution of microphytobenthic biomass (mean ± SD) in the Mdloti Estuary

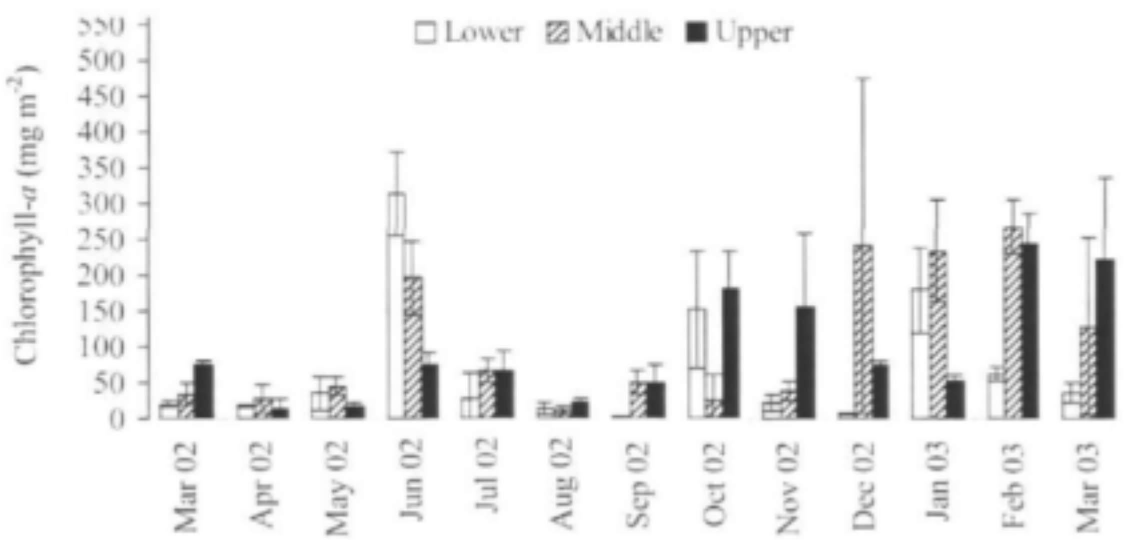


Figure 4.2 Temporal and spatial distribution of microphytobenthic biomass (mean ± SD) in the Mhlanga Estuary

5. Zooplankton

Total zooplankton abundances were higher in the Mdloti than in the Mhlanga, with values in the former ranging from 1.02×10^2 to 1.83×10^6 (mean $2.4 \times 10^5 \pm 9.0 \times 10^5$ ind.m^{-3} SD) and in the latter from 1.08×10^2 to 9.23×10^4 (mean $1.5 \times 10^4 \pm 3.37 \times 10^4$ ind.m^{-3} SD). At the Mhlanga, total zooplankton abundance values ranged from 2.98×10^2 to 9.23×10^4 ind.m^{-3} (mean $2.67 \times 10^4 \pm 4.68 \times 10^4$ ind.m^{-3} SD) during the closed phase, and from 1.08×10^2 to 5.06×10^4ind.m^{-3} (mean $4.46 \times 10^4 \pm 6.65 \times 10^4$ ind.m^{-3} SD) during the open phase (Fig. 5.1). At the Mdloti, total zooplankton numbers varied between 3.61×10^4 and 1.83×10^6 ind.m^{-3} (mean $3.6 \times 10^5 \pm 1.09 \times 10^6$ ind.m^{-3} SD) during the closed phase, and between 1.02×10^2 and 1.3×10^3 (mean $6.3 \times 10^2 \pm 4.91 \times 10^2$ ind.m^{-3} SD) during the open phase (Fig. 5.2). Similar to abundance patterns, low zooplankton biomass values were observed during the open phase, while high biomass values characterised the closed phase, at all three reaches of each estuary. At the Mdloti, mean zooplankton biomass ranged from 0.34 to 7.04×10^2 mg (DW) m^{-3} (mean $2.45 \times 10^5 \pm 3.49 \times 10^2$ mg (DW) m^{-3} SD) (Figure 5.3), while in the Mhlanga mean values ranged from 0.56 to 4.32×10^2 mg (DW) m^{-3} (mean $5.16 \times 10^1 \pm 1.93 \times 10^2$ mg (DW) m^{-3} SD) (Figure 5.4).

Overall, there was a significant difference in total zooplankton abundance between the two estuaries during the survey year (2-way ANOVA, $F_{1, 69} = 4.99$, $p < 0.05$), but not in total biomass ($F_{1, 68} = 1.15$, $p > 0.05$). When open and closed phases were considered separately, using a Mann-Whitney-U-test no significant differences were found in the abundance of the open phase of the two estuaries ($U = 92$, $p > 0.05$), but a significant difference in their biomass ($U = 68$, $p < 0.05$) was obtained. Conversely, during the closed phase, there were significant differences in abundance ($U = 107$, $p < 0.01$), but not in biomass ($U = 173$, $p > 0.05$) between the two estuaries.

Considering the two estuaries separately, there were significant differences in biomass and abundance between the closed and open phases of both the Mdloti ($U = 1$, $p < 0.001$, $U = 16$, $p > 0.001$, respectively) and the Mhlanga ($U = 101$, $p > 0.05$, $U = 88$, $p > 0.01$, respectively). Concerning differences between the different reaches of each estuary, using 1-way ANOVA no significant differences in abundance/biomass were obtained between the three reaches at either the Mdloti or the Mhlanga ($p > 0.05$ in all cases).

The higher zooplankton abundances/biomass recorded during the closed phase of both estuaries, as compared to their open phase, can be attributed to the stability of the system during the closed phase. This is due to reduced freshwater input and the restricted exchange of water with the sea (Kibirige 2002, Kibirige & Perissinotto 2003a). This is consistent with observations from other TOCEs, such as the Nyara (Perissinotto *et al.* 2000) and the Mpenjati (Kibirige 2002, Kibirige & Perissinotto 2003b). The highest zooplankton abundances/biomass coincided with periods of prolonged mouth closure. For instance, at the Mdloti the highest values were recorded in February 2003, following 22 days of mouth closure, and the second highest in October 2002 after only 9 days of closure.

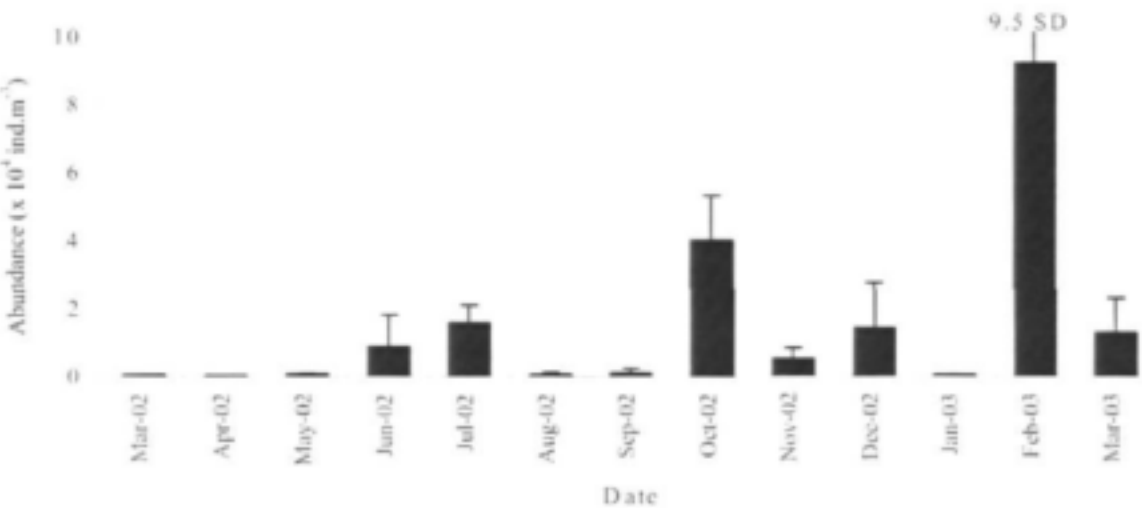


Figure 5.1 Total zooplankton abundance (mean ± SD) at the Mhlanga Estuary from March 2002 to March 2003

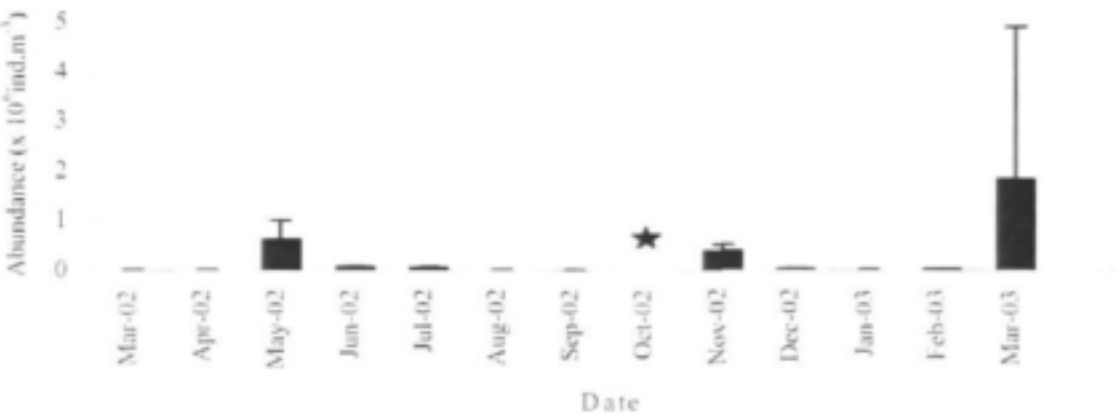


Figure 5.2 Total zooplankton abundance (mean ± SD) at the Mdloti Estuary from March 2002 to March 2003. :: no data.

