Appendix A

Availability of Data on the East Kleinemonde Estuary

Data availability on sediment dynamics, hydrodynamics and water quality

DATA REQUIRED	AVAILABILITY	COMMENT
Simulated monthly runoff data (at the head of the estuary) for present state, reference conditions and the selected future runoff scenarios over a 50 to 70 year period	Provided by Denis Hughes, Institute for Water Research, Rhodes University	Provided for 1920-2002. Low confidence as no runoff data are available.
Simulated flood hydrographs for present state, reference conditions and future runoff scenarios: • 1:1, 1:2, 1:5 floods (influencing aspects such as floodplain inundation) • 1:20, 1:50, 1:100, 1:200 year floods (influencing sediment dynamics)	No catchment flow gauge.	
Aerial photographs of estuary (earliest available year as well as most recent)	Yes	Collected for DEAT
Continuous water level recordings near mouth of the estuary	Water level data available from 2005.	DWAF
Mouth observations	Mouth observations available from 2002.	Collected by Paul Cowley (SAIAB)
Longitudinal salinity and temperature profiles (in situ) collected over a spring and neap tide during high and low tide at: • end of low flow season (i.e. period of maximum seawater intrusion) • peak of high flow season (i.e. period of maximum flushing by river water)	Available for 2006.	Collected by Bernard Mackenzie (SAIAB)
Water quality measurements (i.e. system variables, and nutrients) taken along the length of the estuary (surface and bottom samples) on a spring and neap high tide at: • end of low flow season • peak of high flow season	Available for 2006.	Collected by Bernard Mackenzie (SAIAB)
Measurements of organic content and toxic substances (e.g. trace metals and hydrocarbons) in sediments along length of the estuary.	Available for 2006.	Collected by Tris Wooldridge (NMMU)
Water quality (e.g. system variables, nutrients and toxic substances) measurements on river water entering at the head of the estuary	No catchment water quality data.	
Water quality (e.g. system variables, nutrients and toxic substances) measurements on near-shore seawater	No nearshore water quality data.	

Data availability on microalgae

DATA REQUIRED	AVAILABILITY	COMMENT
Chlorophyll a measurements taken at 5 stations at the surface, 0.5 m and 1 m depths. Cell counts of dominant phytoplankton groups i.e. flagellates, dinoflagellates, diatoms and blue-green algae. Measurements must be taken coinciding with typically high and low flow conditions.	Data from WRC K5/1581 report in microalgae chapter.	Chlorophyll *a* data based on monthly survey. Species composition taken quarterly.
Intertidal and subtidal benthic chlorophyll a measurements taken at 5 stations (at least). Epipelic diatoms need to be collected for identification. These measurements must to be taken coinciding with a typical high and low flow condition (in temporarily closed estuaries measurements must include open as well as closed mouth conditions).	Data from WRC K5/1581 report in microalgae chapter.	Microphytobenthos Chlorophyll *a* and community composition data collected quarterly.
Simultaneous measurements of flow, light, salinity, temperature, nutrients and substrate type (for benthic microalgae) need to be taken at the sampling stations during both the phytoplankton and benthic microalgal surveys.	Data from WRC K5/1581 report.	Collected by Bernard Mackenzie (SAIAB)

Data availability on macrophytes

DATA REQUIRED	AVAILABILITY	COMMENT
Aerial photographs of the estuary (ideally 1:5000 scale) reflecting the present state, as well as the reference condition (if available). Available orthophoto maps	1934 (1:20 000) 1938 (1:20 000) 1956 (1:30 000) 1973 (1:30 000) 1990 (1:30 000) 1998 (1:60 000) 1970 (orthophoto)	Surveys and mapping medium confidence since smaller areas not covered in sufficient detail.
Number of plant community types, identification and total number of macrophyte species, number of rare or endangered species or those with limited populations documented during a field visit.	Yes All data available from quarterly surveys in 2005 and monthly surveys in 2006.	High confidence
Permanent transects: - Measurements of percentage plant cover along an elevation gradient	Data available for three transects from quarterly surveys in 2005 and monthly surveys in 2006.	Plant characteristics sampled in 2005. High confidence
- Measurements of salinity, water level, sediment moisture content and turbidity		Water column sampled in 2006.

Data availability on invertebrates

DATA REQUIRED	AVAILABILITY	COMMENT
Compile a detailed sediment distribution map of the estuary Obtain a detailed determination of the extent and distribution of shallows and tidally exposed substrates. During each survey, collect sediment samples for analysis of grain size [1] and organic content [2] at the six benthic sites.	Sediment particle size analysis available for four sampling trips in 2006.	Complete particle size range available.
Surveys to determine salinity distribution pattern along the length of the estuary, as well as other system variables (e.g. temperature, pH and dissolved oxygen and turbidity) are required for different seasons and for different states of the tide[3]**.** Seasonal (summer winter) physico-chemical data are also required for each of the six benthic sampling sites	Data for salinity, temperature, oxygen, pH, turbidity available for four sampling trips in 2006.	Complete sets of data.
Collect a set of six benthic samples each consisting of five grabs. Collect two each from sand, mud and interface substrates. If possible, spread sites for each between upper and lower reaches of the estuary. One mud sample should be in an organically rich area. Species should be identified to the lowest taxon possible and densities (animal m^{-2}) must also be determined. Seasonal (summer winter) data sets for at least one year are required, preferably collected at spring tides.	Five sites along the estuary each with nine replicates available. Three series available from 1998-1999; all species identified, not quantified. Four series available from 2006; all species identified and quantified.	Good sets of data available.
Collect two sets of beam trawl samples (i.e. mud and sand). Lay two sets of five, baited prawn/crab traps overnight, one each in the upper and lower reaches of the estuary. Species should be identified to the lowest taxon possible and densities (animals m^{-2}) must also be determined. Survey as much shoreline as possible for signs of crabs and prawns and record observations. Seasonal (summer winter) data sets for at least one year are required, preferably collected at spring tides.	Beam trawl surveys not undertaken.	
Additional trip(s) may be required to gather data on the occurrence/recruitment and emigration of key species such as *Callianassa* and *Upogebia* which require a connection to the marine environment at specific times of the year.	Detailed photographic survey of *Callianassa* abundance undertaken during BSc (Hons) project.	Highlights subtidal distribution of *Callianassa*.
Collect three zooplankton samples, at night, one each from the upper, middle and lower reaches of the estuary. Seasonal (summer/winter) data sets for at least one year are required, preferably collected at spring tides.	Zooplankton data available from William Froneman (RU).	Data set is incomplete.

Data availability on fish

DATA REQUIRED	AVAILABILITY	COMMENT
In a small estuary (<5km) collect at minimum three sets of samples from the lower, middle and upper reaches of the estuary. The samples should be representative of the different estuarine habitat types, e.g. *Zostera* beds, prawn beds, sand flats. At least one of the sample sets needs to be in the 0 to 10 ppt reach of the estuary. Sampling should be representative of small fish (seine nets) and large fish (gill nets). In a larger estuary (>5km) sampling can either be at fixed intervals (every 2km) or have the upper, middle and lower reaches subdivided into at least a further three sections each. The samples should be representative of the different estuarine habitat types, e.g. *Zostera* beds, prawn beds, sand flats. At least one of the sample sets should be in the 0 to 1 ppt reach of the system. Sampling should be representative of small fish (seine nets) and large fish (gill nets). Sampling should be done during both the low and the high flow season for the full extent of the system (as far as tidal variation) to allow for predictive capabilities.	Minimum of three small seine (30m x 2m x 5mm bar mesh) and three large seine (50m x 2m x 15mm bar mesh) net samples in the lower, middle and upper reaches. Two gill nets, each panel 10m x 2m, with stretch mesh sizes of 45mm, 75mm and 100mm were set in the lower, middle and upper reaches on each sampling occasion. Estuary has been sampled a minimum of summer and winter during every year from 1994 - ongoing.	Comprehensive data on fish recruitment (including ichthyoplankton), distribution, relative abundance, population sizes and productivity.

Data availability on birds

DATA REQUIRED	AVAILABILITY	COMMENT
Undertake one full count of all water-associated birds, covering as much of the estuarine area as possible. All birds should be identified to species level and the total number of each counted.	Historic data – single 1980 summer count.	
Seasonal (summer winter) data sets for at least one year are required. If this is not possible, a minimum of four summer months and one winter month will be required (decisions on the extent of effort required will depend largely on the size of the estuary, extent of shallows present, as well as extent of tidally exposed areas).	Current data available from 72 bird counts conducted between March 2005- November 2006.	MSc thesis (in progress) Anja Terörde

Appendix B

Specialist Report: Simulated Catchment Hydrology

DA Hughes
(Institute for Water Research, Rhodes University)

INTRODUCTION

The purpose of this report is to explain the process and present the results of applying the Pitman monthly rainfall-runoff model to simulating three development scenarios in the East Kleinemonde River in the Eastern Cape Province. This report follows on from the earlier report on the simulation of natural flows in the river.

A small modification was made to the model setup for the natural flows and the simulated natural mean annual runoff (MAR) is now 2.856 x 10^6 m^3 (it was some 2.77 x 10^6 m^3).

The three development scenarios to be simulated are:

- Present day situation of 300 developed plots with the water supply originating from the Wellington Dam (capacity of 0.206 x 10^6 m^3) supplemented by borehole water abstractions. Wellington Dam is situated on a tributary of the East Kleinemonde River with a catchment area representing approximately 9% of the total catchment area.
- Future scenario (A) of 1010 developed plots with the water supply originating from a new dam constructed on the main river. The assumption is made that the catchment area feeding the dam will represent about 67% of the total catchment area and that the capacity of the dam will be determined appropriately, given the patterns of inflow and the demand.
- Future scenario (B) of 1010 developed plots with the water supply originating from an off-channel storage fed through intermittent pumping from the main river (probably using some type of low level weir). The assumption is that pumping will remove most of the low flows but not impact very much on the short-duration higher flows.

WATER CONSUMPTION PATTERNS

There are three sources of information on the existing water consumption patterns. The first is the Ndlambe Municipalities Water Services Development Plan (WSDP) of 2006 that estimated the annual consumption at 82 x 10^3 m^3. The second is a 1998 consultancy report that estimates 130 m^3 per day out of season and 500 m^3 per day in season consumption. The third is based on estimates from the Ratepayer's Association, which suggests 120 to 160 m^3 per day out of season and 200 to 350 m^3 per day in season consumption.

Table 1 provides a summary of the monthly distribution of demand using an out-of-season daily consumption of 140 m^3 and an in-season consumption of 275 m^3. It has been assumed that the season is all of December, part of January and part of April.

Table 1. Assumed demand based on the current level of development.

Month	**Consumption (10^3 m^3)**	**Fraction of annual demand**
Oct	4.34	0.076
Nov	4.20	0.074
Dec	8.53	0.150
Jan	5.15	0.091
Feb	3.92	0.069
Mar	4.34	0.076
Apr	4.88	0.086
May	4.34	0.076
Jun	4.20	0.074
Jul	4.34	0.076
Aug	4.34	0.076
Sep	4.20	0.074
Total	**56.77**	**1.000**

A further assumption is that these figures ignore losses in the system (i.e. are based on measured consumption at the household level) and therefore have been increased by 15%. This gives an estimated annual demand of 65.3 x 10^3 m^3 with a seasonal distribution the same as column 3 in Table 1.

The MAR of the Wellington Dam catchment is assumed to be 260 x 10^3 m^3 (9% of 2.856 x 10^6 m^3). The WR90 publications (Midgley *et al*. 1994) suggest that a dam of the size of Wellington in this region will be able to support a gross demand of approximately 88 x 10^3 m^3. The indications are that the dam should be able to support the existing demand. However, this assumes that the annual demand is approximately evenly distributed. If the in-season demand were extrapolated to the full year the equivalent annual demand would be over 100 x 10^3 m^3, suggesting that the dam would not always be able to supply the present day demand during the peak summer season. It is understood that this is a reasonable reflection of reality.

The future demand has been taken to be 3.37 times (1010/300) the present day demand and the same seasonal distribution has been used. This translates into an annual demand of approximately 220 x 10^3 m^3. The WR90 publication's approach to estimating the storage requirements for a demand of this size is from a storage placed at 67% of the total catchment area. The MAR at the new dam would therefore be 1.9 x 10^6 m^3 (0.67 x 2.856). To ensure that the peak summer season demand would be satisfied, the equivalent annual demand of the in-season water consumption has been used. To satisfy this demand a dam of approximately 35% of MAR would be needed. In volume terms this represents about 700 x 10^3 m^3. To allow for additional evaporation losses a dam of 1000 x 10^3 m^3 has been used in the simulations for future scenario A.

SCENARIO SIMULATIONS

Present Day Situation

As might be expected with only 9% of the total catchment area above the Wellington Dam, the impact on the natural flows is minimal. The MAR is reduced to approximately 2.748 x 10^6 m^3, a consumption of 108 x 10^3 m^3 (which is reasonably consistent with an abstraction volume of 65.3 x 10^3 m^3), the remaining influence being evaporation losses from the reservoir surface.

Future Scenario A (Instream Dam)

In this situation the MAR is reduced by some 446 x 10^3 m^3 to 2.410 x 10^6 m^3, which is again consistent with a dam occupying 67% of the catchment area and a demand of some 220 x 10^3 m^3. The high flows are not impacted very much, while many of the smaller monthly flows are removed. This is to be expected given the highly variable nature of the regime (a large proportion of the MAR volume represented by a limited number of large volume flow events) and the relatively small size of the dam and water supply demand. Figure 1 illustrates the impact on the 1-month annual flow duration curve.

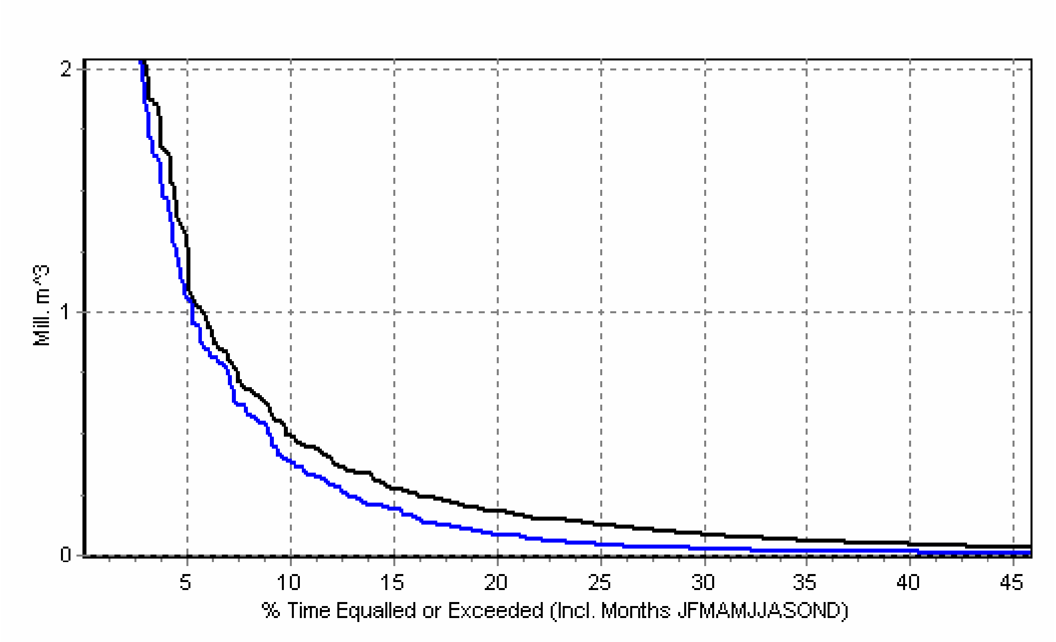

Figure 1. Flow duration curves for Natural (black line) and Future scenario A (blue line).

Future Scenario B (Off-Channel Storage)

This scenario is quite difficult to setup in the model, largely because of the monthly time-step used. The implication is that within the final time series of the model no distinction is made between the flow that occurs as continuous baseflow (and can be pumped) and the flow that occurs as short-duration events (the majority of which would not be able to be pumped unless the in-channel storage is increased). This means that even quite small monthly volumes may not be accessible to the pump as they may occur as very short period events (only a few hours) with flows greater than the pump capacity.

The approach adopted has therefore been to use a baseflow separation approach that attempts to estimate the time series of baseflow (i.e. the proportion of the monthly volume occurring as continuous flow during the month). The method relies on a digital filtering process that uses two parameters in the equation. These parameters were set so that the mean annual baseflow volume would approximately satisfy the water users' demand, also allowing for some evaporation losses from an off-channel storage facility. Calculations indicate that the maximum pumping rate would be some 140 litres s^{-1}, while the average would be about 11 litres s^{-1}.

Figure 2 illustrates that the result has a much lower impact on the moderate flows (10 to 40% frequency of being equalled or exceeded) than future scenario A. The MAR is reduced by some 282 x 10^3 m^3 to 2.574 x 10^6 m^3. It is possible that the estimated baseflow abstraction volume is too low and that not enough has been allowed for the effects of evaporation from the off-channel storage facility. However, the impact of a small additional abstraction volume will not be very great.

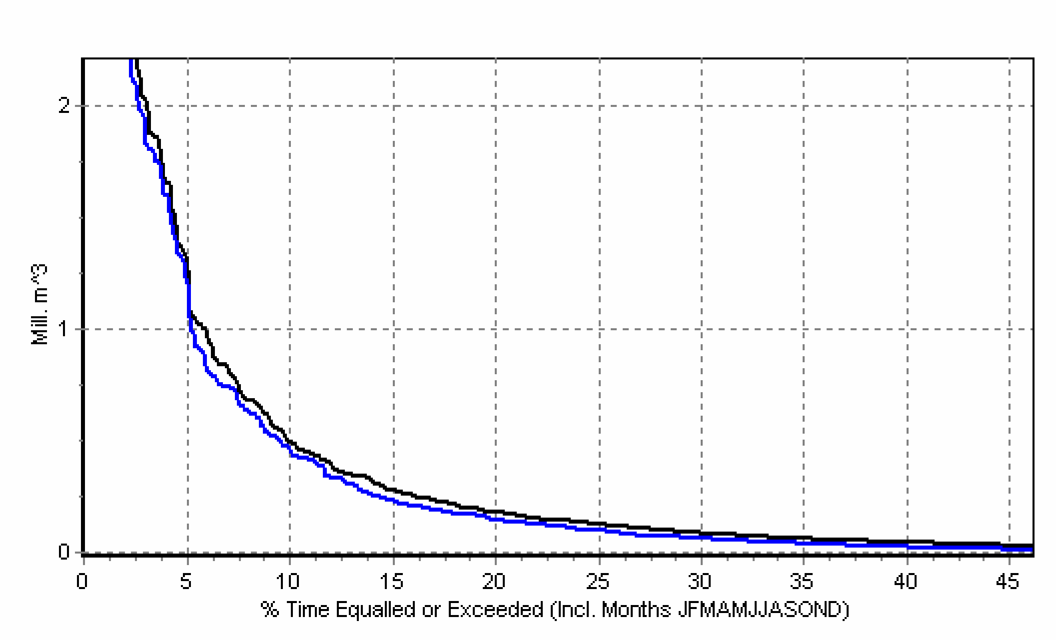

Figure 2. Flow duration curves for Natural (black line) and Future scenario B (blue line).

OBSERVATIONS

It is essential to remember that all the scenarios are based on simulated flows using regional model parameters and that there are no observed flow measurements available with which to confirm the simulated flows. There is therefore a high degree on uncertainty in these results and they should be treated with caution. Clearly, if the simulated natural flows are not representative of the real flow regime of the river, the impacts could be very different. If the flows have been over-estimated the impacts will be greater, while the reverse will be true if the flows have been under-estimated.

REFERENCES

Midgley, D.C., Pitman, W.V. & Middleton, B.J. 1994. Surface water resources of South Africa 1990. Vol. I-VI. Water Research Commission, Pretoria, Reports No. 298/1.1/94 to 198/6.1/94.

Appendix C

Specialist Report: Physical Dynamics

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TG Bornman
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INTRODUCTION

The majority of South African estuaries are isolated from the sea by means of a sand bar. Mouth closure normally occurs during periods of low river inflow coupled with high wave conditions in the marine environment. Of the approximately 250 estuaries recorded along the South African coastline about 70% are classified as temporarily open/closed estuaries (TOCEs) according to Whitfield's classification (1992). TOCEs are predominantly regulated by the amount of river inflow received, the magnitude of which is governed primarily by catchment size and the regional climate that dictates rainfall patterns.

STUDY AREA

The East Kleinemonde Estuary is situated at coordinates 33°32'S and 27°03' E. The estuary has a water surface area ranging widely from approximately 11.6 ha immediately after a mouth breaching event to 35.7 ha when the water level is high during the closed phase. When the estuary mouth is open, the spring high tide covers an area of approximately 26.6 ha. The estuary is 3.7 km long and is approximately 210 m across at its widest section in the lower reaches during the closed phase. The system is shallow with the water depth varying between 1 and 2 m in the deeper sections. During periods of extended mouth closure the water level can rise to 2-2.5 m MSL due to the development of a sand bar at the mouth when extensive back flooding occurs. However, after a mouth opening event the estuary is very shallow with a maximum depth of approximately 1 m.

The catchment of the East Kleinemonde Estuary is estimated to be 43.5 km^2 consisting of a gently sloping high lying region used for pineapple and cattle farming and a relatively undisturbed steep-sloping stream and river valley in the lower reaches.

METHODS

This study reviews the hydrodynamics of the East Kleinemonde Estuary through a combination of field observations and simulated runoff data. Mean annual runoff (MAR) estimates were provided by DA Hughes, IWR, Rhodes University. Daily observations on the mouth state of the East Kleinemonde Estuary were then correlated with the simulated monthly runoff to estimate the flow range that drives mouth behaviour. Recorded estuary water level data, provided by DWAF, were evaluated to get a better understanding of the hydrodynamics of the East Kleinemonde Estuary.

Water level recordings and bathymetric data were used in a volumetric analysis of the East Kleinemonde Estuary to calculate river inflow into the estuary. The simulated monthly runoff data were compared with the calculated inflows derived from the measured water levels in the system to evaluate the method. First estimates were made of seepage losses based on volumetric data.

CURRENT UNDERSTANDING & HYPOTHESES

The relationship between the closing forces (wave energy and sediment availability) and opening forces (river inflow and tidal flows) are unique for each individual estuary. The synergy of these forces defines the duration and frequency of open mouth conditions for temporarily open/closed estuaries (TOCEs).

1. The open mouth condition in small TOCEs is primarily driven by floods and freshettes and to a lesser degree by baseflow. This relationship needs to be quantified for each estuary, but grouping may be statistically possible.
2. Closing forces (wave energy, sediment availability) dominate in small TOCEs because river inflow (base-flows) and tidal flows are low.
3. Small TOCEs require a proportionally larger percentage of their natural MAR compared to permanently open estuaries (POEs) to maintain their structure and function.

4. Semi-closed mouth conditions can develop in small TOCEs and are extremely sensitive to changes in river flow.
5. Baseflow plays an important role in the salinity distributions of TOCEs.
6. Intermittently open estuaries (IOEs) are more sensitive to a reduction in MAR than intermittently closed estuaries (ICEs) but responses are estuary specific and some ICEs can readily change into an IOE.
7. Larger TOCEs tend to stay open longer than smaller ones due to the influence of a larger tidal prism or due to the stronger river flow.
8. ICEs are transformed into IOEs when there is a significant reduction in river inflow. Conversely, increased river flow due the release of treated sewage water can result in an IOE becoming an ICE.

HYDRODYNAMICS PARAMETERS INFLUENCING PHYSICAL DYNAMICS OF THE EAST KLEINEMONDE ESTUARY

Water level data

The Department of Water Affairs and Forestry (DWAF) installed a continuous water level recorder (P4H002) at the East Kleinemonde bridge (Figure 1) in March 2005. The inlet pipe was first installed on 14 December 2004 during a closed phase. When the estuary opened in late December, it was noticed that the inlet pipe was installed too high and a new site on the downstream side of the bridge pillar on the left embankment was identified (33°32’08” S, 27°02’31” E). Water level data have been collected on the system since March 2005. The data are logged electronically at the recorder and offloaded at intervals of about 4-6 weeks and then stored in the DWAF database.

Figure 1. Water level recorder P4H002.

No information was available on whether the water level recorder at the East Kleinemonde Estuary had been corrected to mean sea level (MSL) or not. To validate the measured data, 18 high tide observations for 6 different periods during 2005 to 2006 period were compared with the predicted tide for Port Elizabeth[1]. The water level recordings were found to be an average of 0.04 m lower than the predicted tide. Given the inaccuracy in predicted tides due to local sea conditions and climatic influences, it was not deemed necessary to adjust the water level recorder readings for this study until DWAF confirms the accuracy of the readings.

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[1] East London’s tidal measurements (and predictions) are distorted by local effects such as the fact that it is in an estuary. Port Elizabeth’s measurements were therefore considered the more accurate tidal prediction values for the study.

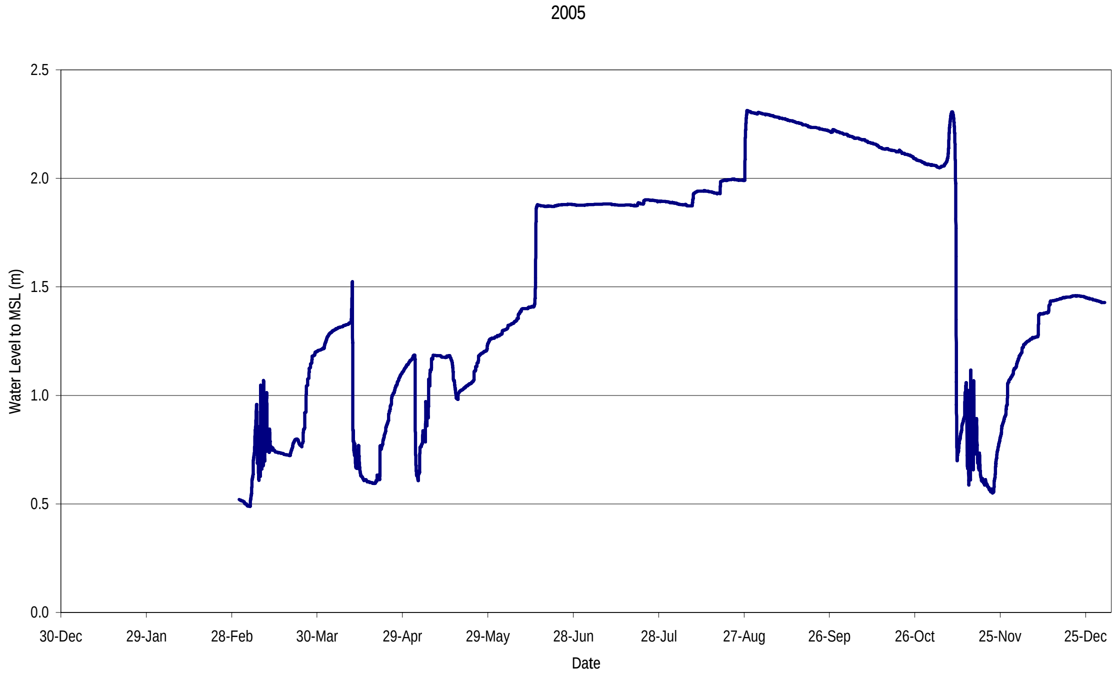

Figure 2. Water levels in the East Kleinemonde Estuary during 2005.