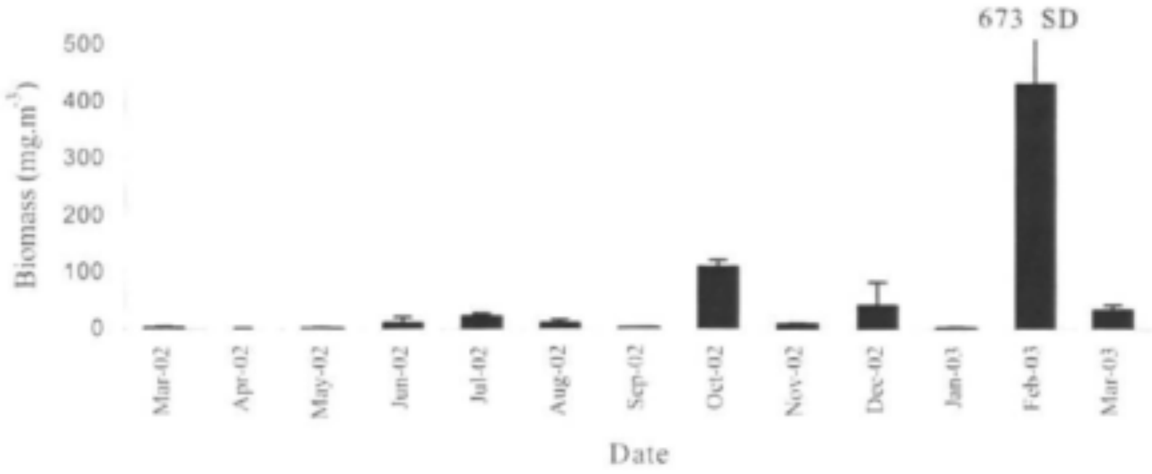


Figure 5.3 Total zooplankton biomass (mean ± SD) at the Mhlanga Estuary from March 2002 to March 2003

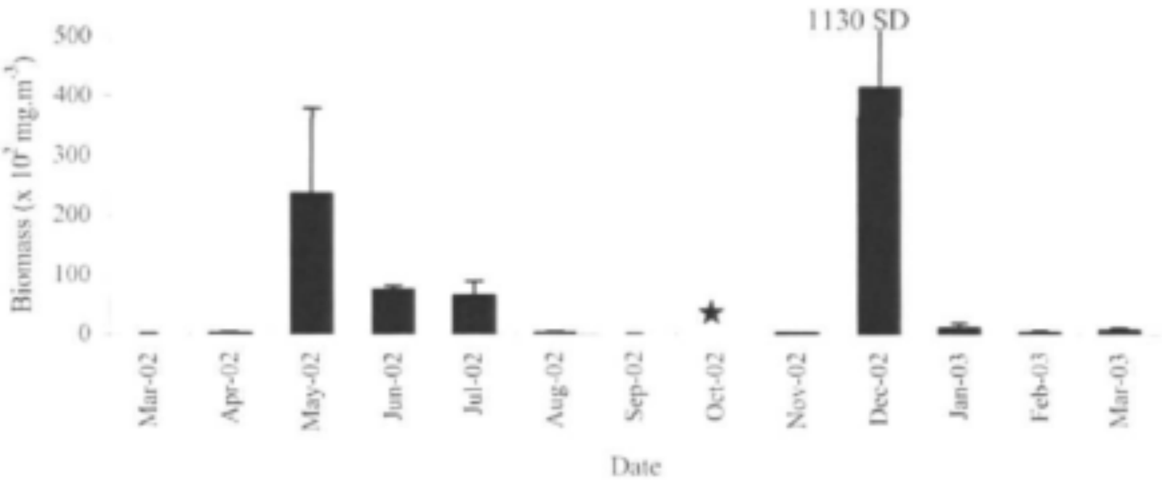


Figure 5.4 Total zooplankton biomass (mean ± SD) at the Mdloti Estuary from March 2002 to March 2003. :: no data

A total of 32 zooplankton taxa were recorded at the Mhlanga Estuary and 27 at the Mdloti Estuary during the survey. At the Mhlanga, a total of 25 zooplankton taxa were recorded during the closed phase and 31 during its open phase. At the Mdloti, a total of 23 taxa were recorded during the closed phase and 24 during the open phase. The zooplankton community of the Mhlanga was dominated by the calanoid copepod *Pseudodiaptomus hessei*, which during the closed phase accounted for up to 98.3% of the total catch in the lower reaches, 91.8% in the middle reaches and 97% in the upper reaches. During the same phase, caridean larvae contributed numerically 0.7%, 2.4% and 0.1% of the total in the lower, middle and upper reaches, respectively. None of the other groups contributed significantly during the closed phase except for branchyuran larvae, which were unexpectedly abundant during February 2003. The open phase of the Mhlanga was also dominated by *P. hessei*, which then accounted for up to 79.8% of the total number in the lower reaches, 88.1% in the middle and 74.4% in the upper reaches. Unidentified species of chironomid and damselfly larvae collectively contributed 7.4%, 6% and 2.5% numerically in the lower, middle and upper reaches, respectively. A freshwater cladoceran, *Ceriodaphnia producta*, on average accounted for 5.3% in the lower reaches and 1.9% and 5.2% in the middle and upper reaches, respectively. Other dominant species were the calanoid copepod *Acartia natalensis* and the cladoceran *C. producta*. At the Mdloti, during the closed phase the zooplankton community was also dominated by the calanoid copepod *Pseudodiaptomus hessei*, which accounted for up to 73.5% of the total catch in the lower reaches of the estuary, while *Ceriodaphna* sp. contributed 46% in the upper reaches. During the open phase, however, *P. hessei* contributed 84.6%, while Cyclopods and Ostracods each accounted for 28.3% of the total zooplankton abundance. There was also unexpected high numbers of rotifers during February 2003. The dominance of *P. hessei* in both estuaries can be attributed to the pioneering nature of this species and its ability to recover rapidly after floods (Jerling & Wooldridge 1995). The spatial distribution of this species both in the Mhlanga and the Mdloti suggests a wide salinity tolerance. This has been reported by other workers in this field (Wooldridge & Melville-Smith 1979, Grindley 1980), who noted tolerance over a salinity range from 1 to 74 ‰. The Mhlanga estuary was dominated by typical estuarine species. The higher abundance of taxa with strong marine links found in this estuary, compared to the Mdloti, can be attributed to the fact that the former opened more frequently than the latter. Prolonged mouth closure can have very pronounced effects on the community structure of any estuary. Studies at the Nhlabane Lake (Jerling & Cyrus 1999) showed a gradual shift in the community structure, from typically estuarine to freshwater dominated, after a period of prolonged mouth closure. Similar observations were made at the Mdloti, after a prolonged mouth closure of 67 days in July 2002. This resulted in the proliferation of the freshwater cladoceran *C. producta*. This species exhibited a well-defined spatial distribution, with the highest abundances recorded in the upper reaches. This phenomenon may be attributed to its inability to tolerate elevated salinity conditions.

Spearman rank correlation analysis of the Mhlanga data revealed significant correlation between total zooplankton abundance/biomass and phytoplankton as well as microphytobenthic biomass, but not with other physico-chemical parameters (Table 5.1). Caridean larvae exhibited a highly significant and positive correlation with temperature (Table 5.1), while *P. hessei* was positively correlated with dissolved oxygen.

Table 5.1 Spearman rank correlation analysis between physico-chemical parameters and abundance of total zooplankton and of its dominant components at the Mhlanga. n = 36. *: significant at $p < 0.05$; **: significant at $p < 0.01$, *** significant at $p < 0.001$.

Variable	Total Zoopl.	Caridean larvae	*P. hessei*	Branchyuran
Temperature	0.15	0.94**	-0.34	0.40
Salinity	0.07	-0.30	0.29	0.18
Dissolved Oxygen	0.42	-0.08	0.63*	-0.10
Rainfall	-0.01	-0.15	0.24	-0.28
Phytoplankton	0.51***	0.56	-0.01	0.17
Microphytobenthos	0.55***	0.25	0.05	-0.08

Concerning the Mdloti, again there were significant correlations between total zooplankton abundance/biomass and both phytoplankton and microphytobenthic biomass, but not with other physico-chemical parameters. However, *P. hessei* was negatively correlated with temperature, while *A. natalensis* exhibited negative correlations with both rainfall and microphytobenthic biomass (Table 5.2).

Table 5.2 Spearman rank correlation analysis between physico-chemical parameters and abundance of total zooplankton and its dominant components at the Mdloti. n = 36. *: significant at $p < 0.05$; **: significant at $p < 0.01$.

Variable	Total Zoopl.	*P. hessei*	*A. natalensis*	*C. producta*
Temperature	-0.58	-0.78*	-0.42	-0.41
Salinity	-0.37	-0.18	0.12	-0.46
Dissolved Oxygen	-0.10	0.09	0.14	-0.05
Rainfall	0.23	0.08	-0.73*	0.52
Phytoplankton	0.48**	0.00	0.28	-0.15
Microphytobenthos	0.46**	0.03	-0.73*	0.31

The high zooplankton abundances/biomass observed during the closed phase in the Mdloti and Mhlanga estuaries are in agreement with reports from other TOCEs in the region (Perissinotto *et al.* 2000, Kibirige & Perissinotto 2003a). In particular, a study conducted at the Mpenjati Estuary concluded that the zooplankton abundance in TOCEs is mainly controlled by the opening and closing of the estuary (Kibirige & Perissinotto 2003b). The higher zooplankton biomass/abundance found in the Mdloti Estuary, compared to the Mhlanga, could be attributed to the stability observed in this estuary during periods of prolonged closure. It has been reported previously that an extended period of mouth closure is required for zooplankton to respond to increases in microalgal availability (Whitfield 1980, Kibirige & Perissinotto 2003b). Conversely, the high frequency of breaching at the Mhlanga resulted in the regular flushing of zooplankton and their phytoplankton food out to sea. The residence time of water in the Mhlanga is, therefore, insufficient to result in biomass build-up of zooplankton within the estuary itself, although their regular export out to sea probably contributes significantly to the productivity of the adjacent coastal zone. This observation is in agreement with the reduction in copepods and mysids abundance/biomass reported from the Mpenjati TOCE after similar breaching events (Kibirige & Perissinotto 2003b).

6. MACROBENTHOS

Neither estuary supported a diverse benthic community; 11 taxa (4-9 per sample period) were recorded in the Mhlanga (Table 6.1) and 12 (4-8 per sample period) in the Mdloti (Table 7.2). Eight were shared, of which four were polychaetes.

The Mhlanga fauna was numerically dominated by polychaetes, particularly *Ceratonereis keiskama*, a common species of low salinity estuarine habitats, followed periodically by *Prionospio cf. steenstrupi.* Both of the South African estuarine thalassinids, *Callianassa kraussi* and *Upogebia africana* were recorded in the lower reaches although it should be noted that the deeper burrowing habit of these species relative to the smaller members of the benthos renders them less vulnerable to the sampling technique used in this study. *U.africana* was extremely rare in keeping with its general absence from systems which are frequently closed. The burrows of *C. kraussi* were very obvious downstream of the lower road bridge and extended upstream at least as far as the middle reaches. This species is well-known to survive in temporary open/closed estuaries and further personal observations in 1975 and by Whitfield (1980) suggest that it has been a long term inhabitant of this system.

Whitfield (1980) recorded the following 12 benthic taxa in the Mhlanga in 1978 , *viz.* the amphipods *Austrochiltonia capensis* and *Corophium triaenonyx*, the bivalve *Musculus virgiliae*, unidentified Cumacea, chironomid fly larvae, the isopods *Cirolana fluviatilis* and *Leptanthura laevigata* , the caridean prawn *Macrobrachium equidens*, oligochaetes, ostracods and the polychaetes *Ceratonereis erythraeensis* and *Dendronereis arborifera.* This list indicates that there has been little change in the species richness, some change in the species composition but a consistent total dominance by the polychaetes. The amphipod *Corophium triaenonyx* followed the polychaetes in 1978 but was not recorded in 2002-2003. Whitfield (*loc.cit.)* recorded only two months during 1978 when the mouth was open as opposed to frequent mouth open conditions during 2002-2003. The significance as regards the presence/absence of different species is unknown. Whitfield (*loc.cit.)* recorded abundance as biomass expressed in $kJ.m^{-2}$ as opposed to $numbers.m^{-2}$ in the present study which complicates further comparisons.

Apart from *M.equidens* and ostracods, Whitfield (*loc.cit.)* found a decline in biomass in all taxa from the lower to the upper stations in 1978. This pattern was not repeated in the present study except in March 2002; the greater frequency of mouth opening and consequent variations in habitat availability in the lower reaches was a likely contributory factor. The consistency in sediment composition at each station during the survey implicates other factors in the variations recorded in the benthos.

The Mdloti benthos (Table 6.2) was dominated in March, June and September 2002 by polychaetes, particularly *Ceratonereis keiskama*, which occurred widely throughout the survey, and *Desdemona ornata,* with the addition of *Prionospio cf.steenstrupi* in June. Numbers of the amphipods *Grandidierella lignorum* and *G.lutosa* began to increase in September and continued this trend through December 2002 and March 2003 with a concurrent decrease in the polychaetes except for *C.keiskama.* The nature of any interaction between the polychaetes and the amphipods is unknown.

Disregarding single records, Blaber, Hay, Cyrus & Martin (1984) recorded the following 12 taxa between December 1980 and November 1981, *viz.* Oligochaeta, the polychaetes *Ceratonereis erythraeensis, Dendronereis arborifera, Desdemona ornata, Owenia sp.* and *Prionospio sp.*, the amphipod *Africhiltonia sp.*, the tanaid *Apseudes sp.*, the caridean *Caridina nilotica,* the gastropod *Ancylus sp.,* chironomid larvae and ephemeroptan nymphs. Comparison with the 2002-2003 data indicates broad similarity in terms of both the taxa present and their total number as well as a high level of similarity as regards the general dominance by the polychaetes although there was a change in the polychaete/ amphipod balance in 2002/2003 as shown above. The benthic biomass in the lower and middle reaches of the Mdloti increased 3-5 fold during the closed period between June and September (Blaber *et al.* 1984), a trend which was discernible in the 2002-2003 data but confounded by the irregular breaching pattern. Blaber *et al.* (*loc.cit.*), as was done by Whitfield (*loc.cit.*), recorded biomass in kJ.m^{-2} which made further comparisons impossible.

The 2002-2003 data showed that numerically the benthos of both systems was generally dominated by polychaetes except for the later period in the Mdloti. Minimum total individual densities in the two systems were similar at *ca.* 50.m^{-2} but maximum densities differed markedly (Figure 6.1). Peaks in total individual densities in the Mhlanga of > 3 200. m^{-2} in June at all sites and > 5 500 m^{-2} in December at the upper site were associated with high numbers of the polychaete *Prionospio steenstrupi.*

Total individual densities in the Mdloti were generally higher than in the Mhlanga and associated with a greater variety and abundance of polychaetes (Tables 6.1, 6.2). Conversely, the highest densities of the year, *ca.* 24 000.m^{-2} in the mouth region site in December 2002 involved only two species, the polychaete *Ceratonereis keiskama* and the amphipod *Grandidierella sp.*

There was no clear indication of any longitudinal trends in species richness nor abundance in either system. The over-riding impression was of low species diversity and high variability in total individual density.

The hypothesis that closure would result in a different macrobenthic community after three months has to date not been demonstrated nor tested because there has not been an unbroken three month closure period. The Mhlanga appeared to be open or have at least some link with the sea for about half of the study period; closure seldom lasted longer than a month. The Mdloti appeared to be closed for slightly longer than the Mhlanga but again there were no extended periods of closure. Visual inspection of the closure periods and the variation in abundance of the macrobenthos suggested that greater abundance did follow periods of closure. The erratic nature of the open/closed periods and the difficulty of obtaining good data on the behaviour of the sand bars means that this relationship would require further testing. In retrospect, the data collected by Blaber *et al.* (1980) on the Mdloti between Dec 1980 and November 1981 and by Whitfield (1984) on the Mhlanga between January and November 1978 are, however, supportive of the hypotheses developed prior to the present study. Both papers indicate an increase in benthic biomass during periods of mouth closure but do not give any indication of a change towards a community dominated by filter feeders although it must be conceded that little is known of the feeding behaviour of the species recorded.

The data collected to date indicate a depauperate estuarine fauna in both systems dominated for much of the time at present by small polychaetes. There remains, however, an interesting contrast between the Mhlanga, which supports a *C. kraussi* population extending at least as far as the middle reaches, as well as small numbers of *U. africana*, and the Mdloti where neither species was recorded. *C. kraussi* cannot breed in very low salinities so the generally higher salinities in the Mhlanga associated with the more frequently observed overtopping of the bar probably contributes to the occurrence of these species in this system. Its survival in the past when the bar was possibly more substantial and the mouth was closed for longer periods presents something of an enigma. The significance of this species in the functioning of the system should not be disregarded as it is certainly a major contributor to the total benthic biomass. The sustained presence of a *U.africana* population is moot as this species has an obligatory marine phase and is dependent on an open mouth for completion of its life cycle. The absence of any records of either species from the Mdloti is indicative of longer periods of mouth closure in combination with longer periods of lower salinities.