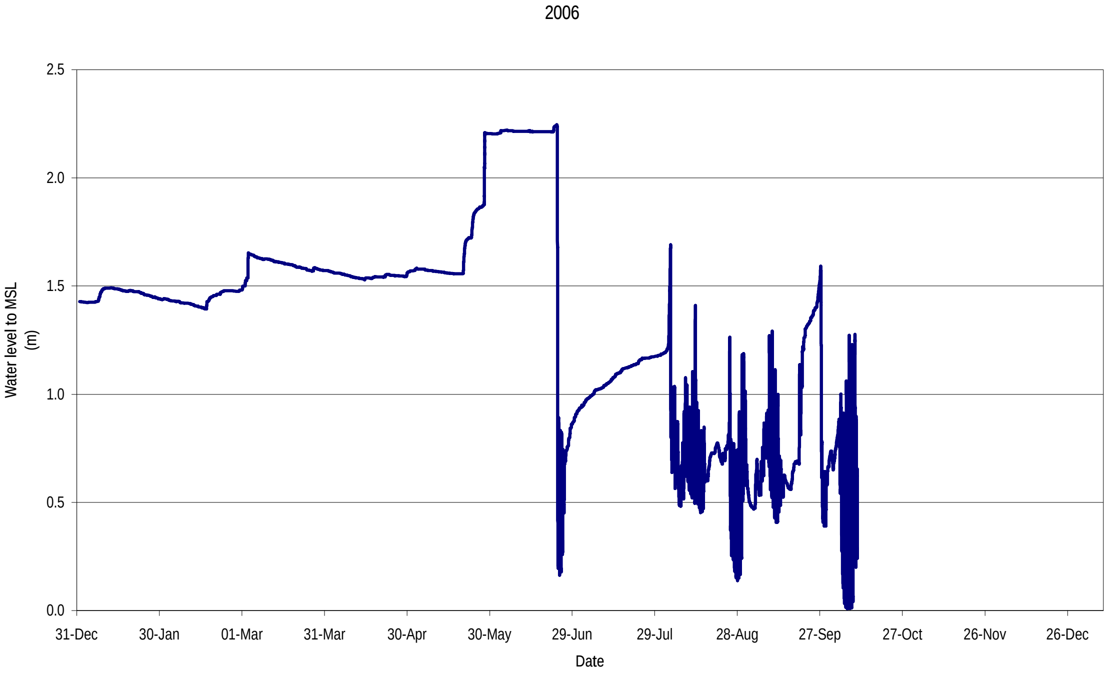

Figure 3. Water levels in the East Kleinemonde Estuary during 2006.

During the measuring period (February 2005 to October 2006) the water level in the estuary varied between 2.3 and 0.18 m MSL (see Figures 2 and 3). An analysis of the water level data and continuous mouth observations shows that the East Kleinemonde Estuary has a tendency to close in the absence of sustained river inflow. Open mouth events seldom last longer than a few days. In the majority of observed breaching events the system closed within a few tidal cycles. In other words the connection to the marine environment is seldom firmly established and is normally only maintained for a few days at a time. This breaching pattern gives a clear indication that the base flow is not sufficient to maintain open mouth conditions in this estuary.

For example, the breaching event of 9 November 2005 shows a rapid increase in water level due to a freshette, a breaching of the system and closure after the water has drained from the system (Figure 4). Tidal variation is only observed at a subsequent breaching event four days later.

If the river inflow is high (> 0.04 $m^3 s^{-1}$) and this elevated flow is maintained for longer than a day, the water level in the estuary generally increases to a level greater than 2.0 m MSL. This contributes to an open mouth condition for longer than a day such as the mouth conditions observed during 23-26 June 2006 and 3-16 August 2006 (Figure 5).

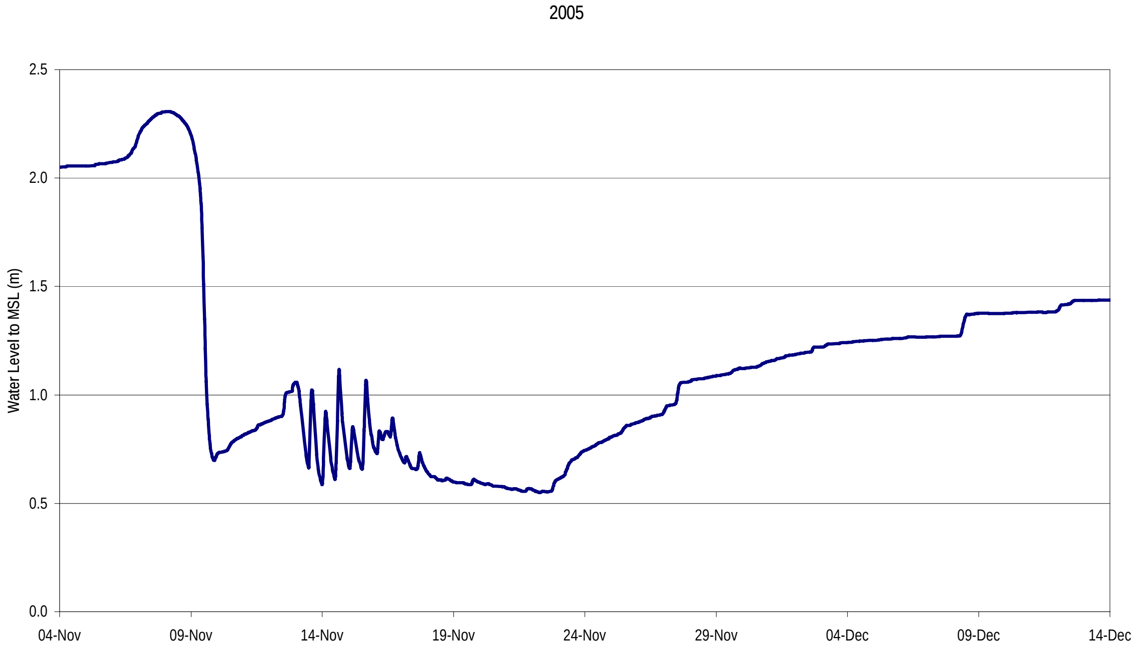

Figure 4. Water levels in the East Kleinemonde Estuary during November 2005.

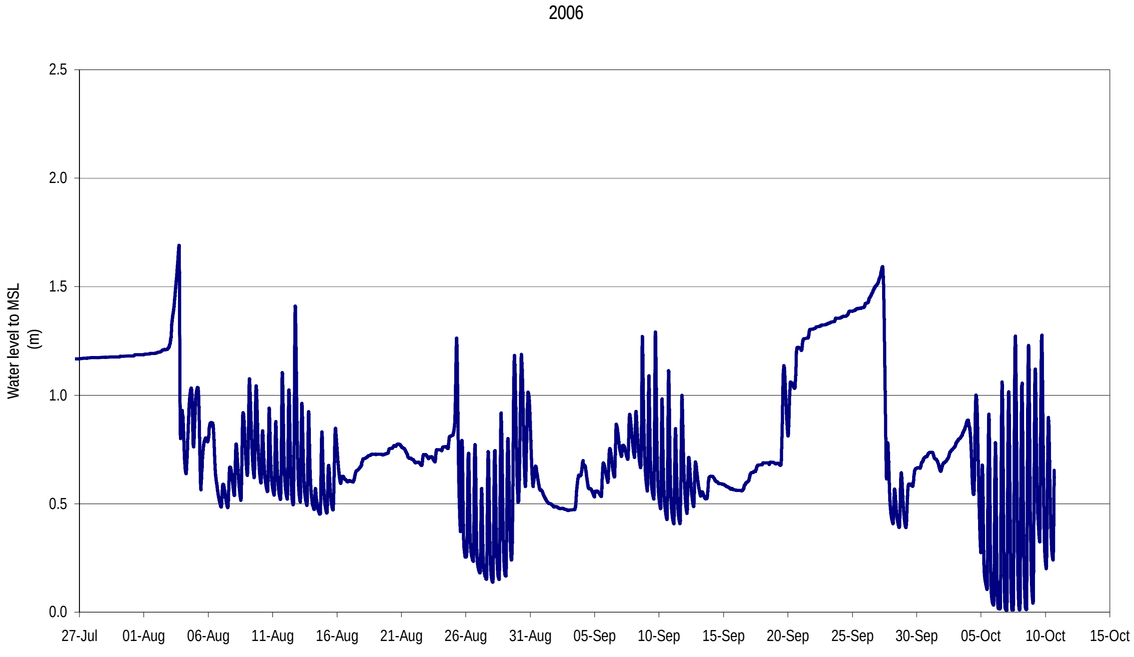

Figure 5. Water levels in the East Kleinemonde Estuary from July to October 2006.

A small estuary such as the East Kleinemonde can be very sensitive to the height of the water level behind the sand bar prior to the breaching. When the water level is high, there is a greater head of water available to scour sediment from the mouth area during a breaching event, thereby increasing the depth. This in turn can result in an increase in tidal flow and in maintaining an open mouth for a slightly longer period. In some instances the breach can occur at such low water levels that very little sediment is scoured from the mouth area and closure occurs with in a few hours.

A close scrutiny of the water level recorder data indicates that what initially appears to be a sustained open mouth event, often consists of a series of rapid breaching and closure events. Figures 3 to 6 illustrate the stop-start nature of the East Kleinemonde Estuary breaching pattern. During these opening events seawater enters the estuary during the flood tide, but is often prevented from freely flowing out during the low tide due to the high level of the sand bar at the mouth. Shortly after such closures the water level recorder data show a step-by-step increase in water level, which is related to marine overwash on high tides when the berm level is still relatively low.

Water levels before breaching events varied between 0.69 and 2.31 m MSL for the period March 2005 to October 2006. These high breaching levels are associated with periods where the mouth stayed closed for an extended time (weeks to months) allowing the sand berm at the mouth to build up to level > 2.3 m MSL. This value of 2.3 m is surprisingly low by comparison with the average of 3.0 to 3.5 m MSL for South Africa's coastline (Perissinotto *et al*. 2004) and may be related to regular overwash which removes the berm crest and prevents berm build-up or localised deflation due to wind erosion in the north-eastern corner of the berm. The lower breaching levels are related to a brief mouth closure of only a few days before the estuary breached again as a result of river inflow or marine overwash.

Table 2. Summary of the breaching events for the period March 2005 to October 2006.

Breached		**Closed**		**Days open**
Date	**Water level**	**Date**	**Water level***	
08/03/05	0.95	13/03/05	0.76	5
11/04/05	1.52	14/04/05	0.63	2
03/05/05	1.19	04/05/05	0.63	1
15/05/05	1.18	18/05/05	1.01	3
07/11/05	2.31	09/11/05	0.70	2
13/11/05	1.06	16/11/05	0.88	3
23/06/06	2.25	26/06/06	0.69	3
03/08/06	1.69	16/08/06	0.61	13
25/08/06	1.25	31/08/06	0.66	6
04/09/06	0.69	05/09/06	0.55	1
07/09/06	0.90	13/09/06	0.54	6
19/09/06	1.14	20/09/06	0.82	1
27/09/06	1.59	29/09/06	0.60	2
04/10/06	0.88			

**Note: The water level for a closing event was taken at the first indication that the tide was no longer freely flowing in the estuary. These values are therefore open to interpretation and a less accurate measure than that listed for a breach, which lists the maximum level reached before a breaching occurs.*

Mouth closure occurred at water levels varying between 0.54 and 1.01 m MSL. The average water level is estimated to be 0.76 m. With the exception of the level reached on 18 May 2005 (1.01 m), closure occurred in a narrow 0.3 m band ranging between 0.54 and 0.88 m. This indicates some consistency in the closing forces at the East Kleinemonde mouth. The lowest water level reached during the recorded period was 0 m MSL in October 2006, after the mouth was scoured deeply by the preceding flood.

Mouth opening events in general were very brief during this study, lasting between 1 and 6 days. Only one breaching event, i.e. that which occurred on 3 August 2006 after the major flood, lasted longer than a week.

River inflow

These data are based on the daily mouth observations and on the simulated flow data provided by D.A. Hughes. Daily observations of the East Kleinemonde record five mouth states: 1) Mouth closed; 2) Mouth closed with little marine overwash; 3) Mouth closed with much marine overwash; 4) Mouth open; and 5) Water trickling out. Table 3 summarises the percentage occurrence of these states from 1993 to 2003.

Table 3. The percentage occurrence (days) of the different mouth states from 1993 to 2003 of the East Kleinemonde Estuary (Cowley unpublished data).

Year	Closed	Little overwash	Much overwash	Open	Trickling out
1993	86.6	10.7	0.3	2.5	0.0
1994	74.0	21.4	1.4	3.3	0.0
1995	42.2	51.5	4.1	2.2	0.0
1996	87.2	10.4	1.6	0.8	0.0
1997	81.1	14.5	1.4	3.0	0.0
1998	97.0	1.9	0.8	0.3	0.0
1999	100.0	0.0	0.0	0.0	0.0
2000	84.2	6.8	1.1	7.9	0.0
2001	71.0	12.1	0.3	15.1	1.6
2002	76.4	6.0	3.0	13.2	1.4
2003	62.7	14.5	5.2	17.5	0.0
Average	**78.4**	**13.6**	**1.7**	**6.0**	**0.3**

Table 4 shows the number of days the East Kleinemonde Estuary was open in a month for the period February 1993 to December 2003. The observed mouth state is recorded as open and should not be confused with states related to overwash or small trickles of estuary water overtopping the berm. The mouth of the Kleinemonde Estuary stayed open for between 1 and 28 days during the observation period.

Table 4. Daily observations of the open mouth status of the East Kleinemonde Estuary.

	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
1993		3	0	0	0	0	0	0	6	2	0	1
1994	1	1	0	0	0	0	0	5	0	0	0	5
1995	8	0	0	0	0	0	0	0	0	0	0	0
1996	0	0	0	0	0	0	0	0	0	0	1	0
1997	0	0	0	3	2	6	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	0	0	0	1
1999	0	0	0	0	0	0	0	0	0	0	0	0
2000	0	0	4	16	0	0	0	0	0	0	9	0
2001	0	0	0	0	0	0	0	11	13	3	21	7
2002	0	0	0	0	0	0	0	16	28	4	0	0
2003	0	0	0	0	21	13	7	0	0	19	4	0

Table 5 gives the summarised average monthly volume of river inflow for the period February 1993 to Sep 2003. The months with open mouth conditions are highlighted in blue in Table 5.

Table 5. Simulated monthly flow volumes for the East Kleinemonde Estuary (million m^3).

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1993	0.008	0.003	0.000	0.009	0.003	0.049	0.016	0.037	0.186	0.058	0.000	0.180
1994	0.120	0.050	0.010	0.000	0.000	0.000	0.000	0.287	0.096	0.000	0.000	1.240
1995	0.454	0.021	0.013	0.126	0.041	0.000	0.000	0.000	0.000	0.003	0.051	0.017
1996	0.000	0.000	0.010	0.003	0.000	0.000	0.004	0.002	0.000	0.000	1.064	0.370
1997	0.005	0.000	0.002	0.258	0.106	0.344	0.113	0.000	0.000	0.001	0.000	0.000
1998	0.000	0.000	0.436	0.147	0.001	0.000	0.019	0.034	0.026	0.010	0.002	0.008
1999	0.003	0.000	0.008	0.003	0.000	0.000	0.130	0.043	0.022	0.339	0.111	0.000
2000	0.389	0.130	0.919	0.306	0.000	0.000	0.000	0.000	0.016	0.085	0.240	0.071
2001	0.045	0.015	0.023	0.040	0.011	0.000	0.145	0.083	0.145	0.047	0.134	0.051
2002	0.002	0.000	0.000	0.004	0.001	0.000	0.333	2.316	1.839	0.373	0.046	0.025
2003	0.003	0.000	0.000	0.033	3.841	1.282	0.027	0.015	0.002			

Table 6 gives the summarised average monthly river flow (in $m^3\ s^{-1}$) for the period February 1993 to September 2003. The months with open mouth conditions are highlighted in blue in the simulated runoff table.

Table 6. Simulated monthly flows for the East Kleinemonde Estuary in $m^3\ s^{-1}$.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1993	0.003	0.001	0.000	0.003	0.001	0.019	0.006	0.014	0.072	0.022	0.000	0.067
1994	0.045	0.021	0.004	0.000	0.000	0.000	0.000	0.107	0.037	0.000	0.000	0.463
1995	0.170	0.009	0.005	0.049	0.015	0.000	0.000	0.000	0.000	0.001	0.020	0.006
1996	0.000	0.000	0.004	0.001	0.000	0.000	0.001	0.001	0.000	0.000	0.410	0.138
1997	0.002	0.000	0.001	0.100	0.040	0.133	0.042	0.000	0.000	0.000	0.000	0.000
1998	0.000	0.000	0.163	0.057	0.000	0.000	0.007	0.013	0.010	0.004	0.001	0.003
1999	0.001	0.000	0.003	0.001	0.000	0.000	0.049	0.016	0.008	0.127	0.043	0.000
2000	0.145	0.054	0.343	0.118	0.000	0.000	0.000	0.000	0.006	0.032	0.093	0.027
2001	0.017	0.006	0.009	0.015	0.004	0.000	0.054	0.031	0.056	0.018	0.052	0.019
2002	0.001	0.000	0.000	0.002	0.000	0.000	0.124	0.865	0.709	0.139	0.018	0.009
2003	0.001	0.000	0.000	0.013	1.434	0.495	0.010	0.006	0.001			

Table 7 is a summary of breaching events and related simulated runoff flow that caused mouth opening for the period February 1993 to September 2003. The table also shows the number of days the system stayed open during a breaching event.

Based on the observed daily mouth-status record, open mouth conditions occurred during 28 months for the 10-year period February 1993 to September 2003 in the East Kleinemonde Estuary. Of the 28 opening events, 22 lasted for longer than a day. The total monthly volume of river water flowing into the estuary that is associated with events ranged between 0.003 and 3.841 million m^3 for the months in which prolonged openings occurred. Some of the prolonged open mouth events were associated with high flows in preceding months and, if one were to exclude these from the association, the estuary seems to stay open for prolonged periods when the river flow is greater than 0.2 million m^3 (or 0.08 $m^3\ s^{-1}$).

Five of the 28 opening events merely drained the system before it closed again. The total monthly volume associated with these latter events ranged between 0.008 and 1.064 million m^3 (or 0.003 and 0.41 $m^3\ s^{-1}$) for the months in which very short openings occurred.

Observed river inflow that is required to breach the East Kleinemonde Estuary corresponds very well with provisional estimates of the required volume of inflow required to breach the system. Based on a surface area of ~ 30 ha the estuary would require between 300 000 and 600 000 m^3 to initiate a breach. This translates to an average monthly inflow of about 0.1 to 0.2 $m^3 s^{-1}$.

Table 7. Summary of breaching events and related simulated runoff to the East Kleinemonde Estuary.

Date		**Simulated flow**		**Days open**	**Comment**
		million m^3	$m^3 s^{-1}$		
1993	Feb	0.003	0.001	3	*Record starts*
	Sep	0.186	0.072	6	
	Oct	0.058	0.022	2	*Remained open due to high flows in preceding month*
	Dec	0.180	0.067	1	
1994	Jan	0.120	0.045	1	
	Feb	0.050	0.021	1	
	Aug	0.287	0.107	5	
	Dec	1.240	0.463	5	
1995	Jan	0.454	0.170	8	
1996	Nov	1.064	0.410	1	
1997	Apr	0.258	0.100	3	
	May	0.106	0.040	2	
	Jun	0.344	0.133	6	
1998	Dec	0.008	0.003	1	
2000	Mar	0.919	0.343	4	
	Apr	0.306	0.118	16	
	Nov	0.240	0.093	9	
2001	Aug	0.083	0.031	11	*Previous month had high flow*
	Sep	0.145	0.056	13	
	Oct	0.047	0.018	3	*Remains open due to high flows in preceding month*
	Nov	0.134	0.052	21	
	Dec	0.051	0.019	7	*Remains open due to high flows in preceding month*
2002	Aug	2.316	0.865	16	
	Sep	1.839	0.709	28	
	Oct	0.373	0.139	4	
2003	May	3.841	1.434	21	
	Jun	1.282	0.495	13	
	Jul	0.027	0.010	7	*Remains open due to high flows in preceding month*

There is a weak relationship between the river inflow and the number of days the system remained open after the breaching event. One of the main reasons for the wide discrepancy between flow and length of opening events is high variability in the closing forces, e.g. storminess and wave heights at sea. This model might also be further refined if measured or simulated daily flow data were used instead of simulated monthly flows.

Bathymetry

Figure 6 illustrates the relative water depths of the East Kleinemonde Estuary relative to MSL. The surface area of the estuary varies between 35 000 after a breaching to 477 000 m^2 when the water level is high in the system (Table 8). Similarly the volume in the estuary varies between 16 000 m^3 and 664 000 m^3 depending on the water level and the height of the sand bar at the mouth (Table 8).

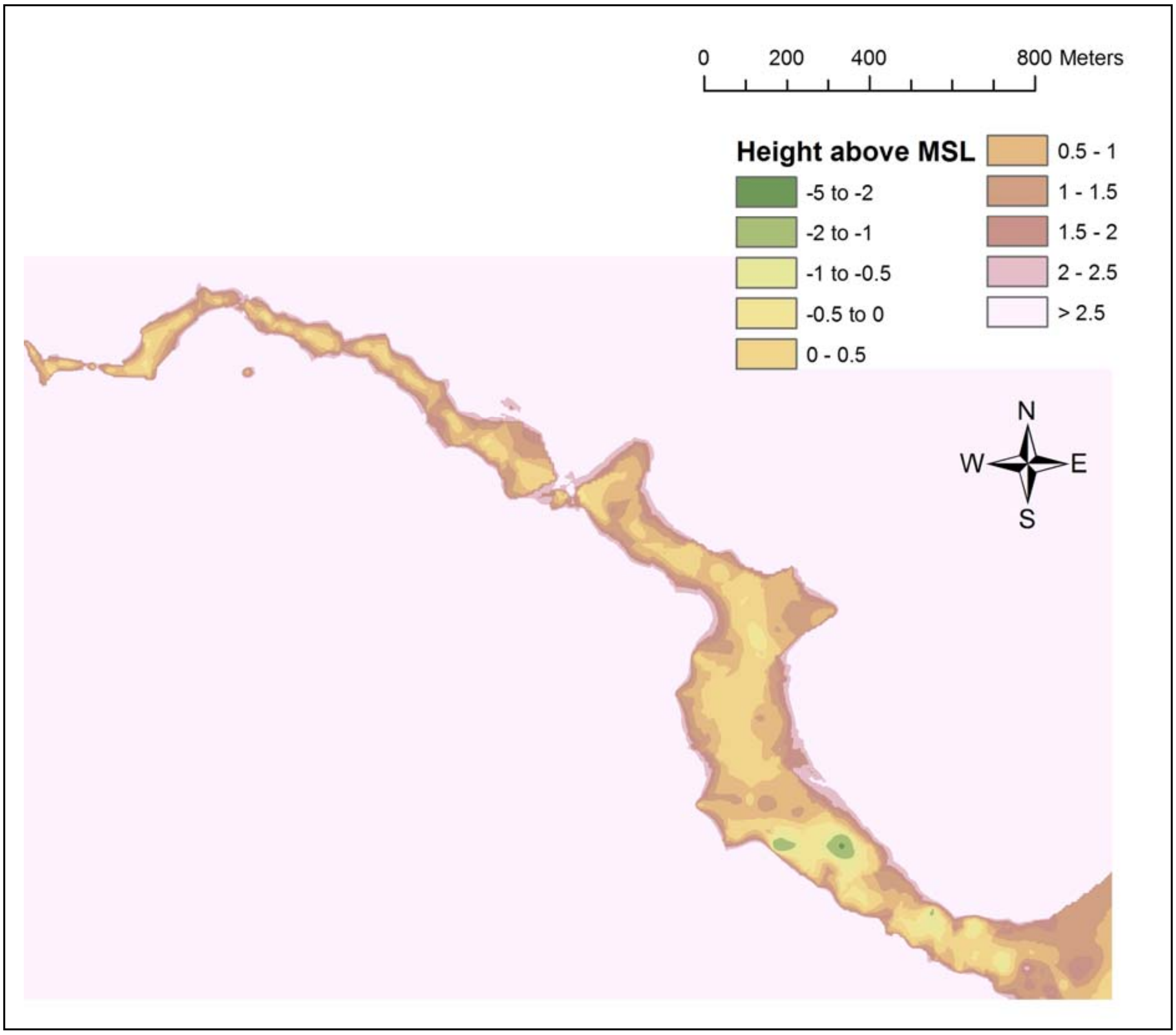

Figure 6. Depth to MSL (m) of the East Kleinemonde Estuary.

Table 8. Surface area and volume of the East Kleinemonde Estuary.

Depth	Area (m^2)	Volume (m^3)
-2.00	143	4
-1.75	676	91
-1.50	1 499	350
-1.25	2 620	838
-1.00	4 238	1 674
-0.75	6 683	2 988
-0.50	11 174	5 100
-0.25	21 021	9 002
0.00	35 061	15 657
0.25	92 100	31 097
0.50	152 222	61 082
0.75	216 815	106 429
1.00	282 489	168 540
1.25	340 066	246 313
1.50	389 171	337 503
1.75	423 427	438 914
2.00	451 973	548 054
2.25	477 488	663 885
2.50	502 313	785 882
2.75	525 539	913 694
3.00	547 695	1 047 239

Volumetric calculation

The volume of water in an estuary depends on the balance between river inflow and seepage, evaporation losses and in- and outflow of groundwater. River inflow to the Kleinemonde Estuary was calculated as a function of the difference in water levels (m) over a set period (day) (see Figure 7 and 8). The volume of the estuary were estimated for different water levels through the relationship $y = 86499x^2 + 94894x - 0.2$.

Median river inflow for 2005 and 2006 was 0.04 $m^3\ s^{-1}$ and 0.03 $m^3\ s^{-1}$ respectively, while average river inflow was 0.12 $m^3\ s^{-1}$ (2005) and 0.14 $m^3\ s^{-1}$ (2006). The calculated median inflow is somewhat higher than the simulated inflow of 0.01 $m^3\ s^{-1}$, while the average simulated inflow is surprisingly similar at 0.12 $m^3\ s^{-1}$, considering that the calculated inflow does not include periods when the mouth was open and flow was high.

The drawback of this method is that the approach is only valid for periods that the estuary mouth is closed. As soon as the estuary breaches, tidal action becomes an additional factor in the equation. This method therefore does not lend itself towards calculating the flow that initiates a breach event or help maintain an open mouth condition in an estuary. However, the method can be used in the absence of any river inflow data to validate simulated river inflow for low-flow periods when gauging stations are often at their least accurate.

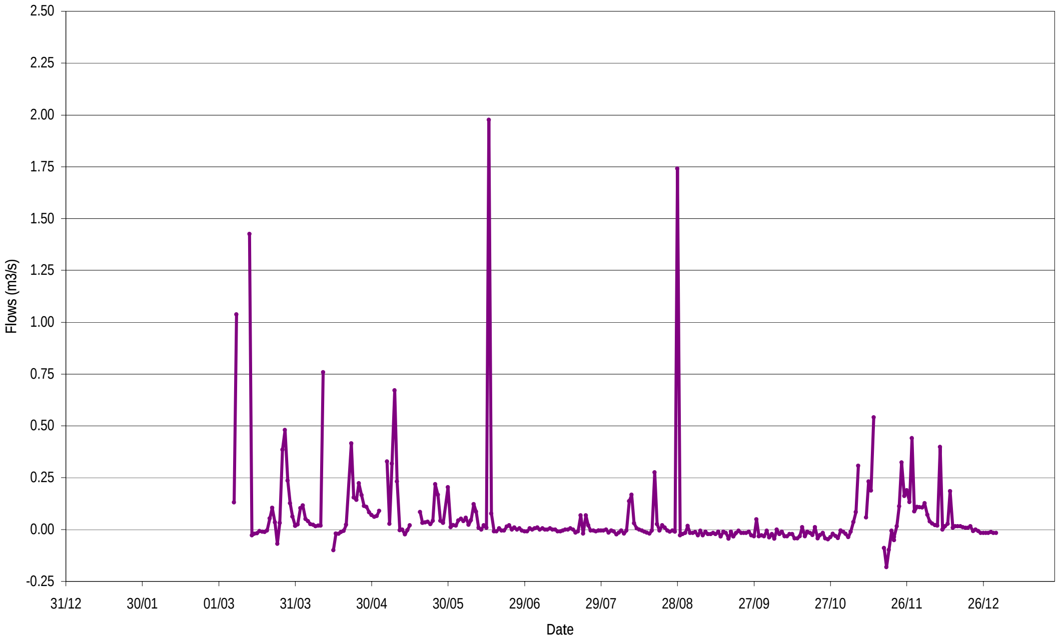

Figure 7. Calculated flow rates ($m^3\ s^{-1}$) for the East Kleinemonde Estuary in 2005.

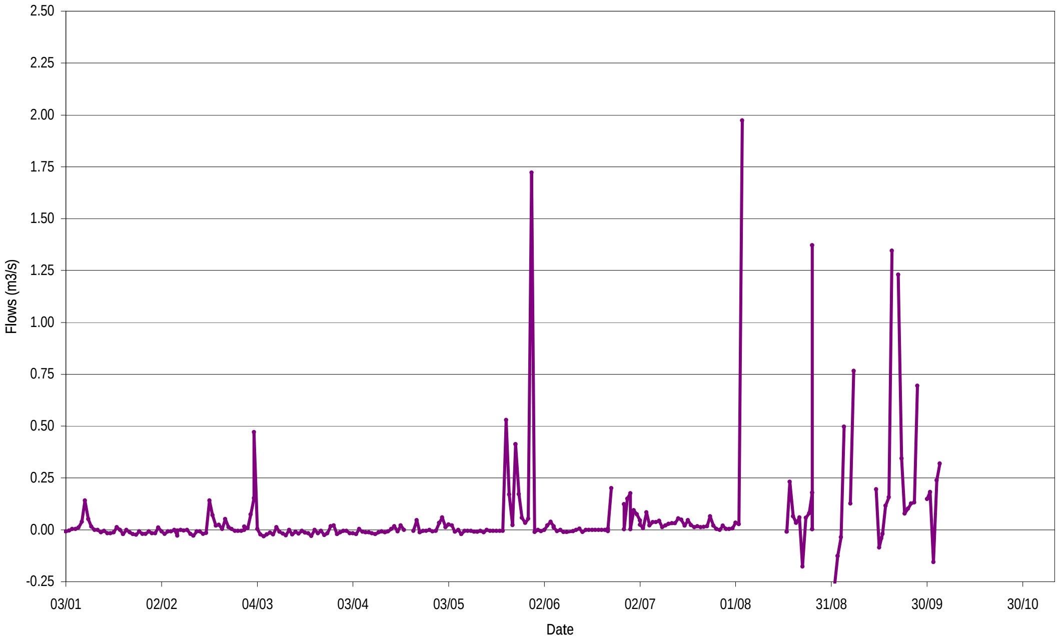

Figure 8. Calculated flow rates ($m^3\ s^{-1}$) for the East Kleinemonde Estuary in 2006.

Negative flows are indicative of evaporation and seepage losses through the berm. For example, in 2005 negative flow between -0.01 and -0.05 $m^3\ s^{-1}$ were calculated for September and October (see Table 9). As spring is still a relatively cool time of the year and the surface area of the estuary is small, these losses can be attributed to seepage through the berm. Unfortunately, some groundwater or river inflow may well have contributed to the volume of water (i.e. water level) in the system and considerable higher seepage and evaporation values would then be expected.

Table 9. Calculated losses from the East Kleinmonde Estuary.

Date	Water level (m)	Seepage or evaporation rate ($m^3 s^{-1}$)
17/06/05 - 21/07/05	1.9	0.000
29/08/05 - 5/11/05	2.3	0.021
12/01/06 - 16/02/06	1.5	0.011
04/03/06 - 17/04/06	1.6	0.011

Evaluations of the water levels after a breaching event indicate that the estuary very seldom breaches and lowers the berm level below 0.5 m MSL. At this level, water losses can be due to an outflow via a small outflow channel (overtopping) or seepage through the berm. Tidal influence also plays a role, as it will reduce (or reverse) seepage during high tides. For the purpose of this study seepage is assumed to be zero $m^3 s^{-1}$ at 0.5 m MSL.

Salinity distribution

Regular salinity measurements were conducted during 2006. Salinity data were, more often than not, showed a homogenous water body while stratification was mostly observed in the upper reaches where freshwater enters the system, but also during an open mouth state. Therefore, to provide a summary of changes in salinity over the measuring period, salinity data were averaged for each measuring trip (note: in certain instances high and low tide observations were conducted on the same day). Figure 10 illustrates the response in average salinity to changes in water levels in the East Kleinemonde Estuary. The average salinity for the measuring period was 23.3 psu, with a range of 14.5 to 31.8 psu. During the closed mouth state the estuary's salinity fluctuated between 15 to 23 psu. Shortly after a breaching event the salinity in the system increased to more than 30 psu. Because of the small size of the estuary, salinity was very responsive to the tidal cycle during the open state with an average increase of 4 psu noted on the flood tide.

Overwash

During 1993 to 2006 the East Kleinemonde Estuary was predominately closed (74.4%), with marine overwash events occurring for 15.4 % of the time and the mouth open for very limited periods (9%). In 2006 the estuary was open for 32.6%, which is significantly above the average for open mouth conditions (see Table 10 and Figure 9). Closed mouth conditions occurred for 41.6% of the time, while overwash occurred for 15.1% of the time.

Four overwash events were also observed (B. Mackenzie pers. comm.) during the study period (see Table 11). Relatively small changes in salinity were correlated with these overwash events, some of which may be related to the sampling method and the averaging of values over the entire estuary.

Figure 9 illustrates the fluctuation in water level, salinity and overwash events in the East Kleinemonde Estuary for 2006. Overwash was observed on 16 February 2006 during the day. It is likely that an overwash occurring during the night of 15 February 2006 elevated the salinity (by 1.1 psu).

The overwash event of 3 March 2006 represents a good example of a significant inflow of marine water at a water level of ~1.5 m MSL. The average salinity in the system increased by about 2.5 psu due to this marine inflow. On 18 March 2006 overwash elevated the salinity by 0.9 PSU.

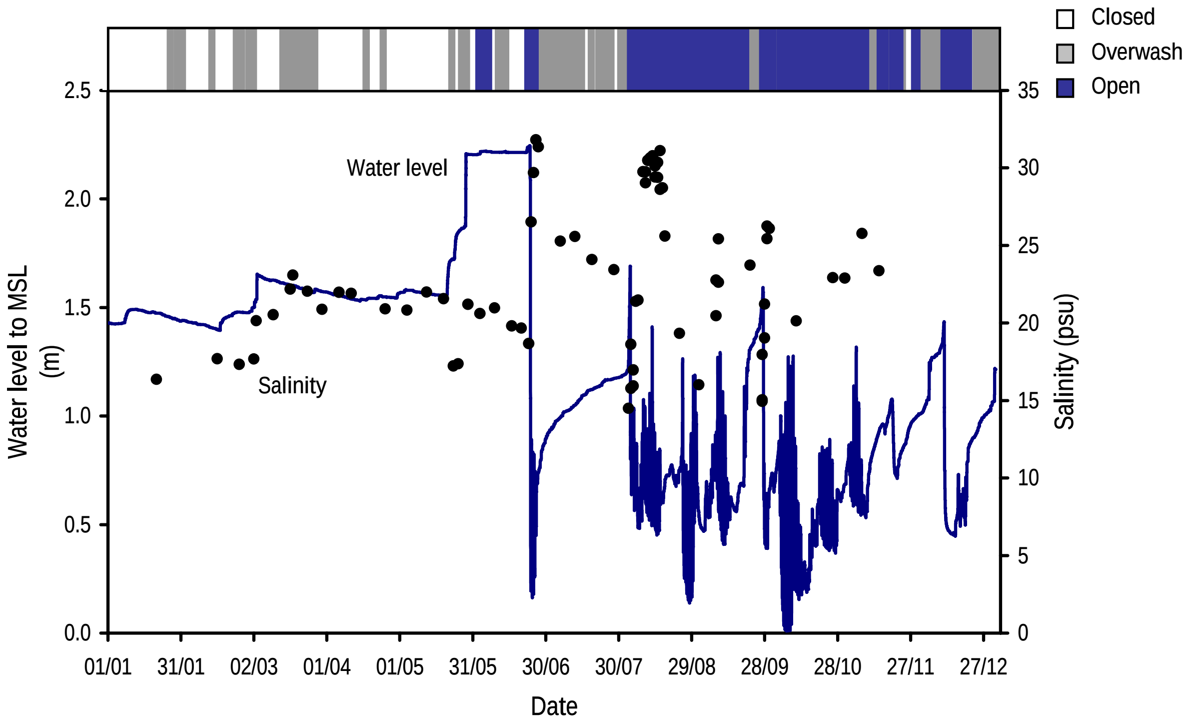

Figure 9. Water levels, salinity and marine overwash events in the East Kleinemonde Estuary for 2006 (water level indicated by blue line, salinity by dots and overwash by bar graph).

Table 10. Mouth status of the East Kleinemonde Estuary from January to December 2006.

Month	Day of month																														
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31
J'06																												1			1
F			1	1												2										1	1	1			
M	1	1	2												1		1	1	1	1	1	1	1	1	1	1	1	1			
A																		1							1						
M																							1				1	2	1		
J			3		4		4				1	1	1	1									4	3	3	3	2	1	1	1	
J	1	1	1	1	1	1	1	1	1	1	1	1		1	1				1			1	1	1	1	1	1				1
A	1	1	2	3	3	3	3	3	3	3	3	3	3	3	3	3	4	3	3	3	3	3	3	3	3	3	3	3	3	3	3
S	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	2	1	2	1	1	1	3	3	3	3	
O	3	1	1	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
N	3	3	3	3	3	3	3	3	2	1	1	1	1	3	3	3	2	2	3	3	3	3	1					3	3	1	
D	1	1	1	1	1	1	2	1		3	3	3	3	3	3	3	3	3	3	3			1	1	1	1	1	1	1	1	2

Table Key:

	Mouth closed
1	Mouth closed with small overwash
2	Mouth closed with large overwash
3	Mouth open
4	Trickling out

Table 11. Observed marine overwash events in the East Kleinemonde Estuary for 2006.

Date of overwash	Salinity change (psu)	From → To	Date of previous measurements	Data Set
15-16/02/06	1.1	16.4 → 17.7	21/01/06	P Cowley
03/03/06	2.5	17.7 → 20.2	02/03/06	B Mackenzie
18/03/06	0.9	22.2 → 23.1	17/03/2006	P Cowley
23/05/06	-4.4	21.6 → 17.2	19/05/06	B Mackenzie
28/05/06	3.8	17.4 → 21.2	23/05/06	P Cowley
26/06/06	2.1	29.7 → 31.8	25/06/06	B Mackenzie
12/07/06	0.3	25.3 → 31.8	06/07/06	B Mackenzie

On the 23 May 2006 an overwash event is related to a big increase in water level but a related initial decrease of 4 psu in salinity. On 28 May 2006 a large overwash event increased the salinity in the system again by 3.8 psu. Figure 9 shows that after the significant decrease in salinity, values recovered again to pre-event values, which indicate that the increase in water level was related to both freshwater and marine inflow during that period. Overwash was also observed during the breaching event of 26 June 2006 and was likely the cause. The overwash event of 12 July 2007 only resulted in a small 0.3 psu increase in salinity.

In summary, overwash contributes to the salinity balance in the East Kleinemonde Estuary. Salinity increases of between 0.3 and 3.8 psu were correlated with observed overwash events. Lower values (<1.0 psu) are associated with small overwash events, while the larger increases (>2 psu) are associated with significant overwash events.

ACKNOWLEDGEMENTS

This study would not have been possible without regular *in situ* observations made by B. Mackenzie.

RECOMMENDATIONS

The following recommendations are based on the findings of this study:

- Continuous water level recorder data provide essential data for investigations into estuarine mouth behaviour. It is strongly recommended that such recorders be installed in all temporarily open/closed systems in South Africa.

- It is recommended that DWAF confirm the benchmark of the East Kleinemonde Estuary water level recorder as this provides the baseline data for a number of further calculations.

- Simulated monthly runoff data do not provide an accurate enough prediction of the mouth state of a small estuary such as the East Kleinemonde. It is recommended that in future, estuarine scientists be provided with simulated daily flows to the estuary, covering at least 15 – 20 years, to assist in developing predictions at the time scales operating in such small systems.

- Volumetric calculations based on water level recordings and local bathymetry can provide data on the volume of water entering the estuary under closed mouth conditions. Unfortunately, this method does not distinguish between runoff from the catchment and overwash from the sea. A further consideration is that this method cannot provide estimates of the volume of flow associated with freshettes and floods when a mouth is open and can therefore not assist in calculating the flows required to maintain an open mouth condition. Volumetric calculations of temporarily open/closed estuaries can be used to validate simulated daily flow in the absence of measured flow.

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Appendix D

Specialist Report:
Coastal Processes and Sediment Dynamics

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TG Bornman
(NMMU, Port Elizabeth)

INTRODUCTION

The hydrodynamics specialist report has discussed mouth conditions, impacts of river flow and tidal flow, which are all intrinsically related to the sediment dynamics and morphology of estuaries. The remaining driver of sediment dynamics in estuaries is linked to coastal processes and this aspect will be examined in more detail here.

Within the coastal zone, there are a number of processes that can transport varying amounts of marine sediment to estuary mouths (e.g. Beck, *et al.* 2004). Marine sediments that have been transported close to the mouth by such processes are then potentially available to be transported into the estuary itself, mainly by means of tidal flow through the mouth (often in conjunction with wave action). There are also a few other processes (such as wind action and wave overwash), which can transport marine sediment directly into the estuary, even when the mouth is closed.

SEDIMENT GRAIN SIZE

Information from CSIR (2000) contains sediment and survey data from the East and West Kleinemonde estuaries. Sand samples taken from the mouth areas of these estuaries indicated an average median sediment grain size (D50) of about 250 µm. Badenhorst (1988) also analysed sediment and survey data from these estuaries. Three vibracores taken near the West Kleinemonde bridge all showed median grain sizes (D50) of about 250 µm.

Sediment samples collected along three beach profiles (Figure 1) from the lower, middle and upper beach during four sampling periods (12 April, 10 July, 12 August and 10 September) in 2006 had a mean particle size of 266 ± 7 µm (n = 40). The medium to fine sand (D50) was moderately to well sorted. No gravel, silt or clay was recorded in the samples. The largest sand fraction was made up of medium to fine sand (Figure 2). On 12 April the sand along the three transects had a higher percentage of medium and coarse grained sand than recorded during the other sampling periods later in the year (Figure 2). No significant difference was recorded between the percentage of medium and fine grained sand in July, August and September. The large flood in August appears to have increased the organic content of the beach sand, especially along Transect 2 situated between the two estuaries (Figure 3).

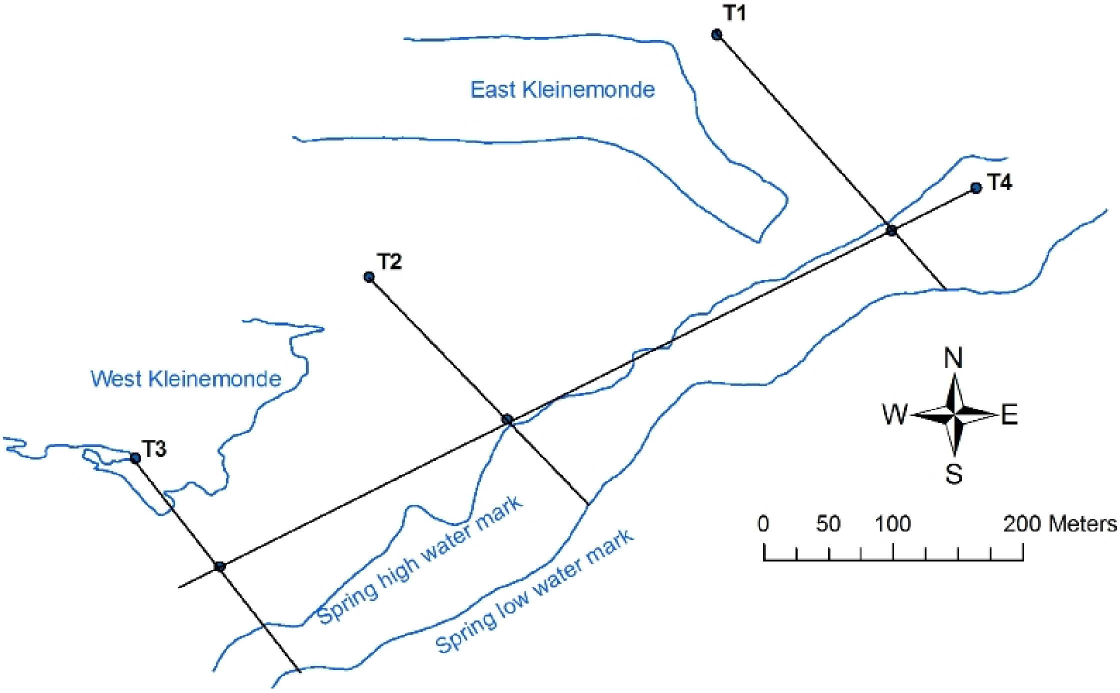

Figure 1. Study site map showing the position of the 4 transects in relation to the East and West Kleinemonde estuaries and the spring low and high water mark of the sea.

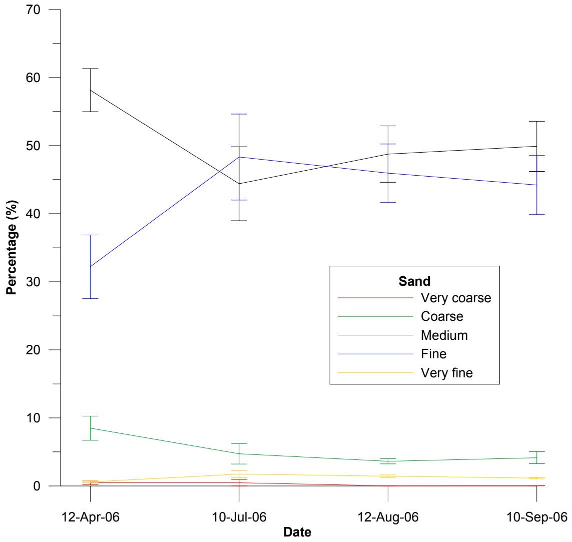

Figure 2. The combined percentage sand fraction for Transects 1, 2 and 3 over the four sampling periods (n = 9)

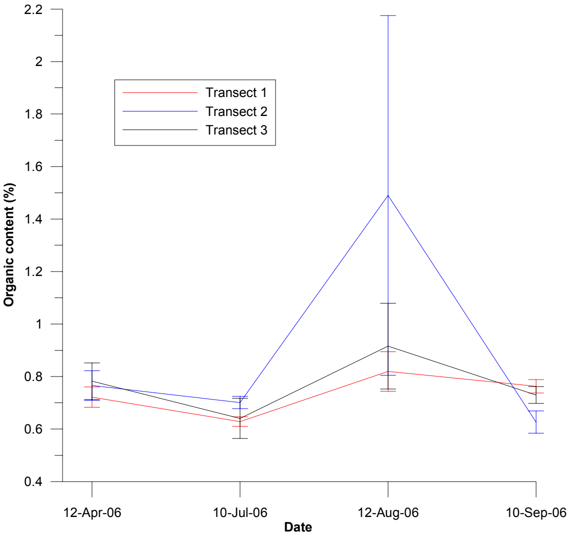

Figure 3. Percentage organic content of the sediment along Transects 1, 2 and 3 during the four sampling periods (n = 3)

BEACH PROFILES

To determine the degree of exposure of a beach, it is important to know both the beach slope and the grain size because these parameters are interlinked (Wiegel 1964).

CSIR (2000) shows that typical beach slopes on the western side of the West Kleinemonde mouth (31 January 1988 and 7 July 1988) between 0 and +2 m MSL were about 1 in 18. From Badenhorst (1988) beach slopes on the western side of the combined mouth area (i.e. both East and West Kleinemonde) were calculated to be about 1 in 21, while beach slopes on the eastern side of the combined mouth area were about 1 in 24.

Beach transects and transects along the crest of the berm were surveyed on 12 April, 10 July, 12 August and 9 September 2006. The three beach profiles are respectively located on the eastern side of East Kleinemonde Mouth, in the middle of the two systems and on the western side of West Kleinemonde mouth (Figure 1). They extended from as near to the dune crest to as far into the sea as practically possible and covered a range of different environmental conditions. (Ideally, the following conditions should be covered: storm event (cold front), strong local onshore wind conditions, typical wind and wave conditions and a very calm day).

Transect 1

Approximately 1.5 to 2 m (vertically) of sand was removed from Transect 1 between April and July 2006. Little change occurred between then and September 2006.

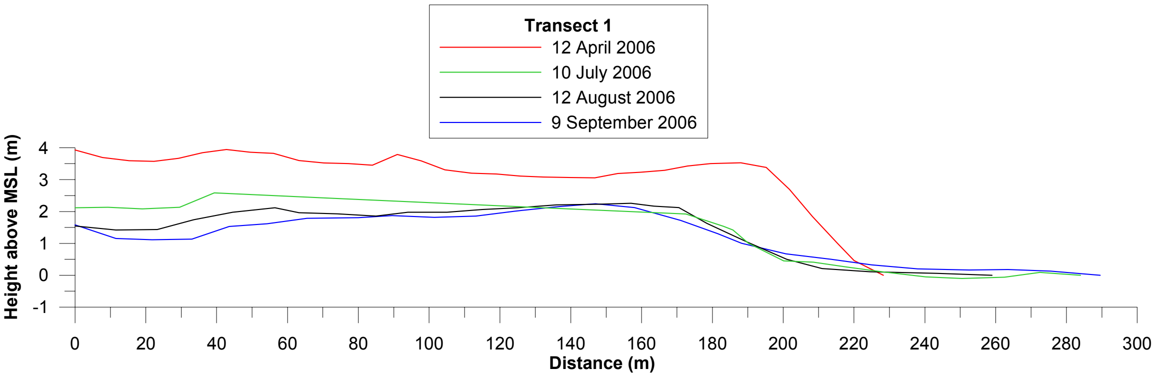

Figure 4. Profile of Transect 1 over time.

Transect 2

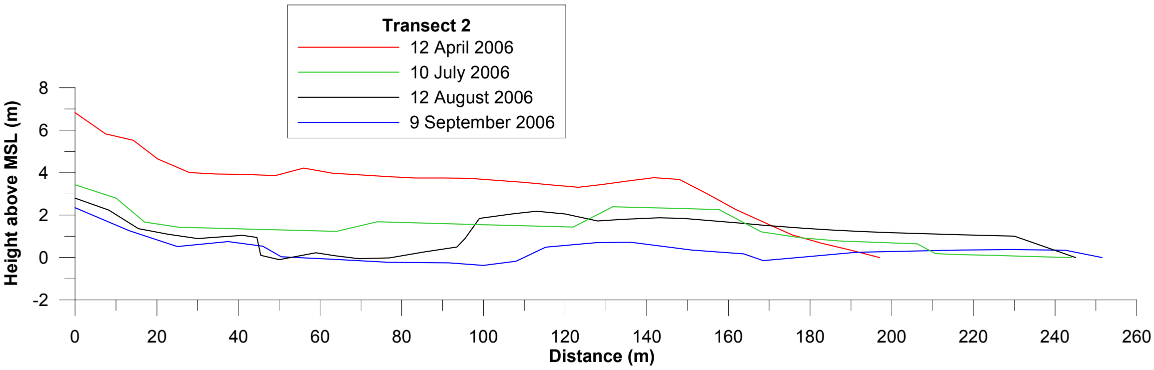

Figure 5. Profile of Transect 2 over time.

Transect 2 showed similar changes to transect 1 with large volumes of sediment removed between April and July 2006. The West Kleinemonde Estuary breached in August, but then closed quickly (the West Kleinemonde channel is situated between 50 and 90 m on the x-axis). Note that the berm is still elevated in that area, i.e. West Kleinemonde was still perched and the breach had not been sufficient to scour out all the sediment in the mouth. Following further heavy rain the West Kleinemonde Estuary breached completely in September 2006 and large volumes of sediment were removed from the mouth area.

Transect 3

Transect 3 showed fewer changes over time because of its position on the berm to the west of the West Kleinemonde Estuary.

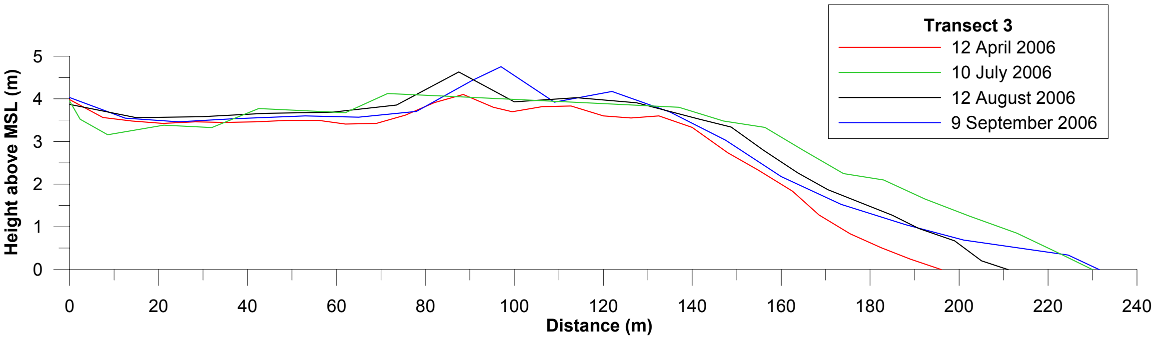

Figure 6. Changes in the profile of Transect 3 between April and September 2006.

Transect 4

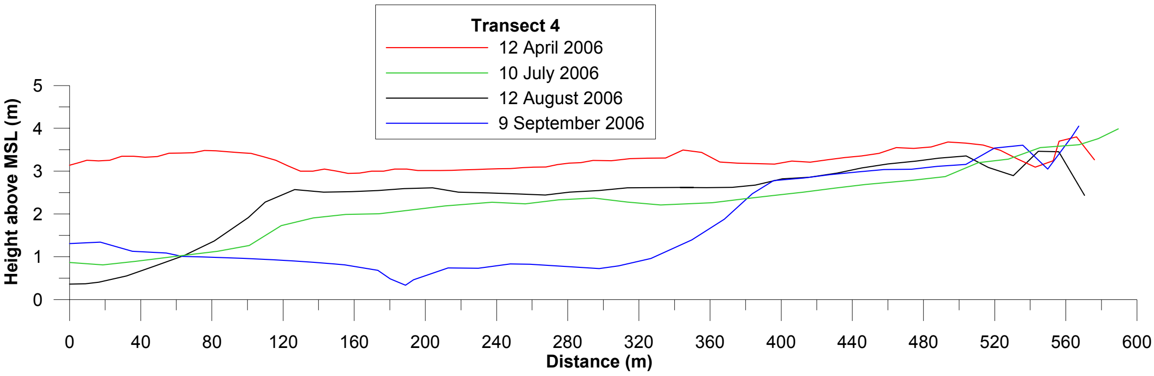

Figure 7. Changes in the profile of Transect 4 between April and September 2006.

Large changes occurred across the berm after April 2006. By July, the East Kleinemonde Estuary had breached briefly and, at the time of sampling, was in a semi-closed perched state and the only water exchange was via marine overwash. In August 2006 both estuaries breached, but the West Kleinemonde Estuary closed quickly again and remained perched, with only surface water draining and with some marine ingress. The West Kleinemonde Estuary breached completely between August and September 2006 and a larger ebb flow discharge was recorded in the West compared to the East Kleinemonde Estuary on 9 September 2006, i.e. 0.6 $m^3 s^{-1}$ in the East and 2.4 $m^3 s^{-1}$ in the West Kleinemonde estuaries respectively.

COASTAL HYDRODYNAMICS

Mean wave height at the outer breaker zone off this estuary typically ranges between about 1 m and 3 m (mostly 1.5-2.5 m), but during rare extreme conditions this can probably range between 0.5 m and 5.5 m. Observed estimates (B. Mackenzie pers. comm.) give a range of 0.5 m to 4 m, with an average of about 2 m.

The degree and type of wave breaking determines the rate of energy dissipation and is also an important parameter in other near-shore processes. Spilling waves gradually dissipate energy while plunging waves dissipate most of their energy at the breakpoint. Collapsing and surging waves dissipate energy close to the shoreline. The wave breaker type at the East Kleinemonde Estuary mouth is usually spilling (rolling down gradually from the top), but sometimes becomes plunging where the top of the wave crashes down onto the foot in one go, especially during offshore or obliquely offshore wind conditions. Mackenzie's (pers. comm.) observations confirm that the wave breaker type is mostly spilling.

The surf-zone width influences the amount of energy dissipation in the surf-zone. A wide surf-zone with many re-breaks means that energy is dissipated over a wider area further away from the shore. Based on an analysis of 13 sets of aerial photographs dated from 1938 to 2004, surf-zone widths (total distance from outer breakpoint to waterline) opposite the mouth area typically range from about 160 m (low waves) to 300 m (high waves). Estimates based on about 69 in-situ visual observations conducted by Mackenzie (pers. comm.) from April to November 2006, give a range of 40 m to 250 m, with an average of about 120 m. However, during extreme conditions the width probably moves outside this range: exceptionally calm - width reduces to approximately 35 m, major sea storm - width greater than 300 m.