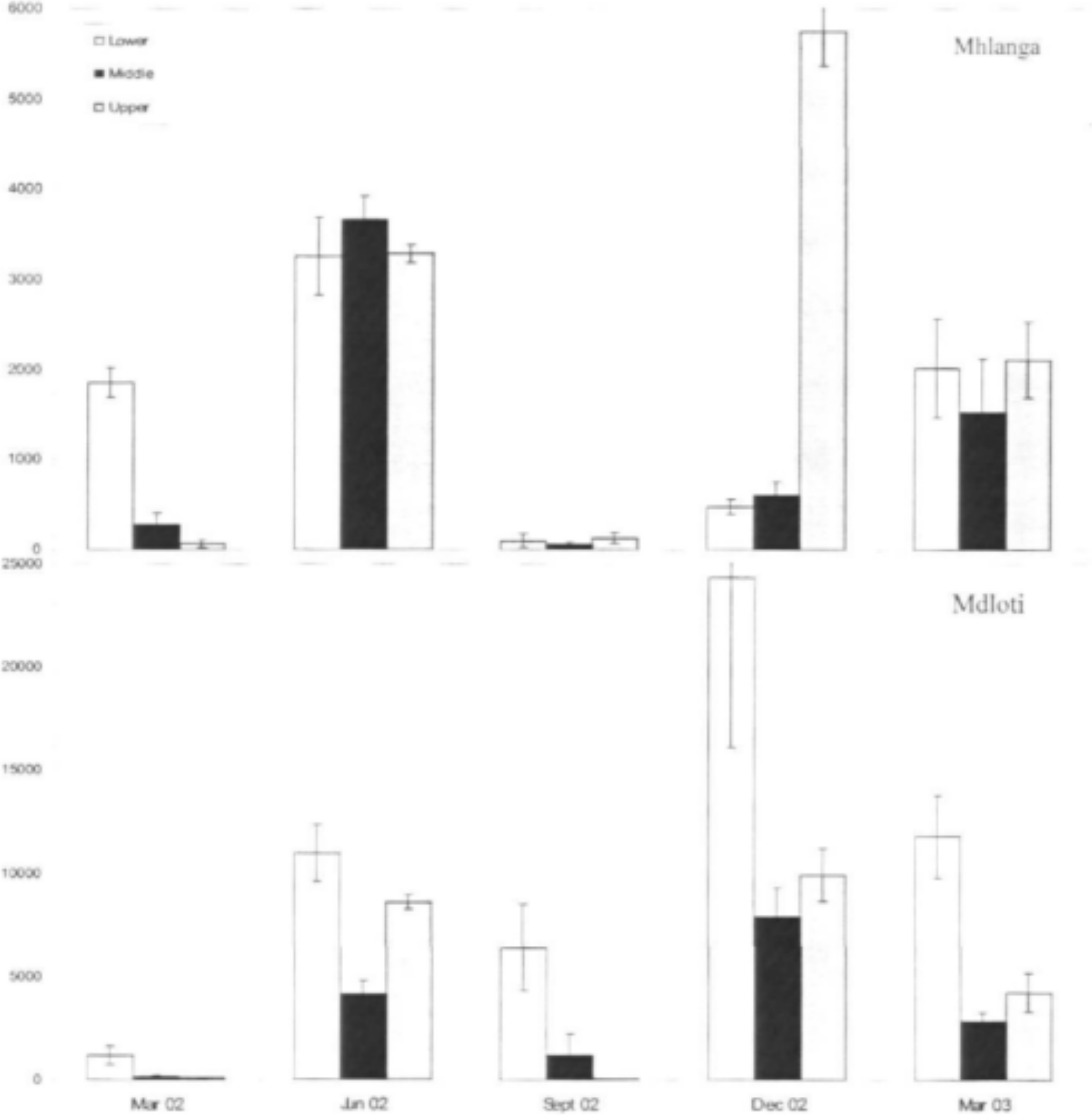


Figure 6.1 Total benthic densities (individuals.m^{-2}) in the Mhlanga and Mdloti estuaries during March 2002 – March 2003. Note different vertical scales. Bars represent standard error.

Table 6.1 Mean benthic densities at three sites in the Mhlanga during 2002 and March 2003. Numbers represent numbers of individuals per m^2

	2002												2003		
	March			June			September			December			March		
Sites	**L**	**M**	**U**	**L**	**M**	**U**	**L**	**M**	**U**	**L**	**M**	**U**	**L**	**M**	**U**
ANNELIDA															
Polychaeta															
Capitellidae	15	10	69	183		54	94	35	10	30	316	232	5	143	
Ceratonereis keiskama	1791	197		1593	30	54		20	49	444	163	187	1983	483	1633
Dendronereis arborifera		64			5					5				94	20
Desdemona ornate						10			5			25			
Prionospio cf. steenstrupi				1480	3626	3167			54		94	5032	20	883	405
CRUSTACEA															
Amphipoda															
Grandidierella sp.	20						5		15		39	276	10	15	49
Cumacea	10														
Tanaidacea															
Apseudes digitalis		5													
Thalassinidea															
Callianassa kraussi	5														
Upogebia africana	5														
Insecta															
Chironimidae	5														5
Total mean densities (individ.m^2)	**1845**	**276**	**69**	**3256**	**3661**	**3286**	**99**	**54**	**133**	**479**	**612**	**5753**	**2018**	**1618**	**2112**
Total number of taxa	7	4	1	3	3	4	2	2	5	3	4	5	4	5	5

Table 6.2 Mean benthic densities at three sites in the Mdloti during 2002. Numbers represent numbers of individuals per m^2

	2002												2003		
	March			June			September			December			March		
Sites	**L**	**M**	**U**	**L**	**M**	**U**	**L**	**M**	**U**	**L**	**M**	**U**	**L**	**M**	**U**
ANNELIDA															
Oligochaeta													123		395
Polychaeta															
Capitellidae		5		133	326	306			10						
Ceratonereis keiskama	1125	109	64	3843	1145	6177	2279	918	25	9877	7765	6033	7114	252	
Dendronereis arborifera	39														
Desdemona ornate		30	10	6073	2491	1756	2955	247							
Prionospio cf. steenstrupi				824	153	331							44		
CRUSTACEA															
Amphipoda															
Grandidierella sp.				10			1164	15		14874	153		4100	173	5
G. lignorum										345			99		
G. lutosa										266			197		
Tanaidacea															
Apseudes digitalis	5							5							
MOLLUSCA															
Bivalvia															
Brachidontes virgilliae													5		
INSECTA															
Diptera															
Ceratopogonidae													10	74	296
Chironomidae		5	5	84	39	39						3907	99	2348	3552
Total mean densities (individ.m^2)	**1169**	**148**	**79**	**10967**	**4154**	**8609**	**6404**	**1179**	**35**	**25363**	**7918**	**9941**	**11791**	**2847**	**4248**
Total number of taxa	3	5	3	6	5	5	4	3	2	3	2	2	8	4	3

7. Fish

7.1 Fish associations and biomass

Although there appeared to be similar trends in the fish communities of the two estuaries during the study period, the Mdloti showed a greater total abundance (3687 individuals) compared to the Mhlanga (1742 individuals). The total biomass, on the other hand, was virtually the same in the two estuaries, with 158 Kg at the Mdloti and 139 Kg at the Mhlanga. There was a temporal inversion in dominance in the fish biomass of the two estuaries, with the Mdloti exhibiting higher stocks than the Mhlanga from June to December 2002, but lower during March of both years (i.e. 2002 and 2003). Major peaks in fish abundance were observed in June and December in the Mdloti, in conjunction with the open state of the mouth (Figure 7.1). The Mhlanga showed much lower abundances, with a more consistent pattern over the period of the survey (Figure 7.1). The majority of fish caught in both estuaries belong in the estuarine-marine dependent species (EMDS) group, although the Mhlanga yielded a greater proportion of freshwater fishes (Figure 7.2, Table 7.1) than the Mdloti (Figure 7.3, Table 7.2). However, the biomass contribution of the EMDS was greater in the Mdloti than in the Mhlanga (Figures 7.2 & 7.3).

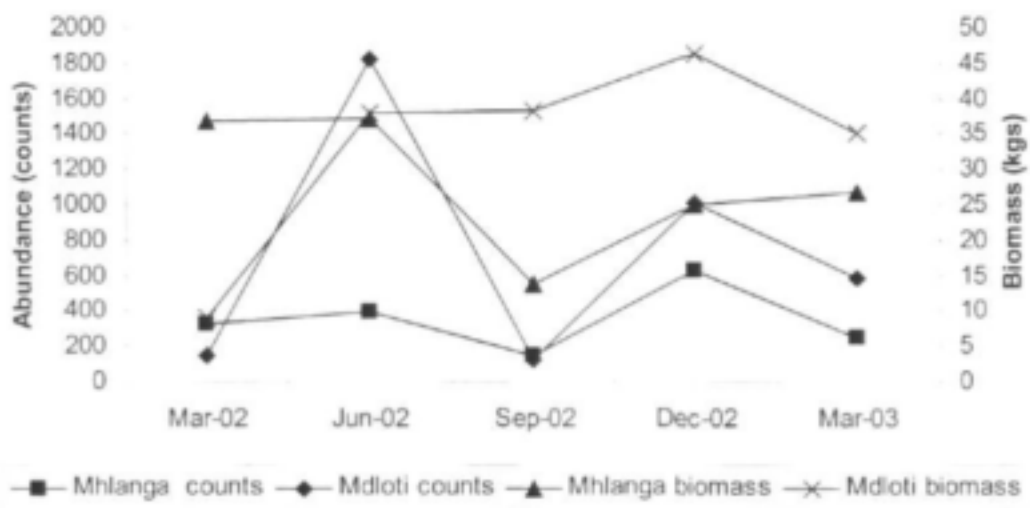


Figure 7.1. A comparison between the total abundance and biomass of the two estuaries, March 2002-March 2003

Table 7.1 Total species list at Mhlanga for both gill and seine nets from March 2002-2003. Numbers in brackets are percentages.