The number of re-breaks (i.e. the wave stops breaking, reforms and breaks again) within the total surf-zone is usually one, ranging from none to two as indicated by both photographs and in-situ observations. A shore-parallel bar and trough system is often observed, but weak rip currents are only occasionally seen in or adjacent to the mouth area.

Beaches are globally categorised according to one of six types, viz. dissipative beaches, four classes of intermediate beaches and reflective beaches (e.g. Battjes 1974). Reflective beaches have a steep beach face with surging breakers that are reflected back towards deeper water. Dissipative beaches have flat beach slopes and the wave energy is dissipated gradually in deeper water further away from the beach. Based on the nature of the surf-zone, breaker type and beach slope, conditions at the East Kleinemonde Estuary mouth are usually dissipative to intermediate, but seldom reflective although the beach profiles for April did indicate a reflective state.

Black Rock Point is located about 1.5 km southwest of the East Kleinemonde mouth (Figure 8). Rocky ledges located around the low-tide level, stretch from there to a point eastwards of the mouth area. In the vicinity of the mouth, only a few small rocky areas are visible above the sand. The open bay east of Black Rock Point, within which the mouth is located, has similarities with so called half heart or “log-spiral” bays, but its configuration is also affected by the submerged reefs and Great Fish Point to the east. Due to Black Rock Point, the mouth area is sheltered from inshore waves approaching from the west, with reduced sheltering as the incident wave direction changes to south-southwest. The mouth area is fully exposed to wave direction from south-southwest to about easterly. The east-northeasterly to northeasterly sector is protected by Great Fish Point.

Figure 8. Satellite photograph of Kleinemonde Mouth coastal area.

SEDIMENT DYNAMICS

The aeolian-headland-bypass system at Black Rock Point together with surf- and swash-zone sediment transport, provide an ample supply of sand to the mouth area from the west. The wide sand beach and dunes to the east (Figure 8) also supply large amounts of sand to the mouth area from the east during south-easterly to easterly wave conditions.

Sediment transport in the nearshore region is usually categorized as longshore (parallel to the shoreline) or cross-shore (perpendicular to the shoreline). Based on the model of Kamphuis (Kamphuis 1991, Schoonees & Theron 1996) and average wave height conditions of 2 m, a wave period of 12 s, a breaker wave angle of 1 - 2°, a median sand grain diameter of 0.25 mm and a beach slope of 1 in 21, an average potential longshore transport of 700 000 m^3 to 1 900 000 m^3 per annum is predicted. The CERC model (CERC 1984) predicts about half of these values. Wave directions near the outer breakpoint off the mouth are mostly close to shore-normal. However, waves breaking with small angles from the west (i.e. up-coast longshore current generated) occur more frequently than waves breaking from the east (i.e. down-coast longshore current). The actual net longshore sediment transport, which is usually up-coast is also dependent on the amount of sediment available to be transported and is probably much less than the potential transport rate. Typical shorter-term net cross-shore rates during a storm would be in the order of a few m^3 m^{-1} $hour^{-1}$ for 24 hours (most southern African sea storms have durations of a few hours to a few days).

The total net rate of possible sedimentation, and even the instantaneous sediment transport rate in the estuary, is orders of magnitude less than the estimated longshore or cross-shore transport rates, which move coastal sediments into the estuary mouth area. Even if the longshore transport rate is sometimes nil, large amounts of marine sediment is usually stirred up by wave action. A more than ample supply of marine sediment is therefore usually present at the estuary mouth, for potential transport into the estuary and mouth closure. Thus the amount of marine sediment intrusion into the estuary is mainly dependent on the net transport capacity of the ebb and flood tidal flows near the mouth, and most probably not on the amount of sediment available outside the mouth. Even the relatively smaller amounts of direct marine sediment intrusion due to wash over of the berm or aeolian transport could be of the same order of magnitude as the possible total rate of net long-term sedimentation in the estuary. It is necessary to note that it has not been proven or factually established that the East Kleinemonde Estuary is undergoing net long-term sedimentation. The literature shows that the sediment balance in estuaries often relies on a subtle balance between dominant flood and ebb tide flows, as well as on catchment sediment yield and river flood regime (Theron *et al*. 2002).

The type of wave breaking, the extent of near-shore currents and the strength of the backwash are all directly related to the slope of the tidal face of the beach. An increase in this slope will lead to more severe surf conditions, with more sediment entrainment and availability for transport into the estuary mouth. The wave and surf-zone characteristics described above confirm that medium to relatively high wave energy conditions occur relatively close to the estuary mouth, largely during high tide, when the then narrower surf-zone includes the steeper part of the beach profile and the water depth increases over the outer sandbar allowing more wave energy to pass. Sediment loads are entrained by the turbulent wave action and carried into the estuary mouth area where the sediment is deposited in this lower energy environment, which consequently has a lower sediment carrying capacity. When this deposition rate exceeds the erosion potential of ebb-tidal and river outflow, a net sediment build-up occurs. If this situation continues for long enough, the mouth is closed.

Thus, besides river inflow and tidal flow, wave conditions, together with marine sediment availability, also play a major role in estuary mouth dynamics and state. Wave data recorded off East London which is approximately 100 km northeast show, for example, that during the period March 2005 to February 2006 both the median and extreme wave heights (1% exceedance) were only slightly larger than the long-term (1992-2005) values for East London, due mainly to more stormy conditions during the spring and summer seasons. More specifically, it can be shown that of the 13 mouth closure events identified in the hydrodynamics section (Appendix C), at least 4 can be directly correlated with sea storms (high wave events). Based on wave recordings off East London these are the events of 9 November 2005, 26 June 2006, 31 August and 20 October 2006.

AEOLIAN TRANSPORT

The main wind directions are approximately parallel to the coastline, with significantly more wind from the west-southwest sector than from the east-northeast quadrant (Figure 8). The large dune fields in the vicinity confirm extensive aeolian activity. The crests of the dune ridges are aligned approximately perpendicularly to the shoreline (or south-southeast to north-northwest). However, the local "layout" and topography, result in the major aeolian input being from the east-northeast sector.

Due to the high ridge between the two mouths (Figure 5) and the low East Kleinemonde Estuary area, shore parallel and obliquely onshore winds from the easterly side are channelled around the "corner" of the higher area north of the East Kleinemonde mouth and up the estuary. Thus, much wind-blown sand is deposited into the estuary mouth area from the large dunefield on the easterly side. From historical aerial photographs and *in situ* observations it seems that the mouth area is always filled in from the east-southeast due to aeolian and perhaps also wave driven sediment transport.

BERM DIMENSIONS

Besides also being related to wave conditions, sediment characteristics and sometimes also aeolian transport, berm formation and berm dimensions obviously have a direct impact on mouth closure and breaching, over-wash into estuaries and seepage (both to and from the sea).

Historical aerial photographs show that the berm width is typically of the order of 200 m. Estimates based on the 2006 observations conducted by B. Mackenzie give an average width of 70 m, with a maximum of about 120 m.

Berm height could potentially range from as low as +0.5 m MSL up to a maximum of +5 m MSL, excluding dunes superimposed on top of the berm. Based on the models of Nielsen and Hanslow (1991), average wave conditions of 2 m, wave period of 12 s and beach slope of 1 in 21, berm crest elevations of 1.2 m to 2.9 m MSL are predicted depending on state of the tide. From estuary water level recordings (see hydrodynamics section), it can be derived that during March 2005 to November 2006, berm crest elevations ranged from about +0.95 m to +2.3 m MSL. Bearing in mind the significant effect of duration, the theoretical predictions and derived elevations are considered to correspond well. The observed estimates (Mackenzie, pers. comm.) of berm crest height above water level in the estuary, give an average of 35 cm to a maximum of 106 cm.

The surveyed beach transects and transect along the crest of the berm indicate that on a number of occasions, berm crest elevations typically ranged between about +2 to +3.5 m MSL when the mouth was closed.

MOUTH AND ESTUARY CHANNEL CONFIGURATION

The position of the water line in both estuaries as well as the spring high and low tide line was mapped using a GPS with ArcPad software on 28 January, 12 April, 10 July, 12 August and 9 September 2006. The positions were compared from one date to the next in GIS maps below.

January vs April 2006

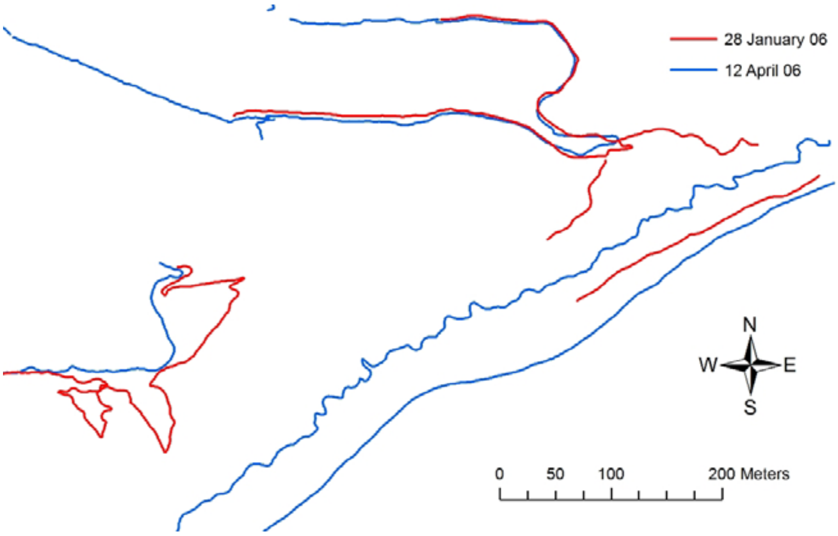

Figure 9. Position of the mouths and waterlines in January and April 2006.

April vs July 2006

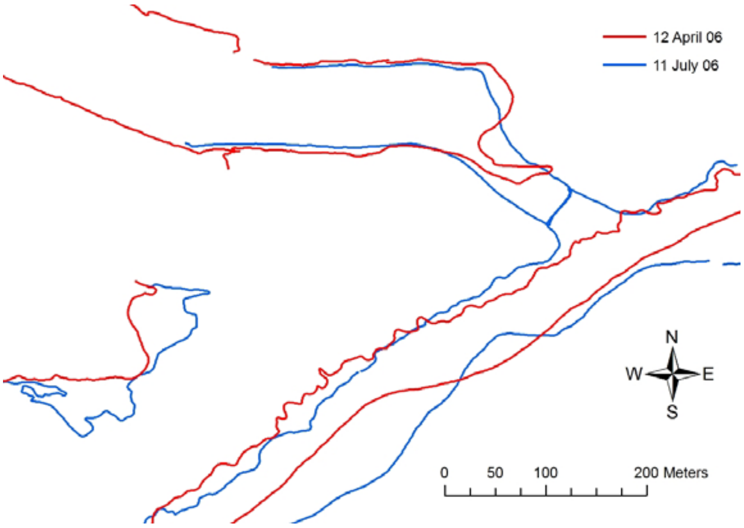

Figure 10. Position of the mouths and waterlines in April and July 2006.

Between April and July the East Kleinemonde Estuary breached briefly and by 11 July was in a semi-closed state with only marine overwash entering the estuary during spring high tide.

July vs August 2006

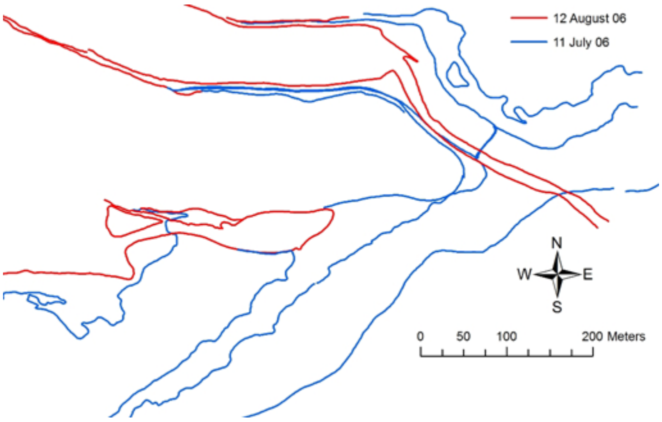

Figure 11. Position of the mouths and waterlines in August and July 2006.

Early in August both estuaries breached. The West Kleinemonde closed soon after breaching and the berm remained elevated in front of the West Kleinemonde (See Figure 3).

August vs September 2006

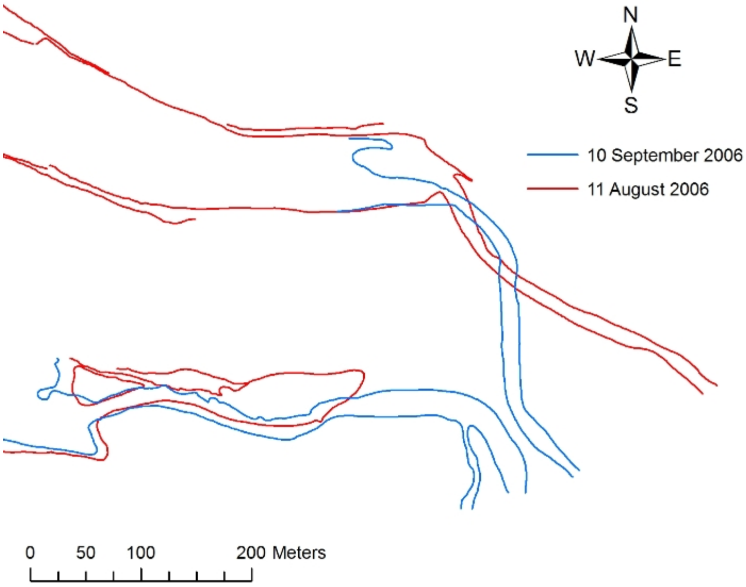

Figure 12. Position of the mouths and waterlines in August and September 2006.

The mouth of the East Kleinemonde Estuary remained open for the whole period but migrated towards the west by more than 100 m. The West Kleinemonde Estuary breached completely and was flowing out very strongly.

Figure 13 shows the difference in beach width between the September dissipative conditions and April reflective conditions. The grain size of the reflective beach in April was significantly coarser than the fine sand typical of dissipative beaches.

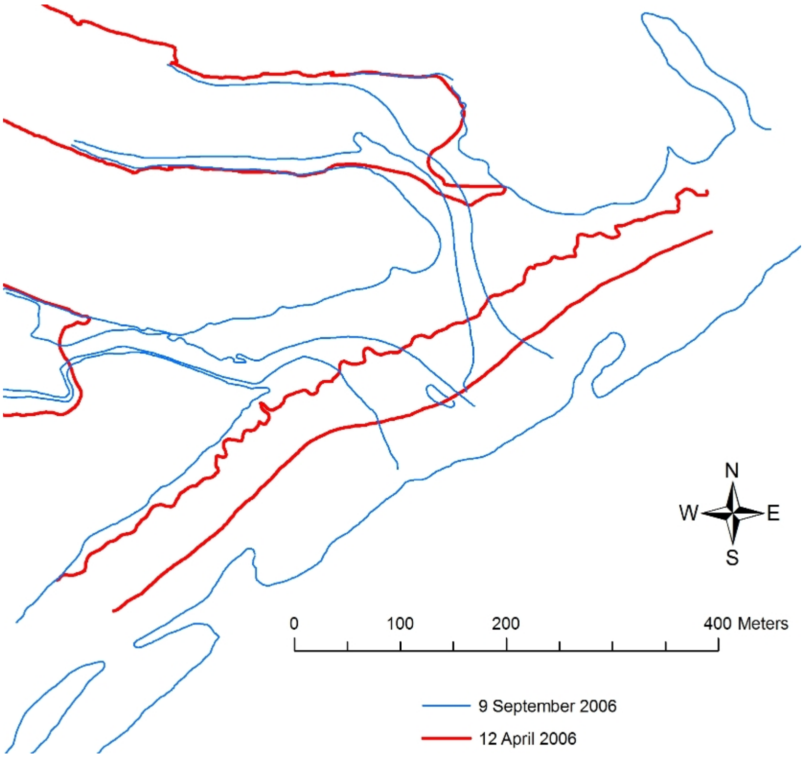

Figure 13. Position of the mouths and waterlines in September and April 2006.

The surveys and transects along the crest of the berm provide some data on the mouth channel dimensions i.e. the width and the depth of the outflow channel, during open mouth conditions. Brief descriptions of the channel configuration in conjunction with the fixed-point photography provide further information. Estimates based on about 69 *in situ* visual observations conducted by B. Mackenzie (pers. comm.) from April to November 2006, give an average channel width of only 8 m, with a maximum of about 20 m. The comparative narrowness of the mouth channel is related to the relatively small breaching volume of the estuary, small catchment, lack of substantial wave shelter and the abundant supply of marine sediment.

Historical aerial photographs (1938-2004) indicate that the main direct anthropogenic impact on the estuarine morphology upstream of the mouth has resulted from the road-bridge and embankments constructed in the early 1960s. The bridge opening permanently fixed the deep channel against the west bank. However, it seems that in this area, the channel was always deeper towards the western side because it was on the outside of the bend. After construction of the bridge, the eastern sand bank downstream of the bridge has widened compared to the pre-bridge configuration. Despite this, the bridge does not appear to have had a major impact on the physical dynamics of the estuary.

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Appendix E

Specialist Report: Water Quality

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INTRODUCTION

Specific hypotheses were proposed for the water quality component of this study. They were as follows:

1. Closure of TOCEs results in a marked change in the characteristics and/or distribution patterns of *physico-chemical variables* (e.g. salinity, turbidity and DO) compared with the open state. Within 1 month after mouth closure the water column becomes almost homogenous with respect to *salinity, turbidity and dissolved oxygen*.
2. Because small TOCEs, such as the East Kleinemonde, are usually shallow (<2 m), low *dissolved oxygen* is not a major concern because of wind mixing. However, low oxygen can become a problem if there is a high organic loading, for example from the die-back of benthic macroalgae.
3. Where the river is a major source of *inorganic nutrients*, as is the case for the East Kleinemonde, reduction in freshwater inflow results in nutrient depleted conditions in the water column because there is no significant *in situ* regeneration of inorganic nutrients into the water column of small TOCEs.

Taking into account these specific hypotheses selected for the water quality component, a generic conceptual model on the water quality characteristics of small TOCEs was proposed by Snow & Taljaard (2006). The conceptual model, and this study, primarily focused on physico-chemical variables (i.e. temperature, salinity, pH, turbidity and dissolved oxygen) and inorganic macronutrients (i.e. dissolved nitrate, nitrite, total ammonium, inorganic phosphate, reactive silicate and total phosphorus). Although organic nutrients, e.g. particular organic carbon and nitrogen, either entering from the catchments or generated within estuaries, can also be an important food source to estuarine biota, it was not possible to include these within the current research budget. However, they require further investigation, not only in terms of organic inputs from the catchment, but also with regard to the exchange of organic nutrients across the estuarine-marine boundary.

The conceptual model identified three dominant states in which small TOCEs can exist, namely:

1. Mouth open: when the mouth is open to the sea, allowing seawater intrusion during high tide with river inflow introducing freshwater into the upper reaches. A horizontal (and possibly vertical) salinity gradient exists in the estuary.
2. Mouth semi-closed: where the berm height prevents continuous seawater intrusion during high tide (i.e. seawater intrusion is usually limited to spring high tide). However, the berm is not high enough to prevent water draining from the estuary into the sea. During this state, salinity in the water column is near homogenous, although some vertical and longitudinal stratification may be evident immediately after closure. Depending on the height of the berm and conditions at sea, sporadic seawater overwash into the estuary can occur.
3. Mouth closed: where the height of the berm prevents seawater from entering the estuary as well as water draining from the estuary into the sea. Low volumes of river water might still be entering the estuary and sporadic overwash of seawater can occur depending on conditions at sea and berm height. At the onset of this state, vertical stratification usually develops as a result of low density freshwater flowing across higher density saline water. Through entrainment of freshwater into the more saline bottom layer, as well as wind mixing forces, the estuary gradually changes into a homogenous brackish water body (i.e. salinity between 10 to 25 psu).

Depending on factors such as estuary size, beach profiles and mouth protection, all three of these states can occur (3-phased systems) or only the open and closed states (2-phased systems). Although the East Kleinemonde Estuary is classified as a 3-phased system, it was only possible to sample the estuary during an open and closed mouth state during this study in 2006.

A simple conceptual model of the water quality (or biogeochemical) structure and characteristics associated with each of these states are described in a review prepared for this study and the reader is referred to that report for further details (Snow & Taljaard 2006). The results presented here will be tested against the hypotheses proposed for the water quality component (see above), as well as the proposed water quality conceptual model provided in the review report.

MEASUREMENT PROGRAMME

Sampling Programme

In order to test the hypotheses of the water quality conceptual model, the following measurements were taken in the East Kleinemonde Estuary:

- *Water column measurements* to investigate the influence of river inflow and tidal exchange on water quality in the estuary sampled during closed (State 3) and open mouth (State 1) conditions
- *Microcosm measurements* to investigate nutrient exchange across the sediment-water interface

Water column (influence of river inflow and tidal exchange)

Water column samples were collected from 10 stations along the length of the estuary, as well as from the sea, on the following occasions (Figure 1):

- 18 March 2006 (closed mouth state)
- 8/9 September 2006 (open mouth state).

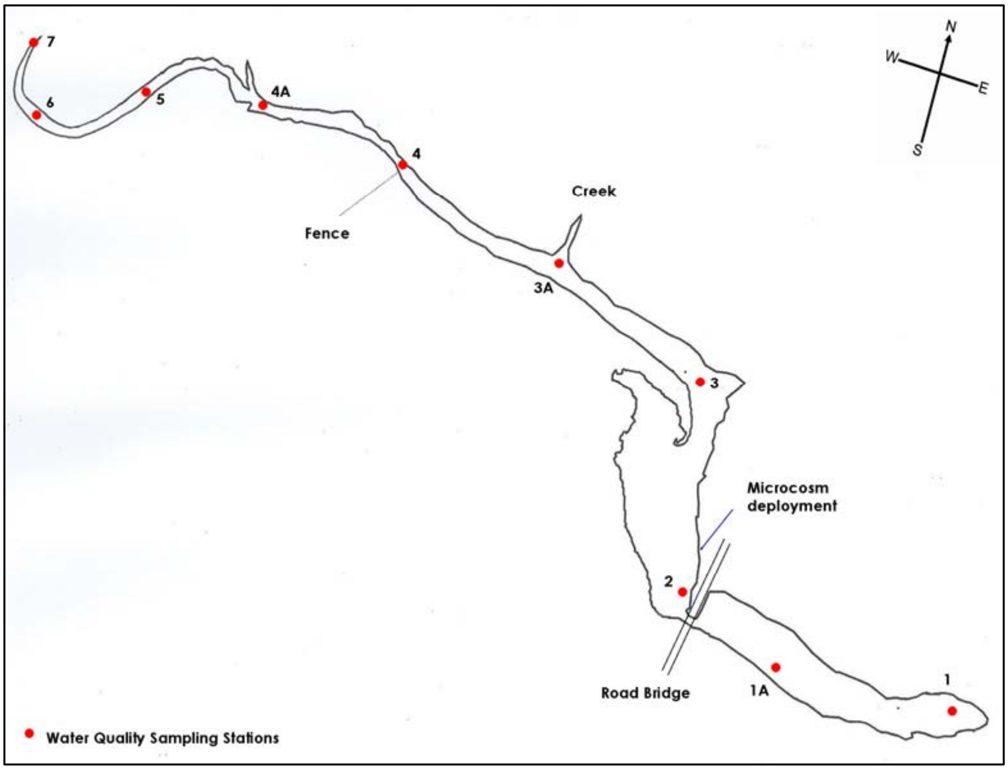

Figure 1. Map showing water quality sampling stations in the East Kleinemonde Estuary.

Depth profiles of temperature, salinity, pH and dissolved oxygen were collected at each station, while samples for dissolved nitrate, nitrite, total ammonium, inorganic phosphate, total phosphorus and reactive silicate analysis were collected from the surface and bottom waters. Bottom water samples were collected using a Student's horizontal sampler.

Additional datasets of salinity, temperature, dissolved oxygen and turbidity were collected for the period March 2006 to November 2006 (Mackenzie pers. comm.). These data were also used in the evaluation of the water quality characteristics of the system.

Microcosms (sediment-water exchange)

A microcosm study was undertaken on 8-9 September 2006 using a customised experimental design used by the CSIR (Stellenbosch) on similar studies conducted in Maputo Bay (Figure 2). The microcosm chambers were deployed in a muddy subtidal area (Figure 1). Muddy areas are considered to be depositional areas, where organic material used in the re-mineralisation process accumulates and where a stable habitat is created for the benthic microalgae responsible for photosynthetic processes at the sediment-water interface.

Figure 2. Microcosm apparatus used during September 2006, showing the light chamber, dark chamber and the closed circuit syringe system that was used to take samples.

Two microcosms were deployed during this exercise at a subtidal location just upstream of the road bridge (Figure 1) - one light chamber, included photosynthetic processes and one dark chamber excluded photosynthetic processes. The light chamber had an inner bottom diameter of 0.385 m and a volume of 12 ℓ, while the dark chamber had an inner bottom diameter of 0.390 m and a volume of 10 ℓ. A container of water was collected at the beginning of the experiment to act as a control sample. Water from this container was used to replace water in the chambers during sampling. Samples from the chambers were collected through a closed circuit syringe system, where a known volume of water (50 ml) was be extracted from the chambers and a similar volume of water flowed back into the chambers (Figure 2). Unfortunately the systems could not be fitted with stirrers. However, to simulate stirring, a 20 ml sample of control water was flushed back and forth twice through the syringe system prior to each sampling. Dye tests showed that this technique was effective in mixing water within the chamber. Temperature, salinity, pH and dissolved oxygen were logged at 1 minute intervals in the light chamber, using an YSI meter, while 60 ml samples for macronutrient analyses were collected hourly from both chambers, being replaced with water from the control sample that was also sampled hourly.

Analytical Equipment and Procedures

Physico-chemical data were measured *in situ*, using an YSI Meter – 6600 for water column measurements and a YSI meter - 556 MPS for the microcosm study. Macro-nutrient samples for both the water column and microcosm studies were filtered in the field through 0.45 µm Millipore filters, with the exception of those used for total phosphorus. Samples were stored in polyethylene sampling bottles and frozen until further analysis.

Water column and microcosm samples collected on 18 March 2006 and 8/9 September 2006 were analysed by the CSIR's accredited laboratory in Stellenbosch using an Auto-analyser and the methods as described in CSIR (1997).

The rate of exchange across the sediment-water interface (assumed to be the dominant factor responsible for variation in the concentration of macro-nutrient in the chambers) was calculated as follows:

$Rate\ (mg\ m^{-2}hr^{-1}) = \{C_t{*}V – [(C_{t-1}{*}V-0.06 + C_{c(t-1)}{*}0.06)/1000]\}/A$

Where

C_t = concentration measured in chamber at time t (in µg ℓ^{-1})
C_{t-1} = concentration measured in chamber at time t-1 (in µg ℓ^{-1})
$C_{c(t-1)}$ = concentration of control sample at time t-1 (in µg ℓ^{-1})
0.06 = volume of sample taken from chamber (and replaced by control sample water) (in ℓ)
V = volume of chamber (in ℓ)
A = bottom area of chamber (in m^2)

RESULTS AND DISCUSSION

Water column

Salinity

Salinity profiles measured in the East Kleinemonde Estuary during the closed state (18 March 2006) and open state (8-9 September 2006) are presented in Figures 3a and b. On 18 March 2006, when the mouth had been closed for an extended period, the estuary was brackish (23-25 psu) showing only a slight horizontal gradient (Figure 3a). However, some rain in the catchment just prior to sampling resulted in a very thin layer (< 0.5 m) of fresher water in the upper reaches, overlying the more brackish bottom water.

Salinity distribution measured on 8 and 9 September 2006, when the mouth had been open to the sea for some time (Figure 3b), were distinctly different to those observed during the closed state. Not only were there strong horizontal and vertical salinity gradients present, but there was also a marked difference in salinity distribution between low and high tide. The degree to which salinity distribution would differ between high and low tide is, however, dependent on the extent of tidal intrusion that is determined by the depth of the mouth and volume of river inflow.

These observations fit the conceptual model proposed for TOCEs, where in the closed state these estuaries generally display a homogenous brackish water column, but develop strong longitudinal and sometimes vertical salinity gradients, during the open state (Snow & Taljaard 2006). They also support the hypothesis that closure of TOCEs results in a marked change in the distribution pattern of physico-chemical variables when compared to the open state.

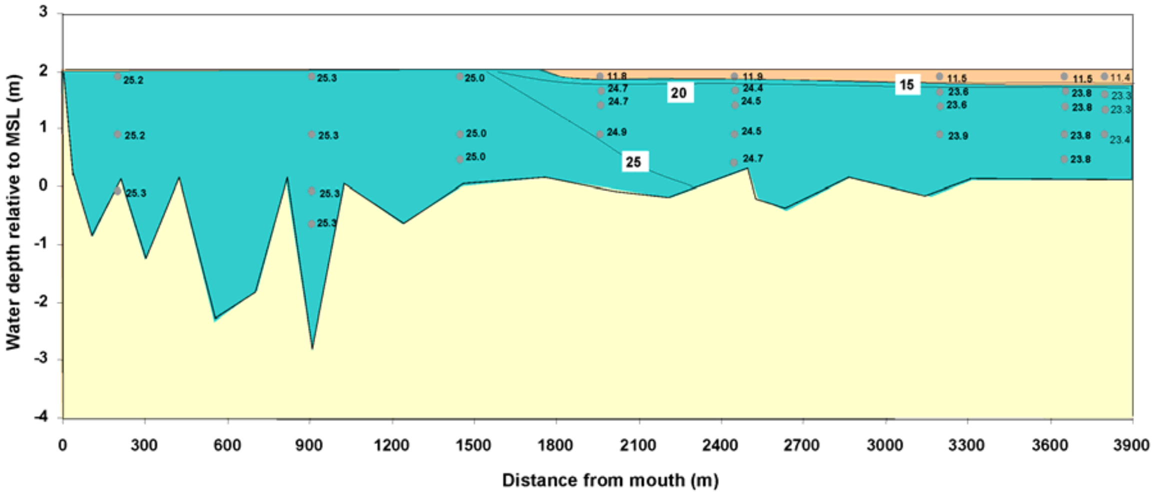

Figure 3a. Salinity profile (psu) measured in the East Kleinemonde Estuary on 18 March 2006 (closed mouth).

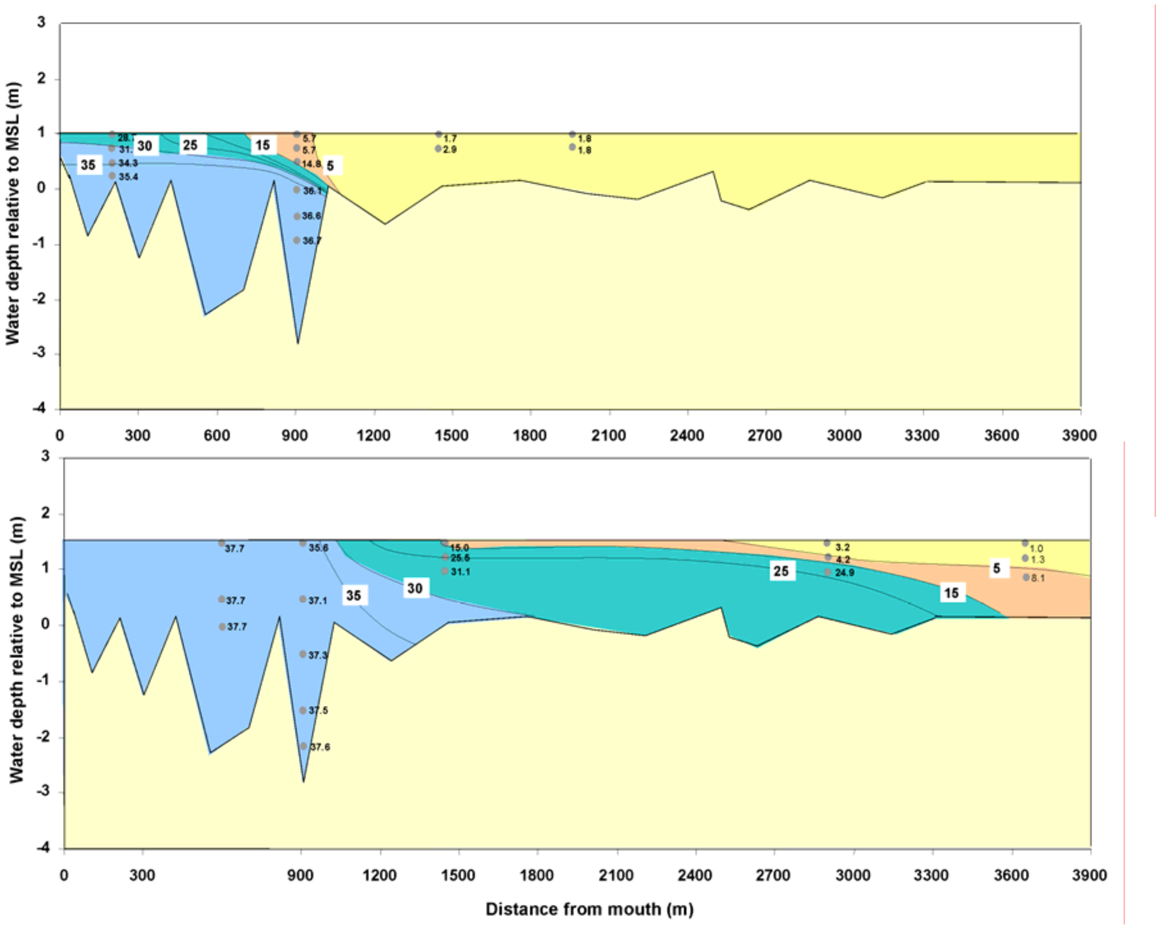

Figure 3b. Salinity profile (psu) measured in the East Kleinemonde Estuary on 8 September 2006 (low tide – upper profile) and 9 September 2006 (high tide – lower profile).

Temperature

Temperatures measurements taken in the East Kleinemonde Estuary, plotted against salinity data, are provided in Figure 4.

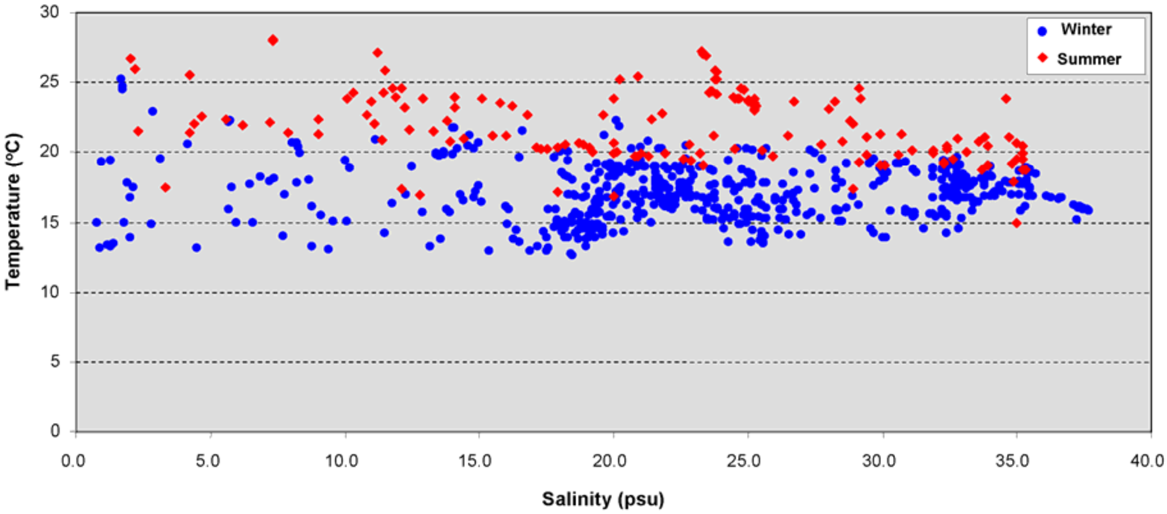

Figure 4. Temperature versus salinity measured in the East Kleinemonde Estuary during summer and winter (March to November 2006).

Figure 4 shows that there is no distinct seasonal pattern in salinity but that temperature had a distinct seasonal pattern over the period March 2006 to November 2006. Results show that summer temperatures are generally higher (20-25°C) compared with winter temperatures (13-20°C). These results also agree with the conceptual model, which proposes that water temperature variations in TOCEs are usually a function of seasonal trends in atmospheric temperature (Snow & Taljaard 2006).

pH

pH values in the East Kleinemonde Estuary did not show any marked variation at different salinity values, along the estuary or between surveys, ranging between 7.7 and 8.3 (Figure 5), as proposed in the conceptual model, where pH levels within TOCEs were expected to range between 7 and 8.5 (Snow & Taljaard 2006).

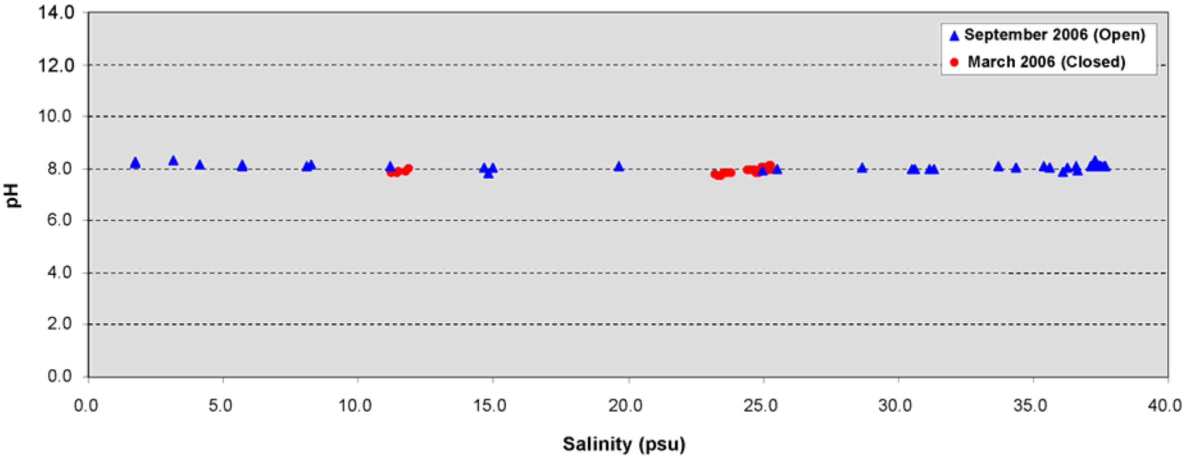

Figure 5. pH versus salinity measured in the East Kleinemonde Estuary during summer and winter (March to November 2006).

Dissolved Oxygen

Figure 6 shows the variation in dissolved oxygen (DO) concentration as a function of water salinity during open and closed states for the period March to November 2006. When the mouth was closed a distinction is made between surface (<1 m) and bottom (>1 m) water measurements. These results clearly show that during the open state when salinity was low as a result of fresh water inflow, the estuary was well oxygenated, with DO levels never dropping below 4 mg ℓ^{-1}. During the closed state there was greater variation in DO concentration. Bottom water was well oxygenated with DO concentrations only dropping below 2 mg ℓ^{-1} on a few occasions. These few occasions were in the mid-salinity ranges when the mouth had been closed for some time.

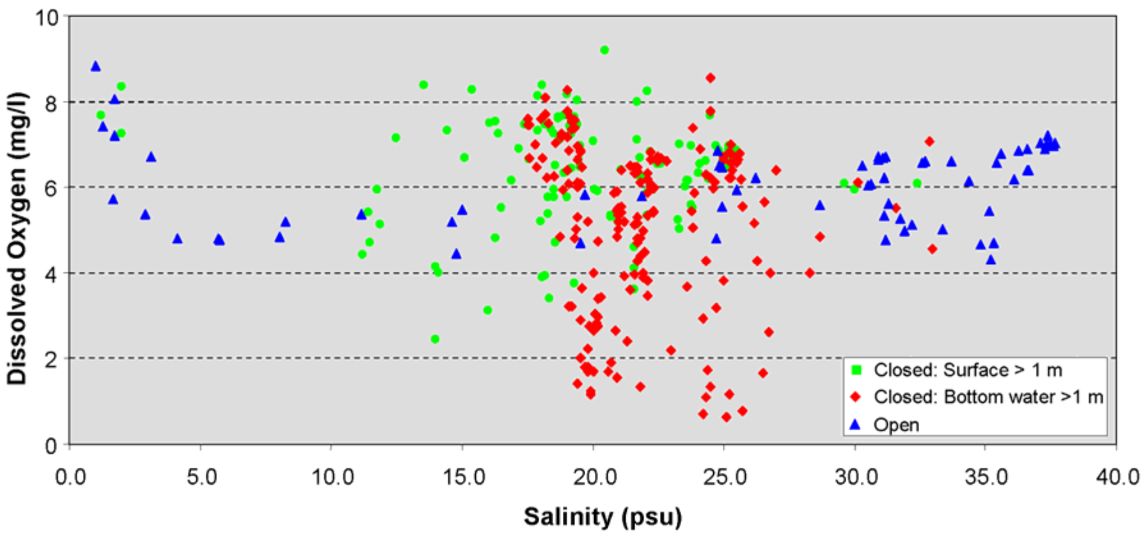

Figure 6. Dissolved oxygen versus salinity in the East Kleinemonde Estuary from March to November 2006, under closed (surface and bottom water) and open mouth states.

These observations fit the conceptual model and support the hypothesis that DO levels in the bottom water, at times, can become hypoxic during a closed state. This is somewhat unexpected, particularly for a shallow system such as the East Kleinemonde. However, deep quiescent zones (e.g. Station 2) did have low DO levels at a depth >1 m owing to strong density differences between bottom and top layers of the water column, which prevented mixing. Similar hypoxic to anoxic conditions have been observed in shallow systems like the Maitland Estuary (Gama *et al*. 2005). A possible explanation could be that these low oxygen events are caused by prolonged periods of very calm weather, when there is no wind mixing that generally prevents such conditions from developing in shallow estuaries. Organic loading (detritus) could also be a possible cause (Nielsen *et al*. 2004) but these arguments need to be confirmed. It also supports the hypothesis that closure of TOCEs results in a marked change in distribution pattern of physico-chemical variables compared with the open state.

Turbidity

Figure 7 illustrates turbidity distribution patterns measured in the East Kleinemonde Estuary, in relation to salinity, for open and closed states during the period March to November 2006. There does not seem to be a clear distinction between turbidity for open or closed states, except that turbidity levels >100 NTU tend to occur during the open state. This is attributed to a stronger freshwater influence during the open state because river water generally has higher turbidity, but also because of the stronger turbulence during strong tidal exchange that may also re-suspend sediments.

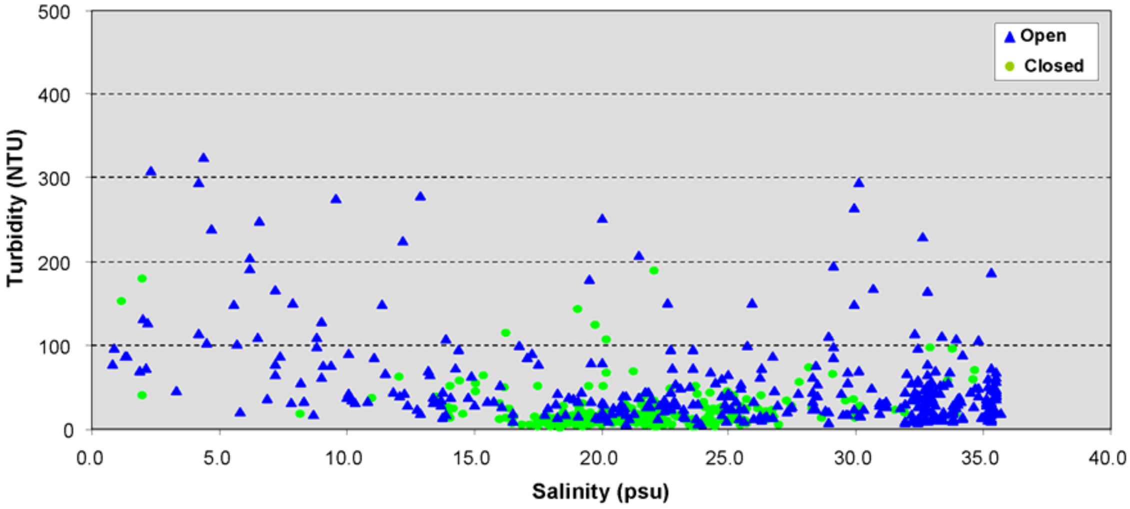

Figure 7. Turbidity versus salinity measured in the East Kleinemonde Estuary from March to November 2006, under closed and open mouth states. The salinity axis illustrates the influence of seawater intrusion.

The assumption that the system is probably more turbid during open states compared to closed states is also supported by the Secchi disc readings taken during March 2006 (closed) and September 2006 (open), where readings were lower during the open state than when the system was closed (Table 1).

Table 1. Secchi disc depths in the East Kleinemonde Estuary (March - September 2006).

Station	**Secchi disc depth (m)**	
	March 2006 (closed)	**September 2006 (open, with strong river influence)**
1	> 1.75	0.65
2	1.45	0.18
3	1.22	0.1
4	-	-
5	1.1	-

In contrast to other physico-chemical parameters these results indicate that there may not always be a marked difference between turbidity characteristics during open and closed states as hypothesised, although on occasions turbidity levels increased markedly during the open state.

Inorganic nutrients

Figures 8a-c provide property-salinity plots for dissolved inorganic nitrogen (DIN), total phosphorus (TP), dissolved inorganic phosphate (DIP) and dissolved reactive silicate (DRS) measured in the East Kleinemonde Estuary in March 2006 (closed state) and September 2006 (open state). The concept of assessing inorganic nutrient versus salinity by comparison with a theoretical line representing conservative mixing of riverwater and seawater has been used and discussed previously by Head (1985), Balls (1994) and others. The use of a mixing diagram shows nutrient concentrations plotted against salinity along the estuarine gradient. Mixing diagrams provide a convenient method for displaying the net effect of nutrient processes within estuaries, whereby deviation from the conservative mixing line is used to interpret results. For example, downward curvature in the mixing diagram implies nutrient uptake, while upward curvature implies nutrient release.

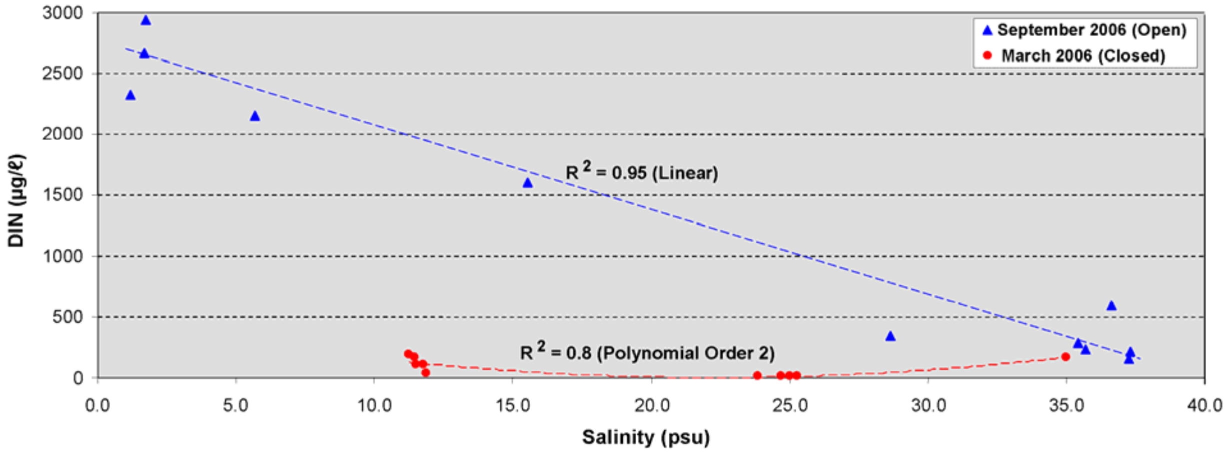

Figure 8a. Dissolved inorganic nitrogen (DIN) versus salinity measured in the East Kleinemonde Estuary during March 2006 (mouth closed) and September 2006 (mouth open).

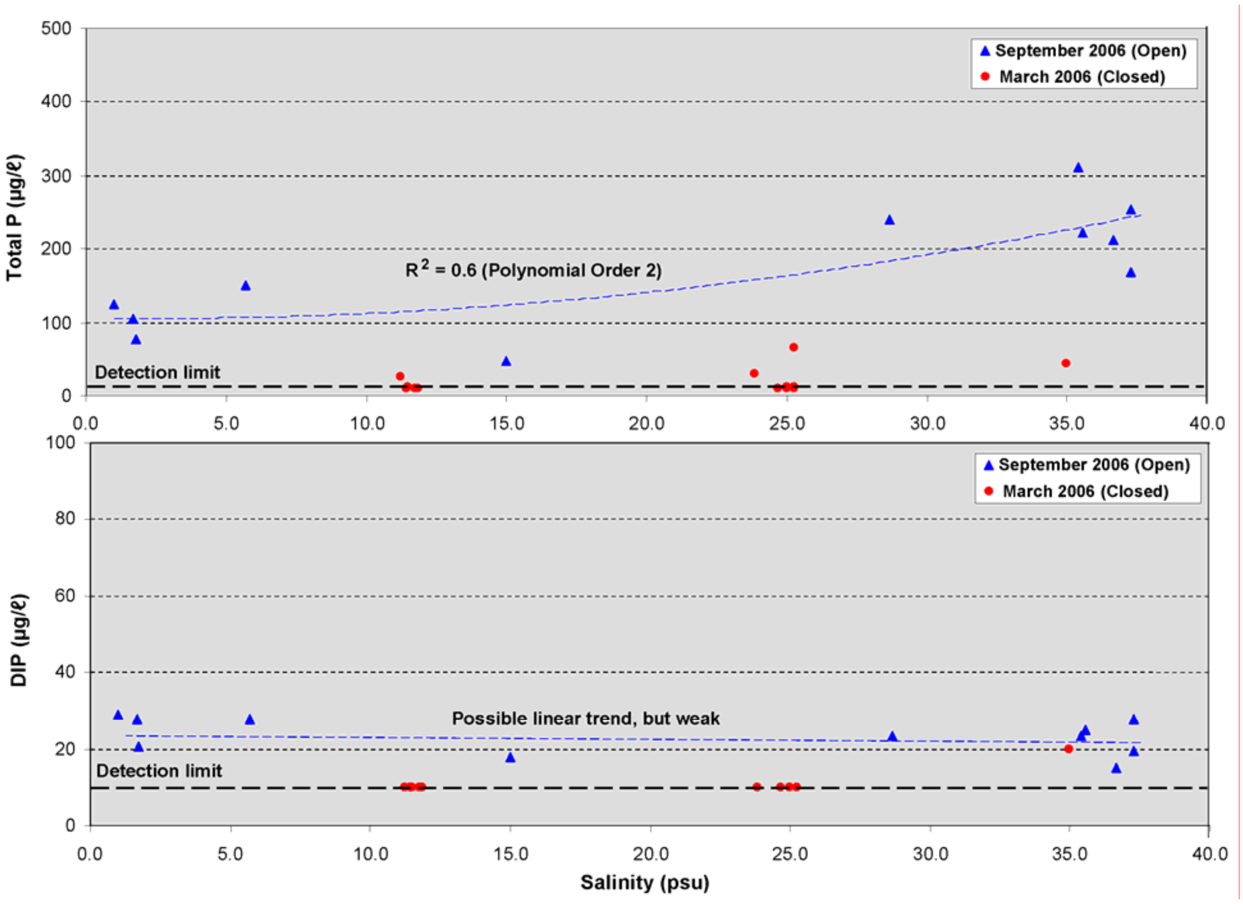

Figure 8b. Total phosphorus (TP) and dissolved inorganic phosphate (DIP) versus salinity measured in the East Kleinemonde Estuary during March 2006 (mouth closed) and September 2006 (mouth open).

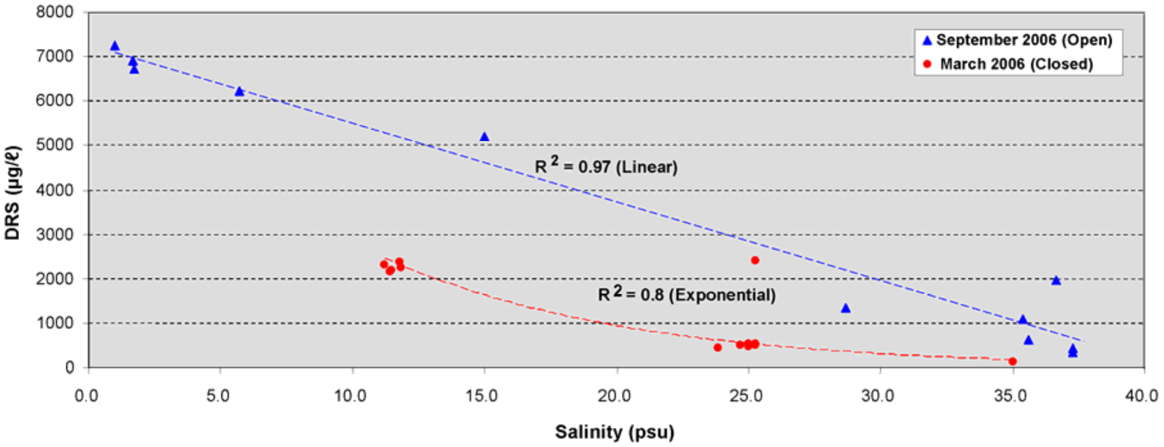

Figure 8c. Dissolved reactive silicate (DRS) versus salinity measured in the East Kleinemonde Estuary during March 2006 (mouth closed) and September 2006 (mouth open).

In the East Kleinemonde Estuary, results showed that during March 2006, after an extended period of mouth closure, the system was depleted of DIN, TP and DIP (Figure 8a and b). Although DRS were still present in significant concentrations, it was still much lower compared to the open state (Figure 8c).

During the open state (September 2006), DIN concentrations (mainly comprising nitrate-N) entering the estuary via the river were exceptionally high (>2500 µg $ℓ^{-1}$), probably due to agricultural enrichment in the catchment. DIN concentrations in the estuary showed a very strong linear correlation (r^2 = 0.95) with salinity, suggesting that concentrations in the water column were largely a function of mixing between river and seawater (Figure 8a). A similar trend was observed for DRS during the open state (Figure 8c).

By subtracting DIP concentrations from TP it appears that during the open state, phosphorus entered the estuary mainly in the particulate form, with the sea being the major source (Figure 8b). Adsorption of phosphorus onto particulate materials can occur either onto cohesive non-biological particles, for example fine sediment particles (<60 µm), or on organic particles, e.g. microalgae, zooplankton and allochthonous debris (Nixon *et al*. 1996). These particulate substances can have their origin from the catchment through river inflow (i.e. as suspended sediment loads and debris), from autochthonous regeneration within the estuary (e.g. re-suspension of sediments and/or localised phytoplankton blooms), or through organic material entering from the sea (e.g. organic debris). In the East Kleinemonde Estuary, the non-linear relationship between salinity and TP (downward curvature) suggests removal of TP from the water column towards the middle reaches, probably due to settling of particulate matter from the water column as a result of reduced turbulence.

Inorganic nutrient characteristics in the East Kleinemonde Estuary largely fit within the conceptual model proposed for TOCEs. During the open state, nutrient concentrations are mainly influenced by concentrations in the inflowing river and seawater, while the distribution of nutrients within the estuary is a function of mixing, i.e. follow the conservative mixing line. Strong water exchange and short flushing times limit the influence of other physical and biochemical processes on water column nutrient distribution patterns. Results from the East Kleinemonde also indicated that after prolonged closure, water column inorganic nutrients largely became depleted. This also agrees with the hypothesis that during prolonged periods of mouth closure, water column nutrients become depleted, probably because there is no significant *in situ* regeneration of inorganic nutrients into the water column.

Microcosm studies (sediment-water column exchange)

Figures 9a and 9b show the results obtained from the microcosm study conducted on 8-9 September 2006 in light and dark chambers, respectively. The experiment commenced around 10:00 am on 8 September 2006 and was terminated around 11:00 am on 9 September 2006. Variation in temperature, salinity, pH and dissolved oxygen, logged at one minute intervals in the light chamber are illustrated in Figure 9a.