SCIENTIFIC NAME	COMMON NAME	GILL NET No	%	GILL NET Mass (g)	%	SEINE NET No	%	SEINE NET Mass (g)	%
Ambassis natalensis	Slender glassy	3	(0.7)	119.0	(0.1)	42	(3.2)	337.09	(0.9)
Ambassis productus	Longspine glassy	1	(0.2)	66.0	(0.1)	5	(0.4)	25.48	(0.1)
Argyrosomus japonicus	kob	1	(0.2)	85.0	(0.1)	0	0	0	0
Caranx papuensis	Brassy kingfish	0	(0.0)	0.0	(0.0)	1	(0.1)	6.27	(0.0)
Caranx sexfasciatus	Bigeye kingfish	7	(1.6)	406.4	(0.4)	1	(0.1)	22.03	(0.1)
Clarias gariepinus	Sharptooth catfish	36	(8.4)	48667.0	(47.1)	1	(0.1)	10943.00	(30.3)
Diplodus sargus capensis	Blacktail	0	(0.0)	0.0	(0.0)	7	(0.5)	9.05	(0.0)
Elops machnata	Ladyfish	2	(0.5)	283.0	(0.3)	0	0	0	0
Gilchristella aestuaria	Estuarine roundherring	0	(0.0)	0.0	(0.0)	44	(3.4)	112.62	(0.3)
Glossogobius callidus	River goby	0	(0.5)	642.0	(0.6)	5	(0.2)	5.98	(0.0)
Leiognathus equula	Slimy	0	(0.0)	0.0	(0.0)	1	(0.1)	6.52	(0.0)
Gerres rappi	Evenfin pursemouth	1	(0.2)	48.0	(0.0)	9	(0.6)	123.90	(0.3)
Iso natalensis	Surfsprat	0	0	0	0	7	(0.5)	4.67	(0.0)
Liza alata	Diamond mullet	34	(7.9)	6451.0	(6.2)	3	(0.2)	125.16	(0.3)
Liza dumerilii	Groovy mullet	32	(7.5)	3384.8	(3.3)	79	(6.0)	2890.09	(8.0)
Liza macrolepis	Large-scale mullet	30	(7.0)	3920.5	(3.8)	7	(0.5)	203.15	(0.6)
Liza sp.		0	(0.0)	0.0	(0.0)	159	(12.1)	27.10	(0.1)
Liza tricuspidens	Striped mullet	6	(1.4)	882.0	(0.9)	9	(0.7)	732.25	(2.0)
Monodactylus falciformis	Cape moony	1	(0.2)	77.0	(0.1)	1	(0.1)	18.11	(0.1)
Mugil cephalus	Flathead mullet	42	(9.8)	5889.5	(5.7)	16	(1.2)	766.14	(2.1)
Myxus capensis	Freshwater mullet	7	(1.6)	2606.1	(2.5)	111	(8.5)	4927.94	(13.6)
Oreochromis mossambicus	Mocambique tilapia	38	(8.9)	11429.0	(11.1)	426	(32.4)	2518.50	(7.0)
Pomadasys commersonnii	Spotted grunter	15	(3.5)	1715.0	(1.7)	3	(0.2)	132.80	(0.4)
Pseudorhombus arsius	Largetooth flounder	0	(0.0)	0.0	(0.0)	4	(0.3)	2.50	(0.0)
Rhabdosargus holubi	Cape stumpnose	10	(2.3)	394.0	(0.4)	104	(7.9)	1138.15	(3.1)
Rhabdosargus sarba	Natal stumpnose	1	(0.2)	40.0	(0.0)	9	(0.7)	217.12	(0.6)
Scomberoides tol	Queenfish	0	(0.0)	0.0	(0.0)	1	(0.1)	5.20	(0.0)
Solea bleekeri	Blackhand sole	0	(0.0)	0.0	(0.0)	7	(0.5)	10.28	(0.0)
Terapon jarbua	Thornfish	3	(0.7)	146.0	(0.1)	14	(1.1)	84.97	(0.2)
Valamugil cunnesius	Longarm mullet	154	(35.9)	15654.2	(15.2)	189	(14.4)	10493.56	(29.0)
Valamugil robustus	Robust mullet	3	(0.7)	382.0	(0.4)	51	(3.9)	284.26	(0.8)
TOTAL		429		103287.6		1313		36173.89	

Table7.2 Total species list for Mdloti both gill and seine nets from March 2002-2003. Numbers in brackets are percentages.

SCIENTIFIC NAME	COMMON NAME	GILL NET NUMBER		GILL NET Mass (g)	%	SEINE NET NUMBER		SEINE NET Mass (g)	%
Acanthopagrus berda	Riverbream	0	(0.0)	0.00	(0.0)	2	(0.06)	17.40	(0.1)
Ambassis natalensis	Slender glassy	0	(0.0)	0.00	(0.0)	20	(0.65)	35.03	(0.2)
Argyrosomus japonicus	Dusky kob	5	(0.97)	1045.79	(0.75)	0	(0.0)	0.00	(0.0)
Caranx papuensis	Brassy kingfish	0	(0.0)	0.00	(0.0)	1	(0.03)	2.97	(0.02)
Caranx sexfasciatus	Bigeye kingfish	5	(0.97)	276.00	(0.2)	1	(0.03)	3.46	(0.02)
Clarias gariepinus	Sharptooth catfish	23	(3.49)	23037.00	(16.43)	1	(0.06)	6637.70	(37.09)
Gerrres rappi	Evenfin pursemouth	0	(0.0)	0.00	(0.0)	5	(0.16)	262.55	(1.47)
Glossogobius callidus	River goby	0	(0.0)	0.00	(0.0)	4	(0.13)	6.30	(0.04)
Gilchristella aestuaria	Estuarine roundherring	0	(0.0)	0.00	(0.0)	147	(4.76)	37.78	(0.21)
Leiognathus equula	Slimy	0	(0.0)	0.00	(0.0)	181	(5.86)	140.54	(0.79)
Liza alata	Diamond mullet	8	(1.55)	2554.97	(1.82)	3	(0.1)	46.50	(0.26)
Liza dumerilii	Groovy mullet	1	(0.19)	103.00	(0.07)	34	(1.1)	1417.27	(7.92)
Liza macrolepis	Large-scale mullet	4	(0.78)	470.19	(0.34)	2	(0.06)	14.34	(0.08)
Liza sp.		0	(0.0)	0.00	(0.0)	1278	(41.4)	911.36	(5.09)
Liza tricuspidens	Striped mullet	32	(6.2)	13135.00	(9.37)	8	(0.26)	259.57	(1.45)
Monodactylus falciformis	Cape moony	15	(2.91)	741.00	(0.53)	0	(0.0)	0.00	(0.0)
Mugil cephalus	Flathead mullet	61	(11.82)	25782.48	(18.39)	48	(1.55)	333.79	(1.87)
Myxus capensis	Freshwater mullet	49	(9.5)	18347.20	(13.09)	142	(4.6)	222.32	(1.24)
Oligolepis acutipennis	Sharptail goby	0	(0.0)	0.00	(0.0)	1	(0.03)	3.06	(0.02)
Oligolepis keiensis	Kei goby	0	(0.0)	0.00	(0.0)	2	(0.06)	0.51	(0.0)
Oreochromis mossambicus	Mocambique tilapia	75	(12.21)	18484.29	(13.19)	92	(1.0)	903.26	(5.05)
Pomadasys commersonnii	Spotted grunter	27	(5.23)	4795.59	(3.42)	128	(4.15)	120.55	(0.67)
Rhabdosargus holubi	Cape stumpnose	0	(0.0)	0.00	(0.0)	543	(17.59)	5040.49	(28.17)
Rhabdosargus sarba	Natal stumpnose	0	(0.0)	0.00	(0.0)	13	(0.42)	76.46	(0.43)
Solea bleekeri	Blackhand sole	0	(0.0)	0.00	(0.0)	9	(0.29)	16.70	(0.09)
Terapon jarbua	Thornfish	0	(0.0)	0.00	(0.0)	258	(8.36)	685.67	(3.83)
Tilapia rendalli	Redbreasted tilapia	0	(0.0)	0.00	(0.0)	6	(0.13)	54.10	(0.3)
Valamugil cunnesius	Longarm mullet	222	(43.02)	28337.79	(20.21)	129	(4.18)	449.29	(2.51)
Valamugil buchanani	Bluetail mullet	1	(0.19)	107.58	(0.08)	0	(0.0)	0.00	(0.0)
Valamugil robustus	Robust mullet	5	(0.97)	2970.00	(2.12)	91	(2.95)	196.06	(1.1)
TOTAL		**533**		**140187.88**		**3154**		**17895.03**	