The diurnal signal in temperature is clearly visible, with temperature decreasing towards the late afternoon and increasing again the next morning after sunrise. The salinity signal showed that initially (about first five hours into the experiment) salinity remained constant at ~2 psu. However as the tide started pushing in, the salinity started increasing over a period of about 7 hours to about to about 12-15 psu, where it remained up to the end of the experiment. Salinity measured in the interstitial water was much higher (~15 psu) (Table 2) than in the overlying water at the onset of the experiment and than in the water initially trapped in the chambers (~3 psu). The increase in salinity measured in the light chamber was therefore attributed to bentho-pelagic water equilibration, e.g. through processes such as diffusion.

The pH level in the light chamber remained uniform throughout the experiment, ranging between 7 and 7.6.

Dissolved oxygen results showed a distinct pattern where levels started to drop markedly just after sunset and decreased to near zero during the night. However, a rapid increase in DO levels was observed the next morning when concentrations returned to between 5-6 mg ℓ^{-1} within an hour after sunrise (at the sampling site). The oxygen distribution pattern correlated very well with the photosynthesis-respiration cycle. During daylight, oxygen was released into the overlying water through photosynthesis, while during the night respiration removed DO to near anoxic levels just before sunrise followed by a steep increase at sunrise. Benthic biological activity was considered to be largely accountable for this observed trend in DO. For example, chlorophyll *a* concentrations in the surface sediments (upper 1 cm) of the experimental chambers averaged 10.7 µg g^{-1} (Table 2) indicative of benthic microalgae activity.

Inorganic nutrients measured in the chambers varied throughout the experiment. In both the dark and light chambers nitrite-N concentration remained low (<20 µg ℓ^{-1}) with no measurable rate of exchange (Figure 9a and b).

In contrast, nitrate-N showed a marked drop in concentration in both the dark and light chambers suggesting an uptake of nitrate-N from the water column (Figure 9a and b). However, the decrease in nitrate-N showed a strong inverse linear correlation to salinity considering that there was a marked difference between the concentration in interstitial water (104 µg ℓ^{-1}) and that of the overlying water at the onset of the experiment (2706 µg ℓ^{-1}). The decrease in nitrate-N concentration was therefore probably not the result of any biochemical or biological uptake process, but rather a case of 'benthic–pelagic equilibration'.

Total dissolved ammonium-N, DIP and DRS did not show any clear trends. Total dissolved ammonium-N concentration in the light chamber increased significantly just after sunset but then stabilised. Although this trend correlates well with a drop in DO level, perhaps associated with some heterotrophic process, one would have expected total ammonium-N concentration in the dark chamber to also increase, which did not occur (Figure 9a and 9b). This could also have been linked to bentho-pelagic equilibration as total dissolved ammonium-N concentrations in interstitial water were higher (104 µg ℓ^{-1}) than in the chamber (47 µg ℓ^{-1}). Again, one would have expected a similar response in the dark chamber, which did not occur.

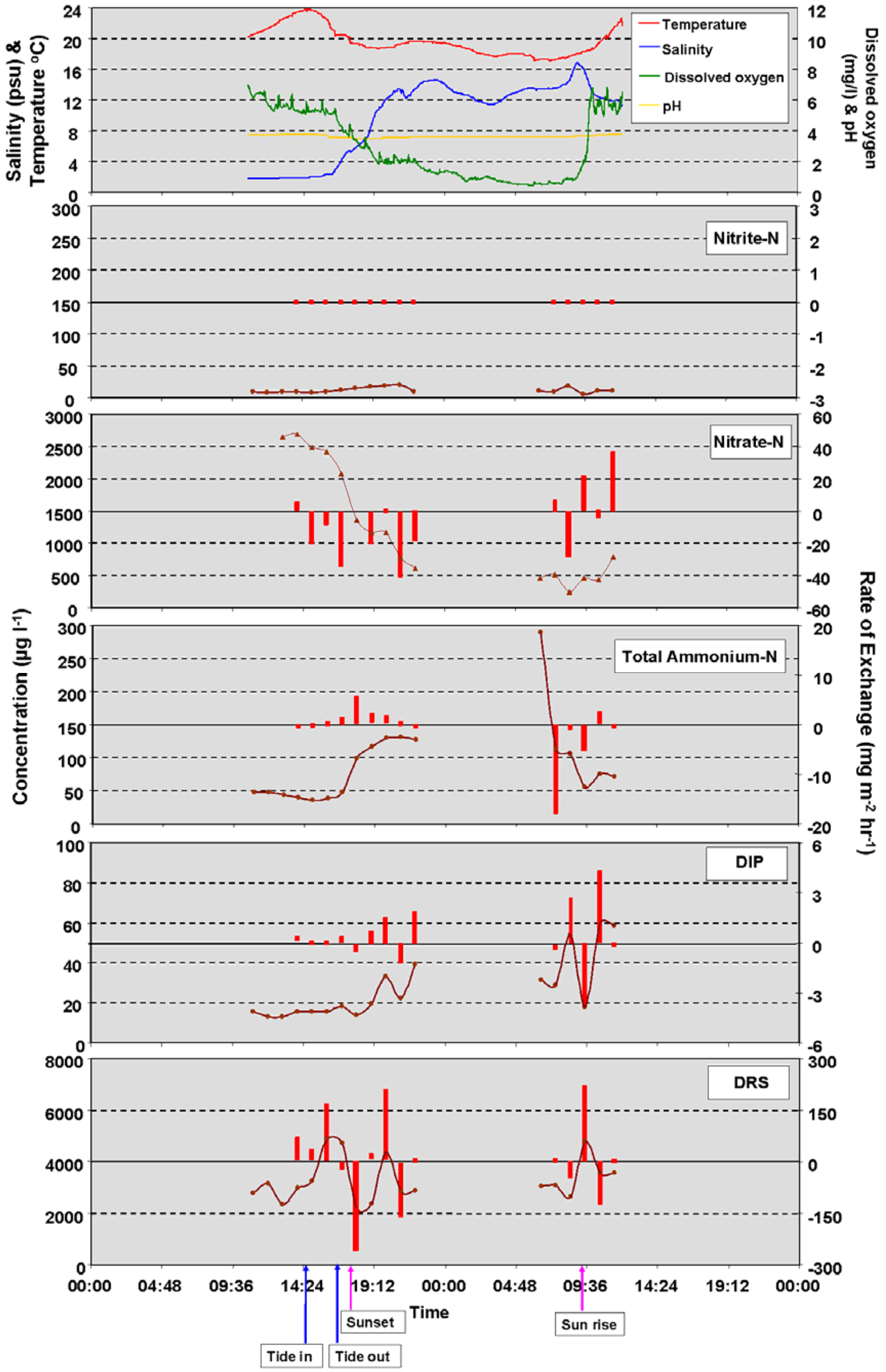

Figure 9a: Temperature, salinity, dissolved oxygen, pH and dissolved nutrient concentrations (brown line graphs) and dissolved nutrient rate of exchange (red bar graphs) measured in the microcosm study (light chamber) on 8/9 September 2006 in the East Kleinemonde Estuary.

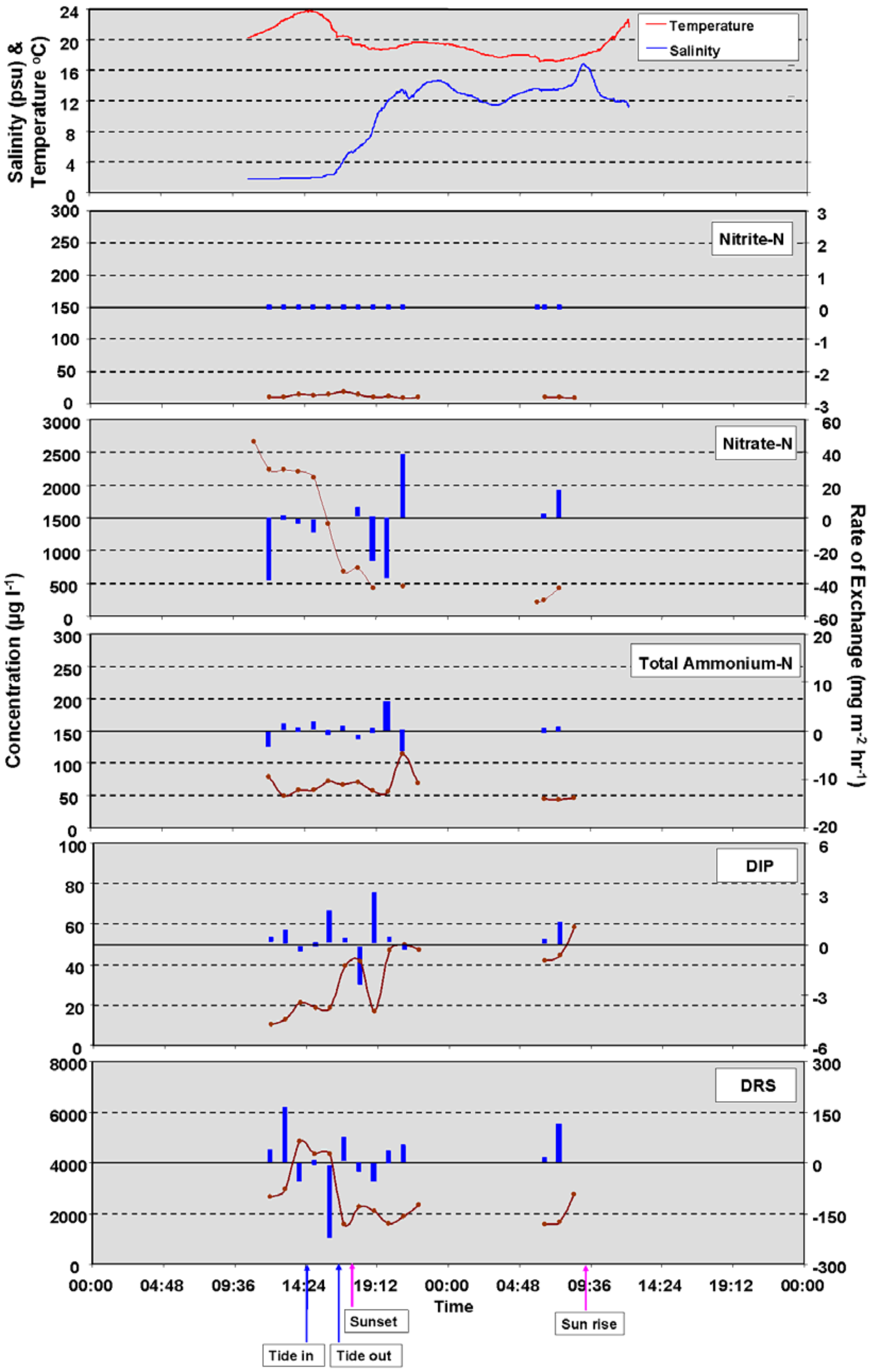

Figure 9b. Dissolved nutrient concentrations (brown line graphs) and dissolved nutrient rate of exchange (blue bar graphs) measured in the microcosm study (dark chamber) on 8/9 September 2006 in the East Kleinemonde Estuary (salinity and temperature from light chamber measurements).

Table 2. Average concentrations (n = 4) measured in interstitial water during the microcosm experiment.

Variable	Concentration
Salinity (psu)	15
Dissolved Nitrite-N (µg ℓ^{-1})	13
Dissolved Nitrate-N (µg ℓ^{-1})	104
Total dissolved Ammonium-N (µg ℓ^{-1})	119
Dissolved inorganic phosphate (DIP) (µg ℓ^{-1})	50
Total phosphorus (interstitial water) (µg ℓ^{-1})	774
Dissolved reactive silicate (µg ℓ^{-1})	2669
Chlorophyll *a* (µg g^{-1}) and (mg m^{-2})*	10.7
Total organic matter** (mg g^{-1}) and (g m^{-1})	0.7 and 825

* Methods as per Snow & Adams 2006 **Determined by loss on ignition

In the case of DIP, there was a weak rise in concentration during the experiment, both in the light and dark chambers, suggesting a net release of DIP into the water column, albeit relatively small. DRS concentrations also fluctuated in both chambers, although there was a small net release of DRS into the water column.

Based on the results from this microcosm experiment, there appears to be relatively little *in situ* regeneration of inorganic nutrients into the water column through biochemical processes (e.g. remineralisation) in the East Kleinemonde Estuary. This further supports the hypothesis that in smaller TOCEs, following prolonged mouth closure and when these systems are largely cut-off from major nutrient input sources (i.e. the river and sea), the water column becomes depleted of nutrients because there is no significant *in situ* regeneration of inorganic nutrients into the water column.

However, taking into account that variation in DO concentrations correlated well with the photosynthesis-respiration cycle, and assuming that benthic microalgae was largely accountable for this trend (e.g. mean chlorophyll *a* in the sediments 10.7 µg g^{-1}), *in situ* regeneration of inorganic nutrients may be significant at the sediment boundary thus sufficient to support benthic productivity.

CONCLUSIONS

The following key conclusions are drawn from this study:

- In general, the East Kleinemonde Estuary fits within the conceptual model proposed in terms of the structure and water quality characteristics of TOCEs under the closed and open states in terms of salinity, temperature, pH, dissolved oxygen, turbidity and inorganic nutrients.
- Results from preliminary microcosm studies seem to support the general hypothesis that in South African TOCEs, *in situ* regeneration of inorganic nutrients through biochemical processes (e.g. remineralisation) are probably not significant in supplying inorganic nutrients to the water column, although these processes may well be important in supplying nutrients for benthic production.
- The East Kleinemonde Estuary appears to be similar to other southern and western Cape estuaries that lie in low rainfall areas with moderate to high evaporation. These become well-mixed brackish systems during the closed state – and sometimes become more saline if the closed state is prolonged. This is in contrast to the KwaZulu-Natal TOCEs, where due to the

generally higher rainfall and higher wave action that causes mouth closure even under fairly high river inflow, these estuarine systems become increasingly fresh during the closed state (e.g. Mdloti) (Perissinotto *et al*. 2004).

Although there are still some unanswered questions linked to the inorganic nutrient variables in TOCEs, the greatest uncertainty lies in the area of organic nutrient cycling, both across the river-estuary and marine-estuary boundaries and *in situ* processes. Further research within this particular field is therefore required if we are to understand productivity in TOCEs on a more holistic scale.

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Appendix F

Specialist Report: Microalgae

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INTRODUCTION

The East Kleinemonde Estuary (33°32'S, 27°03'E) (Figure 1) is a temporarily open/closed estuary (TOCE) that lies within the warm temperate biogeographical region of the Eastern Cape Province (Whitfield 1992). Most TOCEs, as presently understood, are regulated by the amount of river flood flow that is received. The quantity of the inflow is primarily controlled by the size of the catchment including regional climate that influences rainfall patterns. Microalgae (i.e. phytoplankton and microphytobenthos) in a number of these estuarine systems form the base of primary production and thus are an important contributing source of carbon to higher trophic levels (Day 1981).

Since TOCEs are characterised by being cut-off from the sea for a period of time, i.e. months to years, the physical isolation of estuarine water from the sea has a profound influence on the microalgal biomass production, cell densities and community structure. Closed mouth conditions support extensive microphytobenthic communities at the expense of the phytoplankton because water column macronutrients are generally low. Recent studies on estuarine microalgae in TOCEs have, by and large, revealed that biomass as chlorophyll *a* in the microphytobenthos (MPB) form the dominant fraction over the phytoplankton (Perissinotto *et al*. 2000, Froneman 2002a, 2002b, Perissinotto *et al*. 2002, Gama *et al*. 2005). Microphytobenthos biomass has been estimated to be as much as two to three fold greater than that of phytoplankton, mainly when the mouth is closed. Water column stability, light quality and the availability of nutrients from ground water (sediment/water column interface) has been suggested as the reasons for such high biomass values. However, no studies have demonstrated a close link between the quantity of groundwater nutrients, their availability and increased MPB biomass.

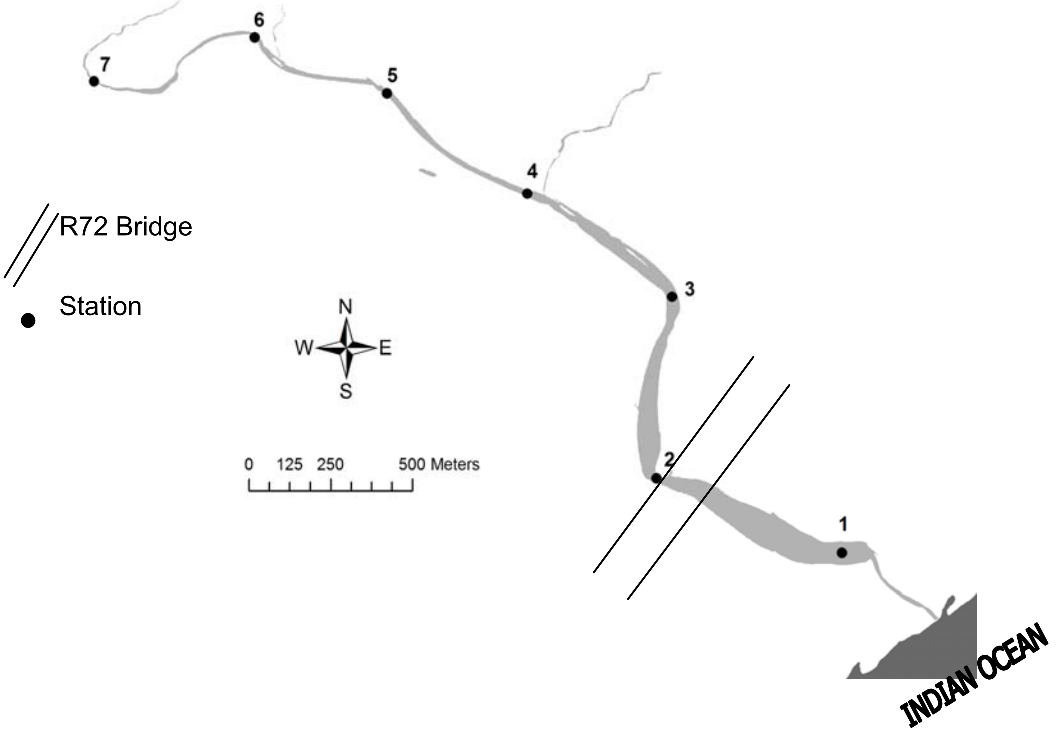

Figure 1. Sampling stations in the East Kleinemonde Estuary during this study.

The botanical component of the East Kleinemonde Intermediate Resource Directed Measures (RDM) study of the East Kleinemonde project required data on microalgae. Data requirements for the reserve determination involved a monthly microalgal survey over a whole year to measure biomass estimates and community structure within in the estuary. The RDM study assessed the present state of the microalgal community, predicted the natural condition and how much the East Kleinemonde Estuary would deviate from its present state under a variety of hypothetical flow scenarios. Mouth opening events from increased river inflow have been shown to significantly influence microalgal chlorophyll *a* concentrations and species composition (Perissinotto *et al*. 2000, Froneman 2002a, Gama *et al*. 2005, Skinner *et al*. 2006).

MATERIALS AND METHODS

Duration and Sampling Strategy

The duration of the project was from 2005-2007 and the field study from March-October 2006. The results reported here cover the latter eight month period. The sampling strategy included quarterly (March, July and September), monthly (March-October), weekly (27 June-19 July) and daily (23-27 June) surveys.

Physico-chemical parameters and chlorophyll *a* biomass

Vertical profiles of the water column were measured at 0.25 m and at 0.5 m intervals for down welling irradiance using a Li Cor 190 underwater 4π quantum sensor linked to a Li Cor 1000 data logger, temperature, salinity, dissolved oxygen, and conductivity using a YSI-650 MDS display logger connected to a YSI-6 series sonde. Horizontal and vertical sampling of physico-chemical variables were carried out monthly and following hydrological events such as mouth breaching and/or closed mouth conditions. These measurements were carried out during high and low flow conditions. Samples for microalgal chlorophyll *a* concentration were collected at five stations along the length of the estuary. Following a flood when the mouth opened, water samples were taken daily for 3-5 days, then weekly for 3-4 weeks in order to capture rapid changes in chlorophyll *a* concentration as a result of changes in river flow after the initial mouth breaching. Duplicate water samples for chlorophyll *a* were collected to estimate phytoplankton biomass (Pérez *et al*. 2006). Phytoplankton community samples were collected using a 3 ℓ Student's horizontal-water sampler from the top (0.5 m below the water surface) and bottom (0.5 m from the sediment bottom) and placed in double distilled pre-washed 1000 ml opaque wide-mouth polyethylene bottles.

Water column samples for chlorophyll *a* analysis were size-fractionated and serially filtered through a 20 μm nitex screen (microphytoplankton), a 2.7 μm Whatman GF/D filter paper (nanophytoplankton) and through a 1.2 μm Whatman GF/C filter paper (picophytoplankton). All filter surfaces were pre-treated with a few drops of a saturated solution of $MgCO_3$ and placed in darkened 20 ml vials, extracted for 24 hr in 90% acetone and kept at 0-4 ^{o}C. Triplicate samples for microphytobenthos chlorophyll *a* were taken quarterly along the length of the estuary (Figure 1) using a 20 mm (ID) corer and placed in 30 ml ethanol for further analysis (Rodriguez 1993). Microalgal community composition was assessed from quarterly samples that had been preserved following standard phytoplankton techniques. Sub-samples were enumerated using a Neubauer haemocytometer under a light microscope at 400x magnification. Phytoplankton taxa were identified to species level where possible, but certainly to the nearest genus using the following identification keys (Bold & Wynn 1981, Prescott 1989, Tomas 1997).

In June 2006, the estuary mouth breached and following this, two stations in the upper reaches were not sampled because the water level was too low. Consequently, the data considered after the mouth opened was for three stations instead of five. The data from the three stations were not significantly different thus the phytoplankton chlorophyll *a* data from each station was pooled and is reported as an average across all stations.

Statistical analysis

Significant differences between data for the effects of sampling date, site and water depth were tested using a three-way analysis of variance (ANOVA). One- and two-way ANOVA analyses were used to test for significant differences between sites and sampling dates with multiple comparisons among pairs of means performed using the Tukey's method (Quinn & Keough 2002).

RESULTS AND DISCUSSION

Physico-chemical parameters

During the study period there were a couple of heavy rainfall events in the catchment that changed the frequency and duration of the mouth stats. Prior to the first sampling run in March, the mouth of the estuary had remained closed for several months. General patterns and specific features pertaining to measured parameters will be referred to as these are used to relate physico-chemical data to those of the microalgae. The mouth was closed prior to the first sampling in March and remained closed until June, when two days of sustained rainfall caused the mouth to breach. During this closed-mouth period the water depth reached a maximum of 6 m at Station 2. Water transparency there, measured with a Secchi disk, ranged between 2.0-2.5 m whereas when the mouth was open it was as low as 0.2 m. When the mouth opened, seawater entered and a vertical and horizontal salinity gradient was established, with salinity levels ranging from oligohaline (1-15 psu) to polyhaline (>30 psu) in the upper and lower reaches, respectively.

Strong freshets following rainfall >50 mm breached the mouth allowing estuarine and seawater to mix. The river water had a high load of silt and suspended matter. These freshwater inputs were responsible for bringing in increased macronutrients that account for improved microalgal production. As discussed in the chapter on water quality, nutrient input as DIP (dissolved inorganic phosphorus), DRS (dissolved reactive silica) and DIN (dissolved inorganic nitrogen,) rose over 2, 5 and 10-fold respectively with the increased river flow compared to previous concentration levels when the estuary had been closed.

Daily and weekly phytoplankton chlorophyll a *responses*

After breaching for the first time on 23 June there were intermittent periods of mouth closure lasting for several weeks. The water column phytoplankton chlorophyll *a* response after the mouth breached was very variable, ranging from 2.8-17.9 µg ℓ^{-1}. The lowest concentration, 2.8 µg ℓ^{-1}, occurred on the first day after the breach, while the highest concentration of 17.9 µg ℓ^{-1} was recorded two days later on the 25 June (Figure 2).

Just prior to the breaching event, nanophytoplankton formed the greatest contributor to total chlorophyll *a* biomass at all five stations. This nanophytoplankton size-fraction contributed over 80% of the total chlorophyll *a* while the picophytoplankton contributed about 18% and microphytoplankton 2%. After the breaching, the pico- and microphytoplankton became important and their contributions to total chlorophyll *a* increased to as much as 75% and 20% respectively. These contributions were also highly variable and may indicate a variation in the availability of macronutrients plus possibly an unequal utilisation of the different phytoplankton size groups (Fernández *et al*. 2003). In addition, the relatively small size of the estuary and increased river flow during the open mouth state gave rise to a short residence time which precluded the necessary time required for the microalgae to establish, develop and reproduce.

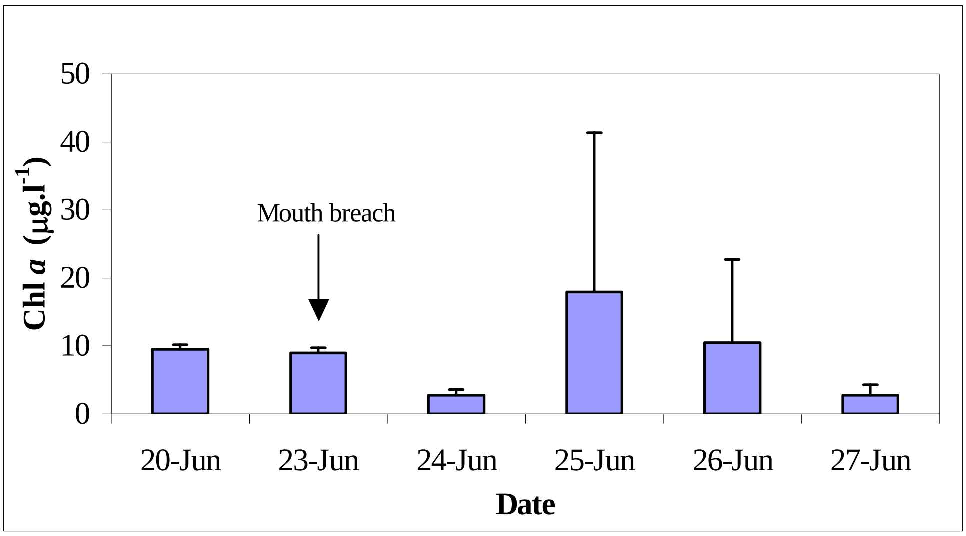

Figure 2. Mean daily chlorophyll *a* concentration taken over six-days (20-27 June 2006) in the East Kleinemonde Estuary. The arrow indicates the day the mouth opened. Vertical lines represent ±1 S.E.

Previous studies on the Mdloti and Mhlanga estuaries have shown that the nanophytoplankton group can contribute in excess of 75% to the total phytoplankton chlorophyll *a* biomass under both open and closed mouth conditions (Perissinotto *et al.* 2004). From the present study, although highly variable, picophytoplankton was the dominant size-fraction immediately after the strong influx of river water that caused the estuary mouth to breach (Figure 3). The weekly phytoplankton chlorophyll *a* response was characterised by a significant reduction ($P < 0.05$) in concentration at stations 1-3. Prior to the opening of the mouth, the chlorophyll *a* concentration in the estuary averaged 9.5 μg ℓ^{-1} but after a week this had decreased to the lowest level of 2.8 μg ℓ^{-1}. Weekly monitoring for the next four weeks showed a gradual increase in chlorophyll *a*, which peaked at 19.8 μg ℓ^{-1} by week five (Figure 4). This pattern was consistent throughout the water column with no significant differences between the surface and bottom layers ($P > 0.05$).

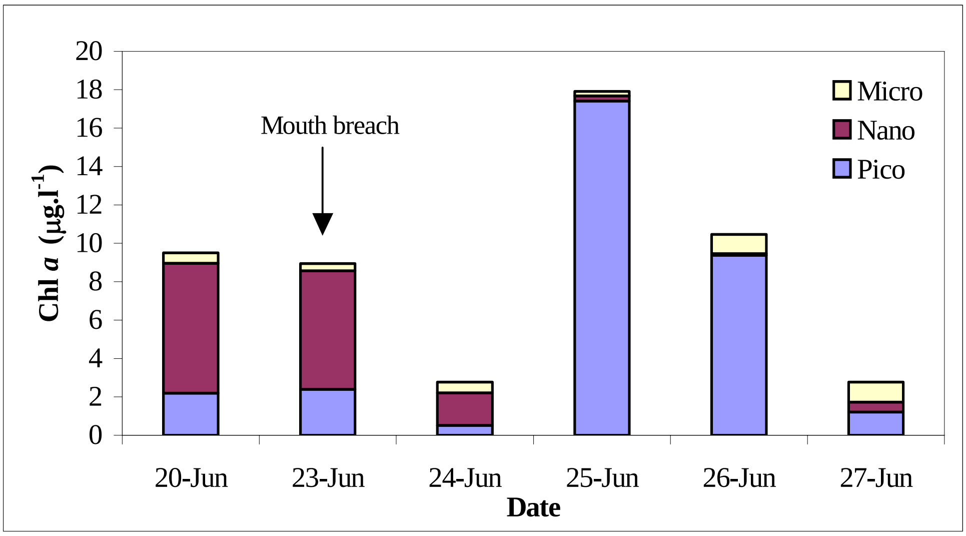

Figure 3. Contribution of the different size-fractions to total phytoplankton chlorophyll *a* measured daily over five days (23-27 June 2006). Arrow indicates day of mouth opening.

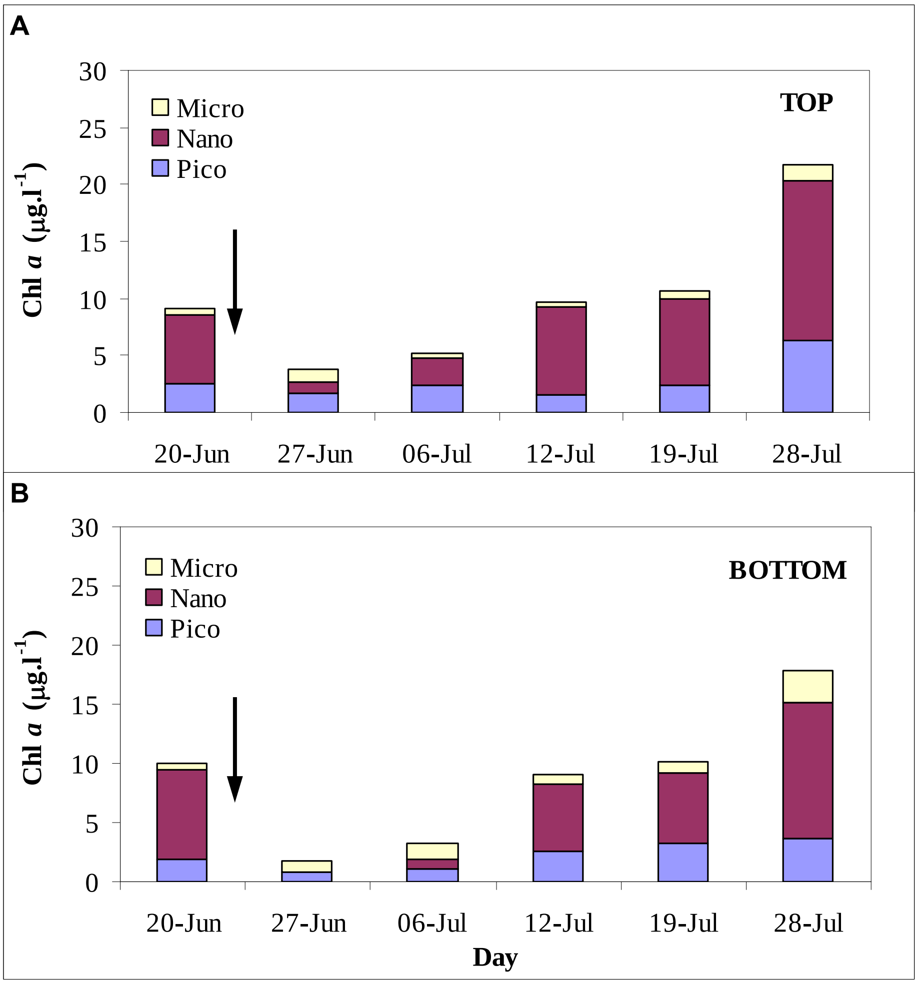

Figure 4. Weekly phytoplankton size-fractionated chlorophyll *a* concentrations taken over six weeks (20 June - 28 July 2006) from the (A) top and (B) bottom layers of the water column (Micro = microphytoplankton, Nano = nanophytoplankton, Pico = picophytoplankton). Arrow shows approximate date when the mouth opened (3 days after 20 June).

In terms of phytoplankton size-structure, although the chlorophyll *a* concentrations were low, the pico-, nano- and microphytoplankton size classes were equally represented throughout the water column one week after the mouth opened. This suggests that all size classes were equally susceptible to the strong river flow that caused the breach. As the river flow eased over the next four weeks, coupled with an increase in the mixing of seawater and freshwater, there was a gradual community recovery with the nano- and picophytoplankton showing the greatest response by contributing ~67 and 27% respectively to total chlorophyll *a*. This is possibly a result of the establishment of a relatively stable riverine and estuarine interface, referred to as the REI zone by Snow *et al*. (2000). This zone is characterised as the region within the water column of maximum phytoplankton biomass brought about by the chemical interactions of particulate organic and inorganic material that form aggregations or floccules when seawater and freshwater mix (Snow *et al*. 2000). The phytoplankton chlorophyll *a* biomass levels recorded five weeks following the opening of the mouth were the highest during the study period (Figure 5). This increase may also be indicative of the time required by phytoplankton to re-establish after a breach by rapidly exploiting the newly available mineral nutrients.

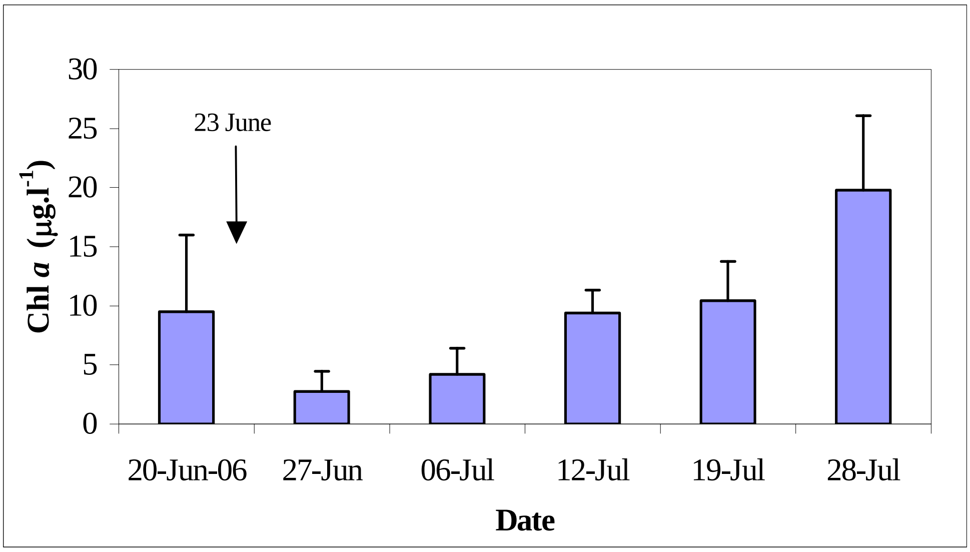

Figure 5. Mean weekly phytoplankton chlorophyll *a* concentrations sampled over six weeks (20 June - 28 July 2006). Arrow shows approximate date when the mouth opened (3-days after 20 June). Vertical bars ± 1SE.

Although there was vertical salinity stratification in the water column during the open mouth state, phytoplankton chlorophyll *a* biomass was not affected by it and there was no vertical stratification of the chlorophyll *a* biomass. Water column chlorophyll *a* distribution patterns did not show significant differences between surface and bottom waters ($P > 0.06$), although some higher concentrations at the surface did exist in the middle reaches of the estuary. A reduction in freshwater flow and an increase in the mixing of seawater and freshwater coupled with a more stable water column environment appear to have generated the suitable conditions for enhanced chlorophyll *a* biomass production by week five. From these data, it seems that five to six weeks are necessary to establish high phytoplankton biomass after a significant breach of the mouth. This observation is in line with what has been reported regarding the period required to produce maximum phytoplankton chlorophyll *a* biomass in estuaries located on the Cape south coast (e.g. Van Stadens Estuary), where a period of approximately eight to ten weeks were needed for the maximum chlorophyll *a* biomass after a breaching event (Gama *et al*. 2005). The phytoplankton size class that was dominant after that period in the Van Stadens Estuary was the microphytoplankton. Studies on the Kasouga (Froneman 2002a) have also demonstrated similar trends, particularly during and post an open mouth phase. This is in contrast to what has been observed in the present study where the dominant group was nanophytoplankton (Figure 4). Except for the period soon after breaching, the nanophytoplankton have been the dominant group in the estuary, contributing over 60% to total chlorophyll *a* biomass even when the mouth had been closed for several months.

The picophytoplankton was the second highest contributor to total chlorophyll *a* biomass, ranging between 1.08 and 5.10 $\mu g\ \ell^{-1}$. The microphytoplankton showed high spatial and temporal variation. There were significant differences between means of the nanophytoplankton in the first week and six weeks after breaching ($P < 0.05$). There were no significant differences between the means of the pico- and the microphytoplankton chlorophyll *a* biomass taken before the mouth was breached (i.e. first week) with those sampled on the sixth week. This suggests that by the sixth week the phytoplankton chlorophyll *a* biomass concentration had recovered to levels observed prior to the breaching event. It may also indicate that, for this estuary, the newly replenished stock of mineral nutrients may have been used up by the phytoplankton, in particular the pico- and nanophytoplankton size-fraction.

Monthly phytoplankton chlorophyll a responses

As with most eastern and southern Cape TOCEs, water column microalgal concentrations are typically low and the East Kleinemonde Estuary is no exception. Monthly phytoplankton chlorophyll *a* concentrations exhibited high spatial and temporal variability. A two-way ANOVA showed no significant differences between surface and bottom depths or among the five sites surveyed ($P > 0.08$). There were also no significant differences between the monthly phytoplankton chlorophyll *a* biomass distribution patterns ($P > 0.05$). The data was thus subsequently pooled across both depth and sites.

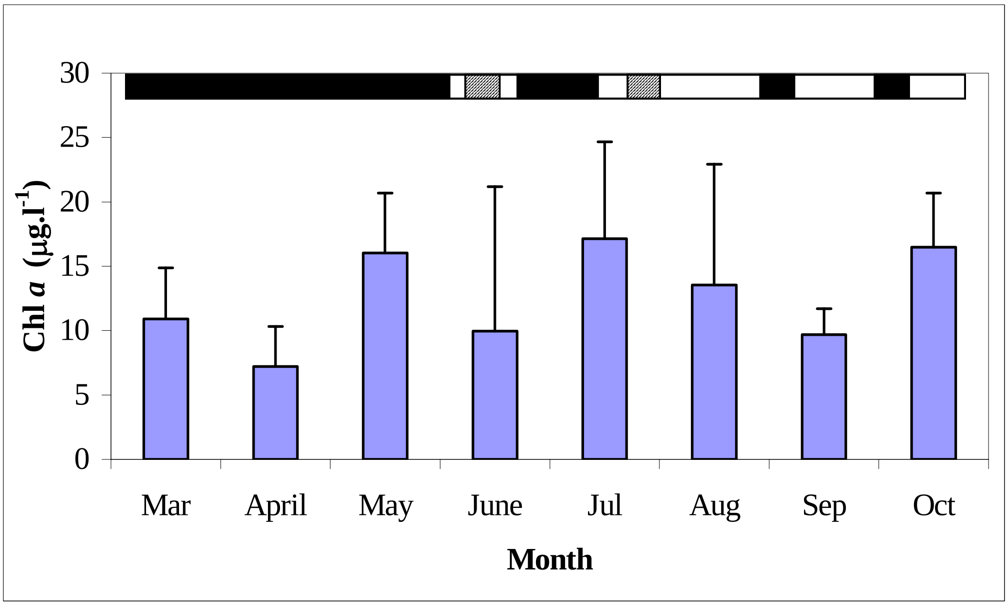

Figure 6. Mean monthly phytoplankton chlorophyll *a* concentrations taken between March and October 2006 in the East Kleinemonde Estuary. Vertical lines represent ±1 S.E. The horizontal bar denotes the three mouth states open bar-open mouth, solid bar-closed mouth, stippled bar overwash.

During the first three months of the study the mouth of the estuary was closed and phytoplankton chlorophyll *a* concentrations reached their peak in May (16.03 µg l^{-1}). Throughout this period the nanophytoplankton were the dominant group with the micro- and picophytoplankton generally co-dominant as the second highest contributor to total chlorophyll *a* (Figure 6). April had the lowest phytoplankton chlorophyll *a* with 7.02 µg l^{-1}. The monthly maximum was 17.13 µg l^{-1} in July six weeks after the mouth opened. A series of intermittent mouth opening and closings including periods of overwash followed for a few days before the mouth remained open for several weeks.

The phytoplankton chlorophyll *a* biomass distribution did not follow a seasonal pattern and were generally evenly distributed over all the months sampled. Seasonality does not appear to be a strong factor controlling phytoplankton chlorophyll *a* biomass in this system, although in the Van Stadens Estuary chlorophyll *a* biomass has been positively correlated with temperature (Gama unpublished data). As with most TOCEs, river flow in the East Kleinemonde has a major influence on phytoplankton chlorophyll *a* concentration and its spatial distribution. After the initial mouth breaching event there were other intermittent mouth opening and closing events that were followed by a longer period of mouth opening, which influenced phytoplankton chlorophyll *a* levels. These irregular mouth states had the effect of reducing as well as stimulating phytoplankton production (Figure 7).

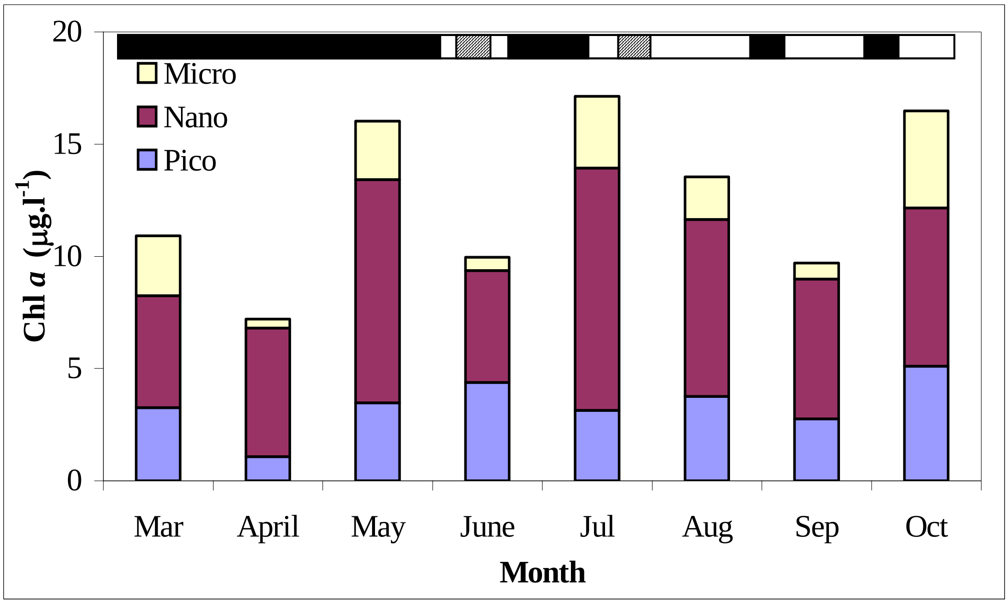

Figure 7. Monthly size-fractionated phytoplankton chlorophyll *a* concentration measured from March-October 2006 in the East Kleinemonde Estuary (Micro = microphytoplankton, Nano = nanophytoplankton, Pico = picophytoplankton). The horizontal bar denotes the three mouth stages (open bar = open mouth, solid bar = closed mouth, stippled bar = overwash).

Temporal phytoplankton chlorophyll *a* distribution patterns measured at the monthly scale may not be adequate in revealing phytoplankton responses as these temporal scales are too coarse and infrequent to capture the rapid changes experienced by small-sized phytoplankton forms (i.e. pico- and nanophytoplankton). Changes in the phytoplankton size structure over short-time scales (e.g. days and weeks) may uncover essential ecological features that may influence food web dynamics that are not observed over longer time scales (Jacquet *et al*. 2002). Although highly variable, the chlorophyll *a* data that were collected daily during the study, does indicate crucial phytoplankton ecophysiological response traits that maybe lost when monitoring regimes are infrequent or over long periods. A number of TOCEs have generally been characterised as naturally nutrient poor (Froneman 2002a, Gama *et al*. 2005) and thus may show increased sensitivity to sudden altered nutrient supplies (Perissinotto *et al*. 2003, Piehler *et al*. 2004) resulting in the unnatural alteration of the food web and trophic balance.

Quarterly phytoplankton, microphytobenthic chlorophyll a *and community responses*

Seven major phytoplankton species were identified in the East Kleinemonde Estuary that belonged to four distinct functional groups (i.e. diatoms, flagellates, dinoflagellates and microflagellates). During March, diatoms were the dominant group at most stations contributing 62% with cryptophytes and dinoflagellates constituting the second highest contributions of 15% and 13% respectively (Table 1). There were no clear spatial distribution patterns observed with the naviculoids and nitzschiods dominant at all the stations. Microflagellates constituted about 10% of the total cell densities in March and made up over 17% in July. A shift in the phytoplankton community composition in July showed the diatoms being replaced by dinoflagellates and cryptophytes, and this similar pattern held throughout September. The high frequency of mouth openings during the study period precluded an extended closed mouth state necessary to test if species diversity declines after a prolonged closed mouth condition. Thus, species diversity remained unchanged when comparing the time before and after the June mouth opening event (H'= 0.4).

Table 1. Major species enumerated at each station along the East Kleinemonde and their associated percent abundance.

	East Kleinemonde Stations Surveyed									
Month	Station 1		Station 2		Station 3		Station 4		Station 5	
		%		%		%		%		%
Mar	Nav.	43.2	Nav.	34.3	Nav.	39.3	Nav.	28.4	Nav.	26.8
	Nitz.	20.5	Nitz.	21.7	Nitz.	30.7	Nitz.	35.9	Nitz.	30.9
	Microf.	6.3	Microf.	12.8	Microf.	9.8	Microf.	6.5	Microf.	16.5
	Pro.	17.4	Cry.	17.7	Cry.	4.7	Cry.	18.7	Cry.	17.7
	Per.	12.3	Per.	13.5	Per.	15.0	Per.	10.5	Per.	8.5
Jul	Eun.	24.6	Cry.	27.4	Cry.	40.2	Eun.	15.0	Cry.	21.7
	Cry.	11.4	Microf.	23.4	Microf.	17.3	Cry.	33.7	Microf.	12.4
	Nav.	16.5	Nav.	10.2	Nav.	9.3	Nav.	11.2	Nav.	18.2
	Per.	34.1	Per.	20.0	Per.	21.0	Per.	15.5	Per.	18.0
	Pro.	12.9	Pro.	19.3	Pro.	12.3	Pro.	24.6	Pro.	29.3
Sep	Eun.	12.8	Eun.	15.2	Cry.	23.6	ND		ND	
	Rhod.	16.9	Cry.	16.3	Rhod.	14.2				
	Nav.	18.7	Nav.	15.8	Nav.	12.5				
	Per.	29.4	Per.	30.7	Per.	34.7				
	Pro.	21.2	Pro.	22.1	Pro.	15.0				

Key to abbreviations: Cry. = Cryptomonas spp.; Eun. = Eunotia spp.; Nav. = Navicula spp.; Nitz. = Nitzschia spp.; Microf. = microflagellates; Pro. = Prorocentrum spp.; Per. = Peridinium spp.; Rhod. = Rhodomonas spp.; ND = No Data collected.

Prior to the sampling survey in March after the long closed state of about 9 months, the microphytobenthos (MPB) did not show any strong spatial distribution pattern. It was characterised by very low chlorophyll a concentrations averaging 7.32 $\mu g\ g^{-1}$ sediment. Under closed mouth conditions these MPBs values are uncharacteristically low given the clear water conditions during that time. In most TOCEs MPBs tend to show high chlorophyll *a* biomass when the mouth has been closed for an extended period (Nozais *et al*. 2001, Froneman 2002b, Perissinotto 2002). As a result of the relative shallowness of the estuary the euphotic depth was always greater than the total depth, except at station-2 near the R72 bridge where the maximum depth >2 m when the mouth is closed.

In July, one week after mouth breaching, the MPB chlorophyll *a* concentration had increased 12 fold over the March levels although the concentration was higher in the mouth area than in the upper reaches (Figure 8). A one-way ANOVA revealed significant differences between March and July MPB chlorophyll *a* concentrations in the lower and middle reaches of the estuary ($P < 0.001$). Microphytobenthic samples were collected a week after the mouth breached. Instead of the expected low concentration as a result of the fall in the water level and possible sediment scour during the breach, they showed higher chlorophyll *a* biomass in the lower reaches compared to the upper station. There were significantly higher MPB spatial patterns during July compared to March and September ($P < 0.05$) (Figure 8). During July and September the mouth was open after it had been closed for less than a week. On both these occasions the MPB chlorophyll *a* biomass did not show any significant change after the mouth closed.

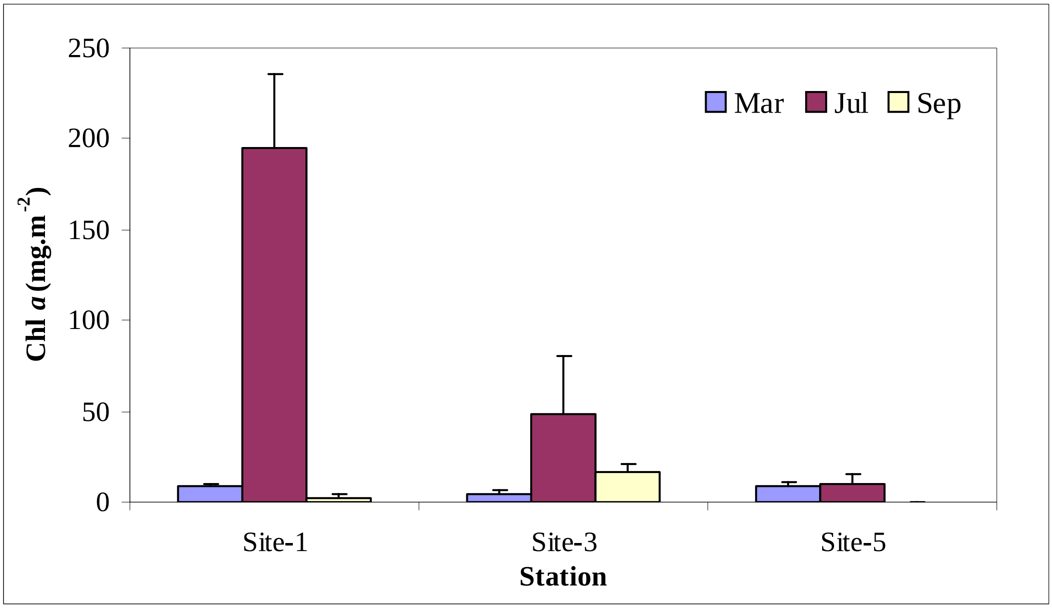

Figure 8. Mean microphytobenthic chlorophyll *a* concentrations taken in March, July and September 2006 from three sites along the East Kleinemonde Estuary. Mouth conditions at the time of sampling were March = closed, July and September = open. Vertical lines represent ±1 S.E.

Increased nutrient inputs took place when the mouth was breached in winter as well as during the intermittent opening events in late winter. At these times the phytoplankton chlorophyll *a* biomass increased five to six weeks after each breach but the microphytobenthos chlorophyll *a* biomass showed an increase only a week after the mouth breached and most of the elevated concentrations occurred only in the lower reaches. This is possibly due to the sediment becoming stabilised sooner than the time necessary to create a favourable environment for increased phytoplankton production. By September MPB chlorophyll *a* biomass had declined to levels observed prior to the mouth opening, although during this period the estuary was experiencing many more mouth opening events than expected, which possibly disrupted the sediment and the development of benthic microalgae. It was not possible, therefore, to determine if a switch would have taken place from a phytoplankton driven production to one driven by benthic microalgae, as the chlorophyll *a* biomass levels in the sediments remained very low (see Figure 8). During this same period the phytoplankton chlorophyll *a* levels were at or near their highest levels. Under closed mouth conditions, seepage areas are generally associated with groundwater input into the estuary. Around these sites dense communities of MPB develop that are important in the biogeochemical cycling of minerals. In the East Kleinemonde Estuary samples were collected in March when the mouth was closed, however during the subsequent quarterly visits the mouth was open and the water level in the estuary had dropped and the previous collection sites were exposed and dry. The data for the March samples indicated a benthic microalgal community composed of mainly bacillariophytes, chlorophytes and mats of cyanobacteria with a proportional representation of 75%, 15% and 10% respectively.

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Appendix G

Specialist Report: Macrophytes

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INTRODUCTION

Macrophyte community structure and composition in TOCEs is influenced by freshwater inflow, tidal exchange, salinity, water level fluctuations and sediment dynamics. Unlike many small TOCEs in KwaZulu-Natal and the south-western Cape whose mouths open seasonally following periods of high rainfall (Bennett 1989, Harrison & Whitfield 1995), the opening of the East Kleinemonde Estuary is unpredictable. The response of macrophytes in these estuaries is event driven. Biomass loss can either be partial or complete as a result of removal by floods or die-back in response to a drop in water level. The ability to colonise available space and re-establish after a disturbance is essential for the long-term persistence of communities and depends on the availability of propagules and the seed bank. The low species richness of macrophytes in TOCEs means that species lost due to changing physico-chemical conditions represents a loss of diversity. The East Kleinemonde Estuary historically had large beds of submerged macrophytes. However, since the early 2000s these have disappeared and the loss of habitat has resulted in the absence of key fish species, such as the estuarine pipefish (Whitfield & Cowley pers. comm.).

MATERIALS AND METHODS

In order to understand the spatial and temporal distribution of macrophytes plant cover abundance, biomass and seed bank structure was measured. Historical aerial photographs were assessed to determine the past vegetation status.

Present status of macrophytes in East Kleinemonde Estuary

Vegetation map

A vegetation map for the present open mouth conditions was completed by Dr T. Bornman (Botany Department, NMMU). The survey included the capturing of photographic records, spatial data (GPS & ArcPad® version 6) and the collection of plant material for identification. The spring low and high water was mapped in August 2006 during the open mouth state to record the extent of tidal inundation (vertically and horizontally). These data combined with the DWAF water level data will be used in the vegetation model to predict the extent of inundation during open mouth conditions.

Plant cover abundance

Three permanent transects (Figure 1) were assessed on a monthly basis from March 2006. These transects were also monitored in 2005 on a quarterly basis (Bezuidenhout 2006) but this overlooked rapid changes and the sampling strategy was changed to monthly for 2006. Duplicate 1 m^2 quadrats were placed every 5 m along the length of each transect and vegetation was assessed for species percentage cover. Monthly colonisation rates (change in mean percentage cover) were calculated for each of the dominant species. The assessment in 2005 included sediment physico-chemical properties, namely electrical conductivity, sediment organic content, particle size, depth to groundwater and groundwater electrical conductivity. These properties did not influence the distribution of macrophytes and were therefore not continued in 2006. Water column physico-chemical conditions were measured monthly and daily when the mouth opened. Water depth was measured along each transect when the mouth was closed. To test the hypothesis that salt marsh dies within 120 days of inundation, regression analysis between cover abundance for *Sarcocornia perennis* and water depth was performed. Because the health of the vegetation at the time of sampling is a reflection of preceding water levels (Steinman *et al*. 2002), a one-, two-, three- and four-month time lag was used, i.e. the water depth in the preceding months.