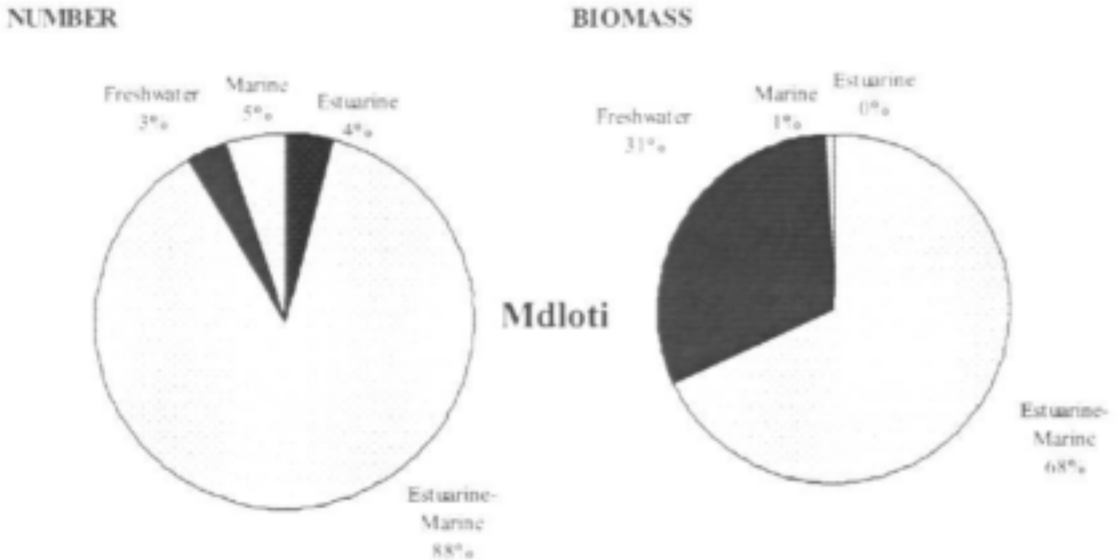


Figure 7.2 Abundance and biomass of fish at Mdloti March 2002-2003

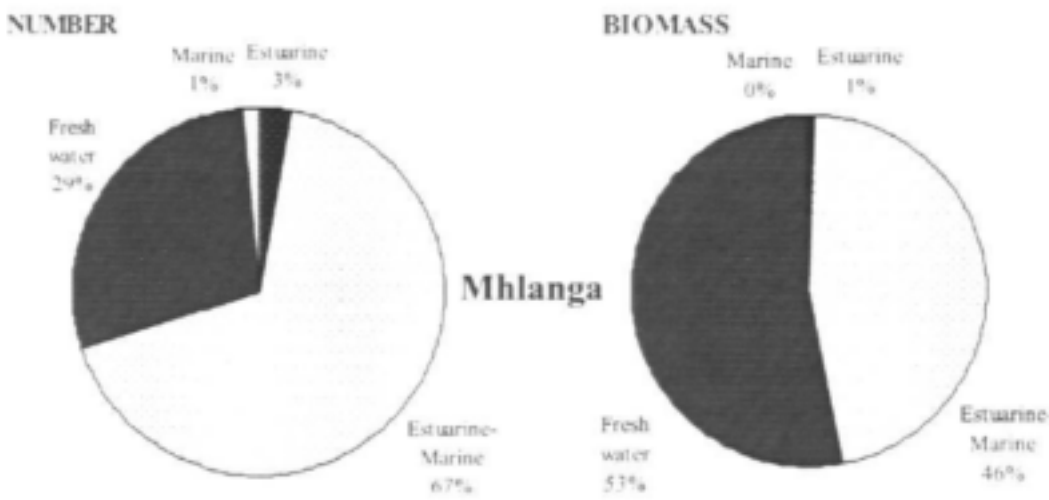


Figure 7.3 Abundance and biomass of fish at Mhlanga March 2002-2003

During the study period, the Mhlanga breached more frequently than the Mdloti, and EMDS fish in the Mhlanga showed a consistently lower percent abundance on each sampling occasion, due to the dominance of freshwater species in the Mhlanga (summarised in Figure 7.2). Diversity was, however, generally higher in the Mhlanga, with an average of 17.6 species collected, compared to 15.4 for the Mdloti. This is consistent with the expectation that mouth open condition would result in higher diversity of fish species. The Mdloti also exhibited this trend, with EMDS abundance peaks in June and December 2002, coinciding with open mouth phases.

A comparison of all mullets between the two estuaries, showed that juveniles of this group were recruiting into both estuaries. The data can be summarised as follows (values in brackets are gillnet catches):

		Mar-02	Jun-02	Sep-02	Dec-02	Mar-03
Mhlanga	Number	227	353	120	119	114
	biomass (g)	7243	19970	8357	10150	13508
	mean mass	31.9	56.6	69.6	85.3	118.5
Mdloti	Number	75	1434 (159)	86	256	272
	biomass (g)	2597	27813(26902)	12421	27458	25368
	mean mass	34.6	19.4 (169.2)	114.4	107.3	93.3

The data show a similar size pattern in the fishes from the two systems. A feature of the June data for Mdloti, was the capture of 1275 small mullets, with a mass of less than 1 g each (1275 fish weighed 907 g). Separation of the gillnet catch from this, yielded 159 fish with a mean weight of 169 g. Since the mouth was open at about that time, this implies that while recruitment was occurring, many of the bigger fishes were still in the system at the time of sampling. Total biomass of mullet was significantly higher in the Mdloti, which yielded a total of 95.6 kg, compared to 59.2 kg from the Mhlanga.

There are significant correlations between the total abundance/biomass of fish and some abiotic parameters, with salinity being the only consistent factor significant in both estuaries (Table 7.4).

Table 7.4 Spearman correlation coefficients of total fish abundance versus abiotic factors in the Mdloti and Mhlanga

a) **MDLOTI**		Total Abundance	
Variable	Lower	Middle	Upper
Salinity (surface)	-0.90*	-0.51	0.80
Salinity (bottom)	-0.92*	-0.51	0.80
Temperature (surface)	-0.40	-0.60	0.10
Temperature (bottom)	0.30	-0.70	0.00
Dissolved Oxygen (surface)	0.30	0.30	-0.10
Dissolved Oxygen (bottom)	-0.10	0.30	0.20
Turbidity (Kd)	-0.82	0.13	0.70
b) **MHLANGA**		Total Abundance	
Variable	Lower	Middle	Upper
Salinity (surface)	-0.60	0.61	0.87
Salinity (bottom)	-1.00***	-0.15	0.67
Temperature (surface)	0.20	0.71	0.62
Temperature (bottom)	-0.20	0.82	0.62
Dissolved Oxygen (surface)	-0.90*	-0.67	-0.05
Dissolved Oxygen (bottom)	0.20	-0.67	-0.21
Turbidity (Kd)	0.20	0.15	-0.98**

Significance levels: * = $P < 0.05$ ** = $P < 0.01$ *** = $P < 0.001$

Fish in the Mhlanga also showed a significant correlation with dissolved oxygen and turbidity (Table 7.4), possibly as a direct result of increases in freshwater flow. The abundance of EMDS correlated significantly with salinity in the lower reaches of the Mdloti, but in the Mhlanga a significant correlation was only obtained between EMDS abundance and turbidity (Table 7.5).

Table 7.5: Spearman correlation coefficients of EMDS abundance versus abiotic factors in the Mdloti and Mhlanga

a) **MDLOTI**		Total Abundance	
Variable	Lower	Middle	Upper
Salinity (surface)	-0.9	-0.1	0.8
Salinity (bottom)	-0.9*	-0.1	0.8
Temperature (surface)	-0.4	-0.6	-0.2
Temperature (bottom)	-0.3	-0.7	-0.1
Dissolved Oxygen (surface)	-0.3	0.3	-0.1
Dissolved Oxygen (bottom)	-0.1	0.3	0.2
Turbidity (Kd)	-0.8	0.1	0.4
b) **MHLANGA**		Total Abundance	
Variable	Lower	Middle	Upper
Salinity (surface)	0	0.7	0.8
Salinity (bottom)	-0.4	0.1	0.7
Temperature (surface)	-0.1	0.3	0.6
Temperature (bottom)	-0.1	0.7	0.6
Dissolved Oxygen (surface)	-0.3	-0.5	-0.1
Dissolved Oxygen (bottom)	0.1	-0.5	-0.2
Turbidity (Kd)	-0.2	0.1	-0.9**

Significance levels: * = $P < 0.05$ ** = $P < 0.01$

Table 7.6. Spearman correlation coefficients of Total fish abundance versus biotic factors in the Mdloti and Mhlanga

a) **MDLOTI**		Total Abundance	
Variable	Lower	Middle	Upper
Zooplankton	0.60	0.70	0.60
Phytoplankton	0.70	0.60	0.30
Microphytobenthos	1.00***	0.80	0.30
b) **MHLANGA**		Total Abundance	
Variable	Lower	Middle	Upper
Zooplankton	0.70	0.36	0.15
Phytoplankton	-0.50	0.82	0.36
Microphytobenthos	0.50	0.15	0.21

Significance levels: * = $P < 0.05$ ** = $P < 0.01$ *** = $P < 0.001$

Significant correlations between the total abundance of fish and their food sources were obtained for the Mdloti (i.e. microphytobenthos , Table 7.6). By comparison, the total abundance of fish in the Mhlanga does not appear to correlate with any food types. This pattern is mirrored by the abundance of the EMDS, which exhibit a significant correlation with microphytobenthos and phytoplankton in the lower and middle reaches of the Mdloti. However, the EMDS abundance in Mhlanga is only significantly correlated to the phytoplankton in the middle reaches (Table 7.7).

Table 7.7 Spearman correlation coefficients of EMDS abundance versus biotic factors in the Mdloti and Mhlanga

a) **MDLOTI**		Total Abundance	
Variable	Lower	Middle	Upper
Zooplankton	0.6	0.7	0.6
Phytoplankton	1.0**	1.0**	0.6
Microphytobenthos	1.0**	0.8	-0.1
b) **MHLANGA**		Total Abundance	
Variable	Lower	Middle	Upper
Zooplankton	0.1	0.4	0.2
Phytoplankton	-0.6	0.9**	0.4
Microphytobenthos	0.7	0.1	0.2
Significance levels: * = $P < 0.05$ ** = $P < 0.01$			

The fish diversity in both the Mdloti and the Mhlanga estuaries appears to have changed since the 1980's, with the number of estuarine-marine dependent species (EMDS) reduced during the present study. From results obtained by Blaber (1984) for the Mdloti, there appears to have been a 20% decrease in EMDS between 1984 and the present study. During the same period there has been a general increase in marine, freshwater and estuarine dependent groups (Figure 7.4). Similarly in the Mhlanga, the EMDS group has decreased by 10%, compared to the findings of Whitfield (1980a), while the marine group has shown a 12% increase since 1980 (Figure 7.4), albeit off a small base. The reduction in the EDMS group in the Mdloti, is probably a reflection of the reduced open mouth condition, resulting from the low rainfall conditions prevailing during the study. However, the Mhlanga has been seen to open frequently during this study, due to the increased freshwater flow from sewage treatment works in the catchment. In this case the reduced saltwater prism has led to a relatively higher dominance of freshwater species in the fish community.

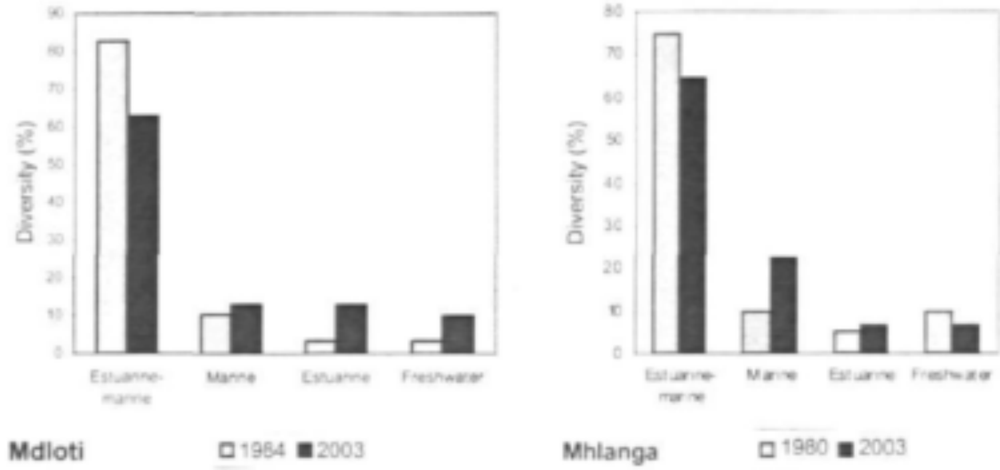


Figure 7.4 A comparision between the species diversity in the 1980's and the present study for both the Mdloti and the Mhlanga (Blaber *et al*, 1984 & Whitfield, 1980a)

The Mhlanga and Mdloti showed similar changes in diversity of fish species since the 1980's, mainly in the EMDS group. The Mdloti did, however, have a greater abundance and biomass of fish compared to the Mhlanga. This may not be due to poor recruitment to the Mhlanga, but rather because the frequently open mouth gives rise to unfavourable conditions in the Mhlanga (Whitfield, 1980d), caused by low retention time. Wallace (1974) found that juveniles are more abundant in open estuaries, but their retention and development within the estuary is dependent on mouth state (Kok & Whitfield, 1986), and water retention within the system. This study has shown that species such as *Liza* spp, are recruiting in during the open phase of each estuary, but growth and retention was apparently better in the Mdloti, resulting in a significantly higher total biomass captured in the gillnets.

In 1984, the Mdloti mouth was open intermittently throughout the year. However, it was completely closed from late May to late August (Blaber *et al*., 1984). A similar pattern occurred during the present study, although heavy rains in July 2002 caused breaching. The Mhlanga was closed between April and September in 1980, only opening intermittently from October to March (Whitfield, 1980a). This is very different to the present study, where the Mhlanga was shown to breach almost once a month.

The total abundance of fish shows a similar trend in both estuaries when correlated with salinity. EMDS specifically show a strong correlation with salinity in both estuaries. Whitfield (1984) showed that over 40 EMDS can tolerate salinities of less than 2‰. Thus the increase in freshwater flow may not be the factor affecting the apparent decrease in numbers of EMDS. However, it is a factor in allowing freshwater dependent species to compete with the estuarine species if low salinities are sustained for long periods.

The EMDS have also correlated well with turbidity in the Mhlanga. Cardona (2000) found that fish are affected more by changes in salinity and temperature, and that turbidity is a secondary factor in fish distribution. Cyrus & Blaber (1987) described 80% of South African species as turbid water taxa that prefer conditions that allow an increase in the survival of juveniles. This is consistent with the current study, particularly in the Mhlanga where increased turbidity appears to support increased numbers of juveniles. An increase in turbidity does, however, result in a decrease in light penetration, which consequently affects the photosynthesis of microalgae and macroalgae (Miller & Dunn, 1980). This could indirectly affect food abundance on which the juveniles rely for survival and growth.

During the closed phase, the total abundance and biomass of fish show significant differences between the two estuaries and more specifically the EMDS show a marked difference between the biomass of adults and juveniles in the two systems. As indicated earlier, the Mdloti has a greater biomass of EMDS compared with the Mhlanga, and this may be related to the longer retention time found in the former estuary compared to the latter. As nutrients enter an estuary via freshwater inflow, they are rapidly taken up by phytoplankton (Bate *et al*, 2002), which is then fed upon by zooplankton. This process requires 2-6 weeks for a substantial food source to build up for the fish (Perissinotto *et al*., 2003). Whitfield (1980) found that the distribution of fish in the Mhlanga was mainly related to food availability. Most post-larval fish feed on macro and microzooplankton (Whitfield, 1985), and the correlation between the total abundance of EMDS and the biotic parameters showed many significant values for the Mdloti. The EMDS group in the Mhlanga, however, was only correlated to phytoplankton in the middle reaches. This may relate to the increase in breaching frequency, at the Mhlanga, that occurred during the study period, with a basin in the middle reaches providing sanctuary for the phytoplankton. Baird *et al* (1998) found that zooplankton and zoobenthos can decrease by as much as 90% when an estuary opens, due to estuarine water loss and the scouring effect. Marais (1982) also found that macrobenthos can only withstand salinities of 1,7‰ for approximately 8 days. In the Mhlanga, there may not be enough time, between breachings, for substantial zooplankton and benthic biomass build up, before the next washed out occurs. On the other hand, the Mdloti has longer retention times and therefore a more stable zooplankton and zoobenthos most of the time.

7.2 Recruitment during overtopping

Figure 7.5. The fry net, being deployed in the surf-edge backwash zone at Illovo estuary.

A summary of the catches is presented in Table 7.8. When looking at this table it is important to note the mouth states for the Illovo Estuary, added to the title of the table. Unfortunately only one genuine overtopping event was encountered, on 10 September, when 8 tiny mullet, 2 *Stolephorus* and 5 *R. holubi* were collected by placing the net in front of gentle overtopping waves running into the estuary, across the wide sand berm (Fig 7.5). Prior to this all efforts at the Mhlanga, using two nets, had proved to be relatively unproductive. A visit the following day, in the hope of a bigger overtopping event, yielded no overtopping, but surf-zone backwash sampling proved very effective. Between spring tide events, rainfall induced the Illovo Estuary to open, and the following three visits, while not adding to the overtopping data, did show consistent recruitment, while the Mhlanga continued to yield virtually nothing. For the last two occasions, the Karridene was visited, because the Illovo remained open. Although no overtopping was seen at the Karridene, the effort there showed the presence of *R. holubi* and *Stolephorus*. In all, nine families were identified, with far larger numbers entering the Illovo Estuary than the Mhlanga. By far the majority were species that spawn at sea, but utilize the estuary as a nursery area (Whitfield, 1998). In this case these were *Argyrosomus japonicus*, *Monodactylus falciformis*, *Rhabdosargus holubi*, *Stolephorus holodon* and the family Mugilidae. Two of other three families can be eliminated, since they are deliberate inhabitants of the surfzone. *Trachinotus* deliberately feeds in the swash zone of the surf edge, while *Iso natalensis* is largely found in the surfzone. The single Tripterygiid was the only larva collected that could be classed as "out of place".

The overtopping study, while part of the brief of this study, was also stimulated by an occurrence at the Mhlanga Estuary, recorded on 13 September 2002, when 6 tiny blacktail *Diplodus sargus*, (15-17 mm SL) and 7 *Iso natalensis*, were netted in the lagoon. A huge sea was running at the time, and vigorous overtopping was being attended by several alert birds, including Little Egret and Black-headed Heron. Since these two species of fishes are not normally associated with KZN semi-closed estuaries, it appeared to be a case of involuntary overtopping caused by the very rough seas, and perhaps the attendant birds were an indication of the risk involved.

The present study has undoubtedly revealed that overtopping of the berm/sand bar does provide a migration route for estuarine-dependent fish to enter the estuary. On one occasion, a direct link was recorded, as catches were taken from waves that were washing over the berm into the estuary. Past reports have used indirect techniques, such as collecting fish larvae/juveniles from isolated pools on the sand bar following overtopping events into the estuary (Cowley *et al*, 2001). This investigation is therefore the first to provide conclusive evidence that juveniles enter the estuary via overtopping.

The dominance of estuarine dependent species in the catches, almost to the exclusion of other marine species, demonstrates a remarkable ability of these small larvae to manoeuvre and orientate, in the surfzone, for the purpose of recruitment to estuaries. Also noticeable, was the extremely uniform size cohort of the *R. holubi* juveniles. The cohort matches larvae of the sparids *Diplodus sargus* and *R. sarba*, both of which one of us (A. Connell) has reared from eggs in the aquarium. For these two species, this size represents a juvenile of about 28 days post hatch. This single cohort in the catches is remarkable, and begs the question whether adult spawning, and hence juvenile arrival at the beaches, for recruitment to estuaries, might be linked to the lunar cycle in this species.

There was obviously a large difference between the numbers of juveniles collected at Illovo/Karridene, and Mhlanga/Mdloti. A comparison of their physico-chemical conditions reveals that they were approximately the same on each sampling occasion. The main differences between the systems were observed when comparing the slopes of the berms, the sand compositions and the chlorophyll *a* contents within the two estuaries.

Table 7.8: Ichthyofauna catch composition collected at the Mhlanga (Mh), the Lovu (Lo), the Mdloti (Md) and Msimbazi (Ms) estuaries. *preflexion larva. Lovu 11/9/2003, mouth closed, overtopping; Lovu 12/9 mouth closed, surf-edge backwash sampling; 26/9 Lovu mouth open, sampled on seaside edge of mouth in shallow waves sweeping into estuary; 10/10 and 27/10 Lovu, as per 26/9; 10/11 and 24/11, Msimbazi estuary, mouth closed, surf-zone backwash sampling.

Family	Species	13/8 **Mh**	27/8 **Mh**	10/9 **Mh**	11/9 **Lo**	12/9 **Lo**	26/9 **Mh**	26/9 **Lo**	10/10 **Mh**	10/10 **Lo**	27/10 **Mh**	27/10 **Lo**	10/11 **Mh**	10/11 **Ms**	24/11 **Md**	24/11 **Ms**
Sciaenidae	*Argyrosomus japonicus*	–	–	–	–	–	–	1*	–	–	–		–			–
Atherinidae	*Iso natalensis*											1				
–	?*Etrumeus* sp.	–	2	1	–	–	–	–	–	–	–		–			–
Monodactylidae	*Monodactylus falciformis*	–	–	–	–	3	–	–	–	–	–	4	–			–
Mugilidae	Unidentified	1	–	–	14	48	–	18	–	9	–	3	–			–
Sparidae	*Rhabdosargus holubi*	–	–	–	9	93	–	100	–	5	–	94	–	9		3
Engraulidae	*Stolephorus holodon*	–	–	–	4	2	–	10	–	1	–		–	2		2
Synodontidae	Unidentified	–	1	–	–	–	–	–	–	–	–		–			–
Carangidae	**Trachinotus** sp.													1		10
Tripterygiidae	Unidentified											1				
Total no. of fish caught		**2**	**4**	**1**	**27**	**146**	**0**	**129**	–	**15**	–	**103**	–	**12**		**15**
Invertebrates	Nematodes	1	2	2					–	–	–		–			–
	Gastrosaccus bispinosa				20+	lots										
	Emerita austroafricana	4	-	1	lots	lots		lots	–	–	–		–			–
	Green crab	-	-						–	–	–		–			–
	Penaeid larvae							1		1				2		4
No. of nets used		2	2	2	1	1	1	1	1	1	1	1	1	1	1	1

Chlorophyll *a* content is used as an indicator of phytoplankton biomass. Phytoplankton is one of the main food sources of estuarine zooplankton, upon which larval fishes would feed (Whitfield, 1998, Wooldridge, 1996). These results, however, indicate that phytoplankton biomass was not a major determinant in fish recruitment via overtopping, as the Mhlanga Estuary had a much higher chlorophyll *a* content than the Illovo. Chlorophyll *a* in the Mhlanga Estuary reached levels of > 70 $mg.m^{-3}$, while at the Illovo, the highest reading was approximately 12 $mg.m^{-3}$.

The most obvious physical difference between the four estuaries, is that both the Mhlanga and Mdloti have steep, high-energy beaches, while the Illovo and Msimbazi have wider-bermed, less steep, low-energy beaches. They can be defined as such based on their sediment structure and the slope of the beach face (Brown and McLachlan, 1990). The berm of the Mhlanga and Mdloti have coarse sediment, while that of the Illovo and Msimbazi are dominated by small sand particles making up a fine sediment. In high-energy systems the wave energy is reflected off the beach face, while in low-energy systems most of the wave energy is consumed in the surf zone (Brown and McLachlan, 1990).

The Mhlanga and Mdloti both have a steep beach face, typical of a high-energy beach, as uprunning swashes drain into the coarse sediment and hence reduce backwash (Brown and McLachlan, 1990). More sand is carried up the beach with the incoming wave, than back again, resulting in a steep slope (Brown and McLachlan, 1990). Thus the Mhlanga and Mdloti are described as exposed beaches. The Illovo and Msimbazi, on the other hand, are located on protected or sheltered beaches, where the sand tends to remain water-logged because of the sediment's low permeability. Hence, each swash is followed by a backwash (Brown and McLachlan, 1990). This flattens the beach and leads to the formation of a gentle slope.

Beach energy, appears to be playing a critical role in the assemblage of estuarine-dependent juveniles adjacent to a particular estuary. Fish juveniles waiting to enter an estuary appear to prefer the dynamics of a low-energy sandy beach, especially the gentler slope of the berm, and hence these estuaries will have far more recruitment taking place. Overtopping on such low energy beaches, would then seem also to be more likely than associated with estuaries adjacent to higher energy beaches, since under low energy conditions, overtopping must be less risky.

8. BIRDS

The bird faunas of the two systems were dominated by piscivorous species rather than typical waders which is in keeping with the generally non-tidal nature of these estuaries. The major piscivorous species were goliath herons *Ardea goliath* (singles on both systems), fish eagles *Haliaeetus vocifer* (pairs seen on both systems),white breasted *Phalacrocorax carbo* and reed *P.africanus* cormorants (usually about 6 on each system), pink backed pelican *Pelicanus rufescens* (occasional on the Mdloti), darters *Anhinga rufa*, giant *Megaceryle maxima*, pied *Ceryle rudis* and malachite *Alcedo cristata* kingfishers (singles or pairs regularly seen on both systems). During summer several hundred common *Sterna hirundo* and swift *Sterna bergii* terns used the sand bar at the Mdloti, which is wider and where there was generally less human activity than at Mhlanga, as a rest area. Greenshanks *Tringa nebularia*, grey plovers *Pluvialis squatarola* and common sandpipers *Actitis hypoleucos*, which are considered as wading species, were recorded on the Mhlanga but the numbers never reached double figures.

Begg (1978) refers to a variety of published and unpublished information which, in addition to the above species, indicates that on the Mhlanga ospreys *Pandion haliaetus* were present in the early 80's while "a variety of waders" as well as "flamingos (species not stated) were often seen in earlier years". Records from the 40's refer to black heron *Egretta ardesiaca* and breeding white backed night heron *Gorsachius leuconotus* and African finfoot *Podica senegalensis*. The latter two species are shy and cryptic and likely to be adversely affected by any sort of disturbance. The implication of a "variety of waders" is less simple to extrapolate to present conditions.

In relation to the Mdloti, Begg *(loc. cit.)* refers on the basis of personal communications to "an impressive amount of estuarine birdlife" including 66 species recorded over a five year period. This included large numbers of seabirds, especially common terns *Sterna hirundo*, using the sandbank for roosting. He refers also, without specifying, to a "large variety of waders and piscivorous" species. Without indications of numbers this does not really permit comparison of past with present conditions.

The hypothesis that an open mouth condition would increase the diversity and abundance of the avifauna, and particularly species using inter-tidal areas, was not supported during the period of the survey. Much of the areas in both systems exposed after breaching became supra-tidal rather than inter-tidal and the resulting desiccation would have rapidly killed any of the small, shallow-burrowing species that might have occurred in these areas when submerged.

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Water quality modelling of estuaries

JH Slinger, S Taljaard, M Rossouw and P Huizinga

The development of estuarine water quality monitoring expertise was identified as a priority research requirement by the **Co-ordinated Programme on Decision Support for the Conservation and Management of Estuaries**. This project investigated the suitability of the one-dimensional Mike 11 Water Quality Model to predict water quality in South African estuaries. The two estuaries selected were the Berg and the Swartkops, both of which are relatively long and narrow with permanently open mouths which suit one-dimensional modelling. In addition, both are data-rich by South African standards. The model showed good correlation between measured and simulated temperature and dissolved oxygen (DO), even predicting the low DO levels in the upper reaches of the Berg Estuary in the summer, although the high variability near the mouth was underestimated. This is possibly due to insufficient data on the inshore marine environment. One area of difference between these estuaries and those of the Northern Hemisphere is the sediment oxygen demand. It was postulated that this could be the result of a relatively small freshwater input. The effect of the 'black tide' on the Berg Estuary was modelled successfully. This indicates that Mike 11 can also be used for linking water quality to biological processes.

Nutrients such as soluble reactive phosphate and silicate were strongly correlated to salinity, but total dissolved nitrogen showed no correlation to any parameter either measured or modelled. Another current limitation is that the model cannot, in its present form, simulate bacterial water quality.

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