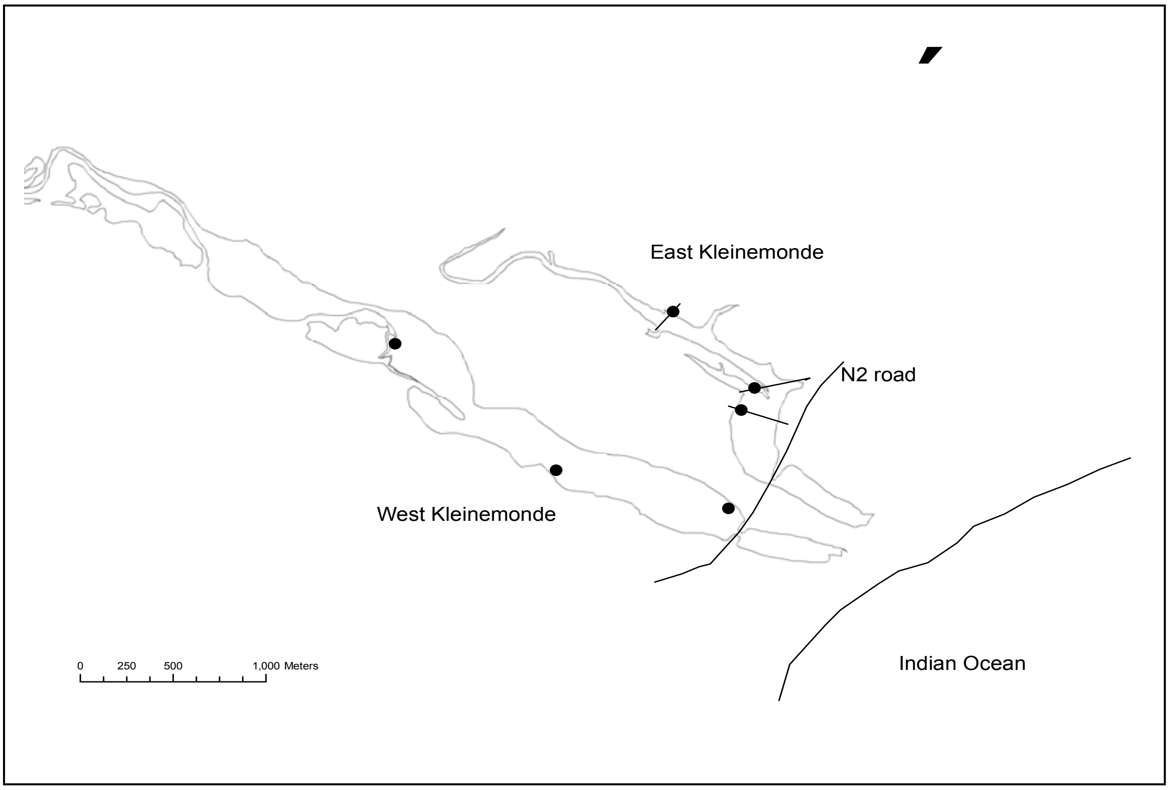


Figure 1. Location of the permanent transects for plant cover abundance (▬) and seed bank sampling sites (●) in both the East and West Kleinemonde estuaries.

Biomass

Optimum conditions for submerged macrophyte growth (stable water levels, clear water) were not present in the East Kleinemonde during 2006. The adjacent West Kleinemonde had almost pure stands of *Ruppia cirrhosa* due to extended mouth closure for two years. Biomass of *Ruppia* in this estuary was included to provide an indication of maximum potential biomass for the East Kleinemonde Estuary. Biomass was sampled in March, May and August 2006 in the West Kleinemonde Estuary and when the plants were present in the East Kleinemonde Estuary (March, May and June). Biomass was measured by clearing an area within a submersed bottomless bucket (30 cm Ø, n=4). All above and below-ground material was harvested and measurements represent total biomass. This was done to overcome variation due to seasonal allocation between above-ground and below-ground biomass. Samples were placed in sealed plastic bags and transported to the laboratory where they were washed free of animals, silt and debris, through a 1 mm sieve. Dry mass was determined after air-drying at 60°C for 48 hours until a constant dry mass was achieved. Results were expressed as g dry mass m^{-2}.

Seed bank structure

Sediment cores were collected from three sites in both the East and West Kleinemonde estuaries (Figure 1) in early autumn (19 March 2006). This time of collection represented the end of the growing season of the dominant species and seed bank replenishment was assumed to be high. Sediment was also sampled in May to determine the distribution of seed with depth and in August to determine the influence of flooding and mouth opening on the seed bank. These data are not included in this report. At selected sites, a 10 x 10 m plot was selected within which 45 random cores were sampled (4 cm Ø and 5 cm depth), representing a total collection surface area of 0.012 m^2 for each site. To account for the spatial heterogeneity of seed distribution, samples from each site were aggregated into 3 ℓ buckets and homogenised. To test the potential for habitat development given ideal conditions, as well as to quantify seed banks, a 3x3 factorial experiment with three replicates per treatment was established in a glasshouse at Nelson Mandela Metropolitan University (NMMU). Treatments represented three salinities; 0, 17 and 35 PSU and three water level conditions; exposed, waterlogged (pots in standing water) and submerged (inundated with 5 cm water). This provided a range of environmental conditions to promote maximum germination. The seedling emergence method was used to measure germination potential for different salinity and hydrological conditions (Peterson & Baldwin 2004, Boedeltjie *et al*. 2002, Baldwin *et al*. 2001, Baldwin *et al*. 1996, Leck & Simpson 1995, Van der Valk & Davis 1978). The cumulative number of seedlings present at the end of the incubation period was extrapolated to get the number of viable seeds per m^2. Direct counts of the remaining sediment were done to determine total seed bank since some seeds may be dormant. Percentage germination indicates whether seed banks are transient (remaining in the sediment for less than one growing season) or persistent (more than one growing season).

Past status of macrophytes in the East Kleinemonde Estuary

A total of 11 aerial photographs obtained from the South African Department of Surveys and Mapping were used to assess historical vegetation changes. These spanned a 20-year period; the earliest was 1934 and the latest 2004. Photographic interpretation and quantitative mapping was done using ArcGIS™ Version 9.0 software. Historical rainfall data for Port Alfred, obtained from the South African Weather Bureau and water levels for the East Kleinemonde Estuary were used to assist with photographic interpretation.

Predictive vegetation model

Elevation profiles for the three transects were measured using a Wild Heerbrugg Dumpy Level and the positions recorded using a GPS with ArcPad® (Version 7) software. These data give an indication of total area of inundation under open mouth conditions. Together with changes in cover abundance and past vegetation status, a predictive vegetation model for mouth condition and water level scenarios was formulated. This will be extended for the entire estuary using bathymetric data.

RESULTS AND DISCUSSION

Present status of macrophytes

Figure 2 shows the present distribution of vegetation in the East Kleinemonde Estuary under open mouth conditions. The potential area for submerged macrophyte development under favourable conditions, i.e. stable, clear water levels, has been included. The area covered by the different vegetation types is shown in Table 1.

Plant cover abundance

Figure 3 shows the change in the mean cover abundance for intertidal salt marsh (monospecific stands of *Sarcocornia perennis*), submerged macrophytes (averaged for *Ruppia cirrhosa, Chara vulgaris* and *Halophila ovalis* and reed and sedge stands (*Phragmites australis*) dominated. Data were averaged for the three transects and water level values of 0 m imply an open mouth. Table 2 shows the colonisation rates calculated as the average change in m^2 per year. Supratidal salt marsh is not presented because this community is only influenced by extremely high water levels under closed mouth conditions.

As water levels increased under closed conditions, the cover abundance *of Sarcocornia perennis* decreased. Regression analysis of mean cover abundance against water depth showed the strongest association of cover with water level one month preceding sampling ($R^2 = 0.196$). The plants showed rapid recovery from existing material together with seedling establishment when water levels were low.

Table 1. Area cover for each of the habitat types in the East Kleinemonde Estuary.

Estuarine Vegetation Type	**Cover (ha)**	**Comments**
Intertidal saltmarsh	2.45	*Sarcocornia perennis* dominant.
Supratidal saltmarsh	2.77	*Sporobolus virginicus* and *Sarcocornia pillansii* dominant, limited *Limonium scabrum.*
Saltmarsh mosaic	1.78	Mixture of *Juncus* spp., *Sarcocornia* spp., *Triglochin striata, Samolus* porosus, *Sporobolus virginicus*, *Bassia diffusa* and other sedges.
Salicornia	1.02	*Salicornia* sp. dominant. May become bare during exposed periods or submerged during closed mouth phases.
Paspalum	0.57	Wetland grass growing in the water, prefer closed and stable water levels.
Juncus	1.81	*Juncus kraussii* dominant but also *Juncus acutus.*
Reeds and sedges	1.01	*Phragmites australis* dominant.
Exposed mudbanks	9.83	Mudbanks exposed during spring low tide (open phase). These mudbanks had very limited vegetation growth during 2006, but may be colonised by *Sarcocornia* spp. during the open phase and submerged macrophytes during extended closed periods.
Estuarine water area	12.69	Measured during spring low tide (open phase). Includes side channels and streams influenced by tidal exchange.
Sandbank seaward of the bridge	1.78	Only large sandbank occurs seaward of the bridge. Mapped during spring low tide (open phase).
Total	**35.71**	
Potential submerged macrophytes	14.5	Given high, stable water levels and brackish (< 15 psu) conditions.

Submerged macrophytes increased under closed mouth conditions. This change took place exponentially (Figure 3). This is typical of studies elsewhere where *Ruppia* spp. recolonisation was shown to be a self-accelerating process because of positive feed back, i.e. increased plant cover increases light conditions and therefore increases growth (Hemminga & Duarte 2000). The submerged macrophytes colonise the open areas slower than *Sarcocornia* because growth occurs from seed since a complete loss of biomass occurred after each mouth opening as a result of desiccation and die-back. *Sarcocornia* by contrast recovered from existing vegetation because high water levels were not experienced for long enough to result in the complete removal of existing vegetation.

Phragmites australis showed a decrease in cover abundance under open mouth conditions, as well as with increasing water level (Figure 3). In systems where sedimentation poses a problem, colonisation values of 0.15 ha yr^{-1} (Siyaya Estuary, Benfield 1984) and 0.13 ha yr^{-1} (Nhlabane Estuary, Riddin 1999) has been recorded.

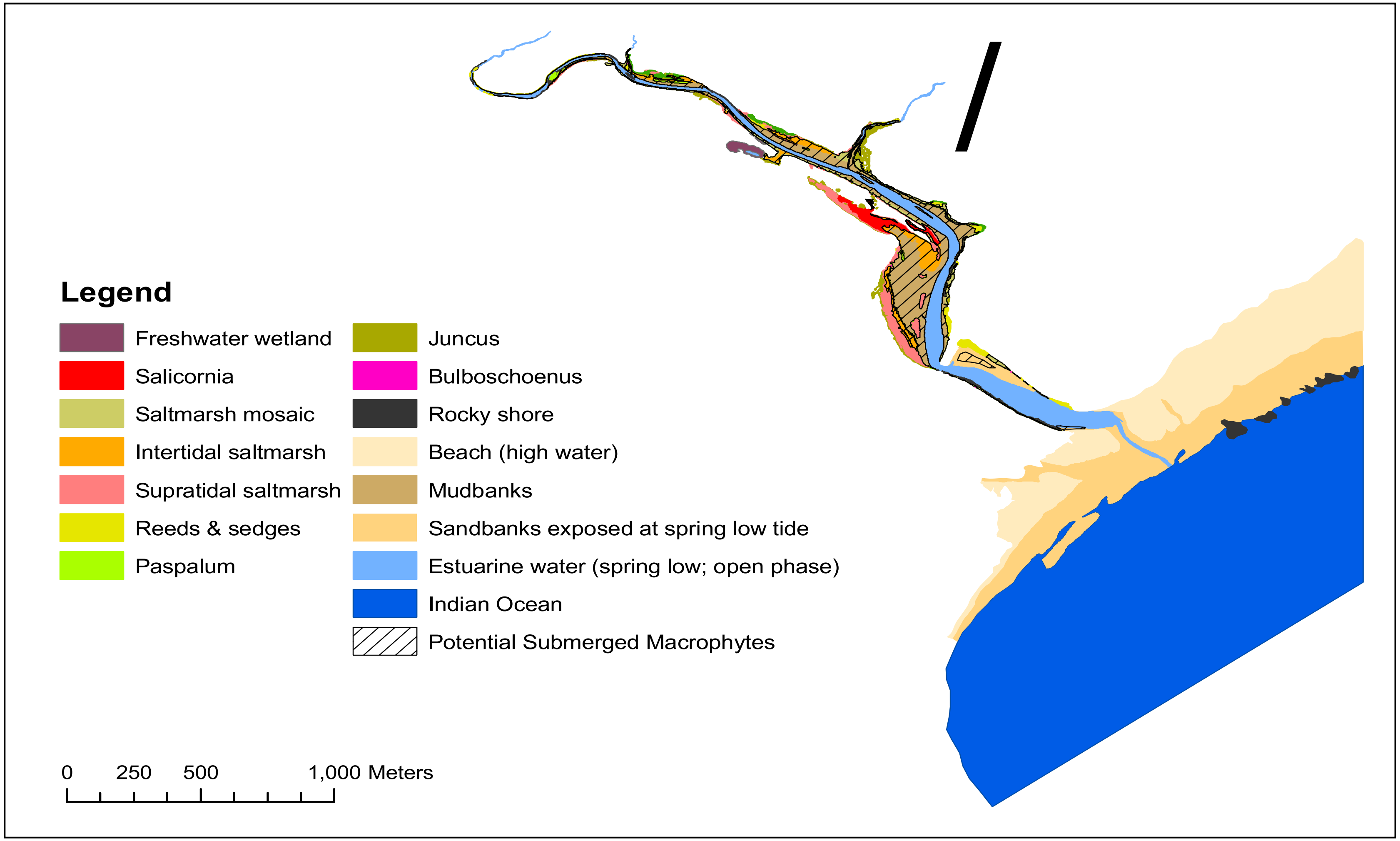


Figure 2. Present vegetation map of the East Kleinemonde Estuary under open mouth conditions.

Biomass

Maximum potential biomass (2883 g dry mass m^{-2} in August 2006) for *Ruppia cirrhosa* in the West Kleinemonde is shown in Figure 4. By comparison, the maximum value for East Kleinemonde was 706 g dry mass m^{-2}. This value would have increased had water levels remained stable for long enough. In Fra Ramon, a temporary lagoon in the Mediterranean, Gesti *et al*. (2005) found maximum biomass of 800 g dry mass m^{-2} where *Ruppia* had a perennial life cycle. They concluded that reproductive effort is highly plastic and this is an adaptation to temporarily flooded areas.

Halophila ovalis had a maximum biomass of 101 g dry mass m^{-2}. *Halophila* populations are characterised by low biomass relative to other seagrasses (Duarte & Chiscano 1999). However *Halophila ovalis* has a fast leaf turnover, which enables it to act as a pioneer (Vermaat *et al*. 1995). Biomass in the Swan Canning Estuary (Western Australia) ranged from 60-120 g dry mass m^{-2} (Hillman *et al*. 1995).

The maximum biomass for *Chara vulgaris* in the East Kleinemonde was 599 g dry mass m^{-2} in June 2006. In sections of Transect 2 100% cover of this species was recorded. *Chara* spp. are the first colonising species after a disturbance and as a group they are usually regarded as indicators of healthy, clear-water ecosystems (Krause 1981). They are characterised by relatively high biomass, high plant density and low growth form (< 20 cm in height). Other studies have reported biomass values between 20-200 g dry mass m^{-2}, although up to 500 g have been recorded (Steinman *et al*. 2002, Fernandez-Alaez *et al*. 2002). Submerged macrophyte biomass increased in response to an increase in water level in both estuaries.

Seed bank structure

The dominant species represented in the seed bank of both estuaries were *Sarcocornia perennis*, *Ruppia cirrhosa* and *Chara vulgaris*. *Potamogeton pectinatus* Linnaeus also occurred but in small amounts. Praeger (1913) and Combroux and Bornette 2004) found that seeds of *P. pussilus* can float for more than one day and their presence could be attributed to *ex situ* transport during a mouth opening event. The lack of *P. pectinatus* seeds may suggest that establishment is by fragmentation or individuals uprooted during floods and not by seeds. The first species to emerge in the laboratory study was *Sarcocornia* after three days. Similar observations were made in the field where seedling emergence occurred within one week of the water level dropping as a result of mouth opening. Other studies have also shown that the germination period in halophytes is very short, between 2 and 12 days (Rubio-Casal *et al*. 2002, Naidoo & Kift 2006, Naidoo & Naicker 1992).

The intermittent germination observed over the 91-day laboratory study is an adaptation of halophytes to unpredictable environments and occurs as a result of seed polymorphism, where the parent plant produces seed of different sizes. Seed densities for *Sarcocornia perennis* ranged from 11 309 - 105 121 m^{-2}. These figures compare favourably with other studies where densities for *Salicornia europaea* were 32 096 - 142 906 m^{-2} (Philipupillai & Ungar 1984). *Ruppia cirrhosa* emerged after 10 days, with some only emerging after 32 -70 days of submergence. De Winton *et al*. (2000) reported emergence after 37 days of inundation. Seed density for *Ruppia cirrhosa* was 2 947 - 5 105 m^{-2}. The pioneer species *Chara vulgaris* germinated after 10 days and seed density was 5 105 - 22 596 m^{-2}. Van den Berg (1999) suggested that under closed canopies of *Chara*, oospore densities of about 10 000 m^{-2} could develop. *Halophila ovalis* was not found in the seed bank, possibly because these plants are short and seed fall is immediately incorporated into the sediment from where there is limited dispersal. The areas where the sediment cores were taken did not have *Halophila* growing in them, and this species only occurred in the main channel along Transect 1.

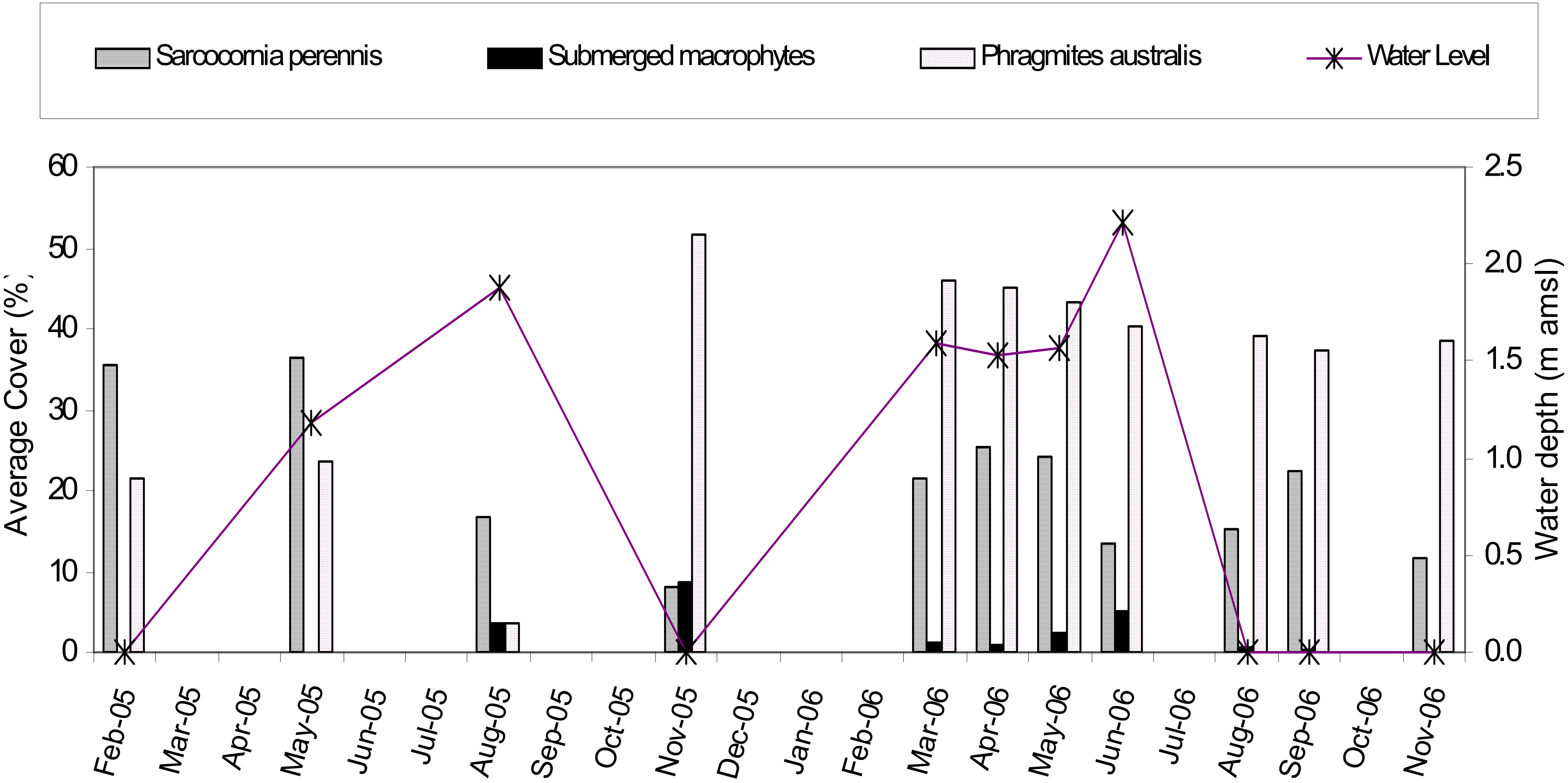
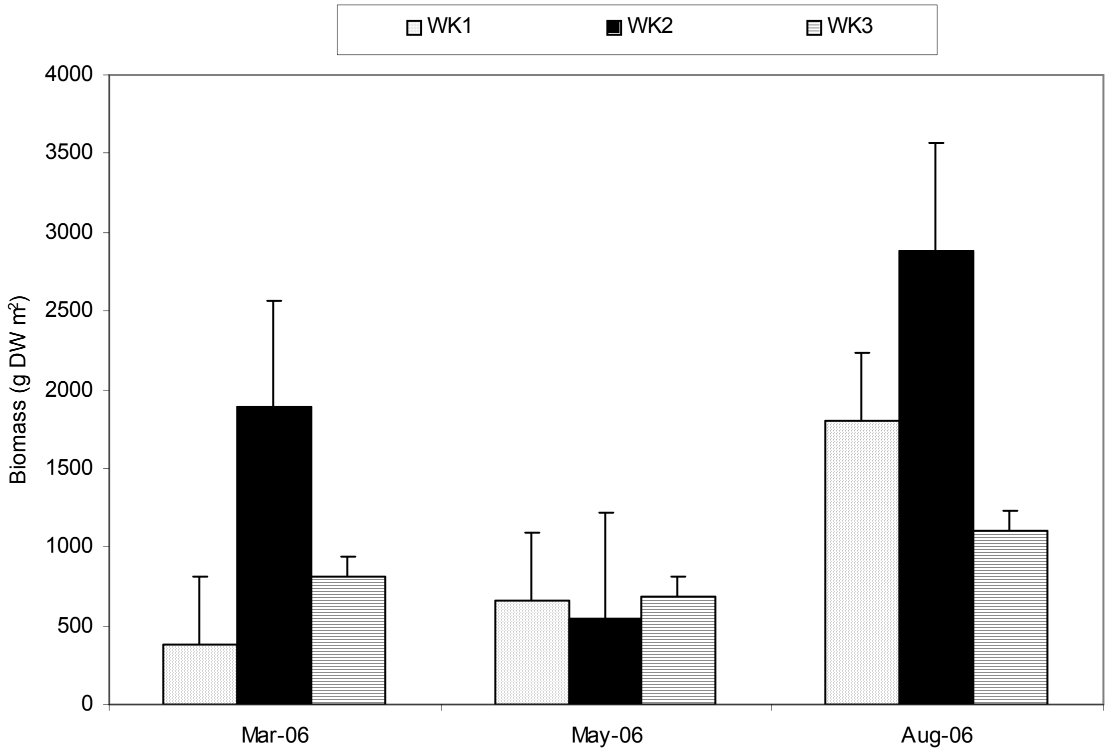


Figure 3. Change in average cover abundance of dominant macrophytes in the East Kleinemonde Estuary during 2005 and 2006. A water level of 0 m represents open mouth, tidal conditions.

a)



b)

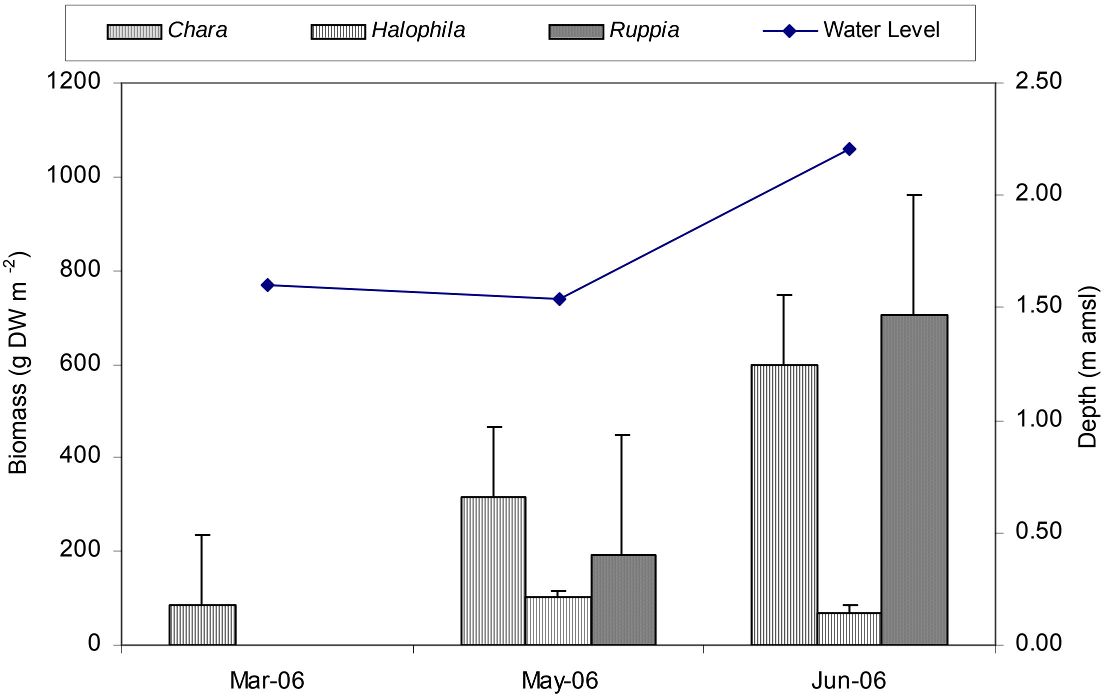


Figure 4. Biomass of *Ruppia cirrhosa* in West Kleinemonde Estuary (a). Biomass of submerged macrophytes in the East Kleinemonde Estuary (b).

Past status of macrophytes

The past status of the macrophytes was assessed using aerial photographs in conjunction with discussions with residents and scientists who had either lived or worked in the area for some time. The following events had a significant effect on the macrophytes:

- Historical rainfall records (Figure 5a) show that in 1939, 1974 and now in 2005/2006 rainfall figures of 862.4 mm, 1254.7, 1286 mm (2005) and 836.5 mm (2006) were recorded. This indicates the occurrence of cyclical 1:30 year floods. The hypothesis is that these floods removed large quantities of sediment, reducing the base level of the estuary and increasing frequency and duration of mouth opening, as observed at present. Over time the estuary will once again build up due to the natural net gain of sediment.

- Construction of the R72 Bridge over the East Kleinemonde Estuary began in 1971. A report on the sediment dynamics of the East and West Kleinemonde (Badenhorst 1988) stated that the bridge had not influenced the rate of sedimentation in the estuary. However localised sedimentation is occurring downstream of the bridge on the east bank where *Phragmites australis* growth has been encouraged.

- Submerged macrophytes were extensive in the 1990s and early 2000s (14.5 ha, Figure 2) as a result of stable water level conditions following prolonged mouth closure (maximum of 2 years in the late 1990s, Figure 5b). The dominant species was *Ruppia cirrhosa*, but as salinity decreased with freshwater inflow, *Potamogeton pectinatus* grew. When the mouth opened in August 2001, large beds of *Potamogeton pectinatus* were still observed in the middle and upper reaches. Because the East Kleinemonde was perched, breaching probably resulted in only partial drainage. The estuary remained brackish as the small tidal exchange would not have caused salinity to greatly increase.

- Reed beds at the end of Transect 1 (on the east bank above the R72 bridge) have only established since 1995. Before this the submerged macrophyte *Ruppia cirrhosa* occurred there up to water depths of 1 m (Cowley pers. comm.). These changes can be attributed to disturbance of the banks due to residential development, fertilizer and freshwater runoff from the adjacent lawns.

PREDICTIVE VEGETATION MODEL

The change in cover abundance of intertidal salt marsh in response to inundation shows that a water level of 1.5 m for two months had a negative influence. Using this information, together with elevation profiles of the three transects, present and past vegetation status and physico-chemical conditions, the following three estuary scenarios were proposed with respect to vegetation change (Figures 7, 8 and 9):

- Open mouth conditions (tidal)
- Closed mouth, low water levels < 1.5 m MSL
- Closed, high water levels > 1.5 m MSL.

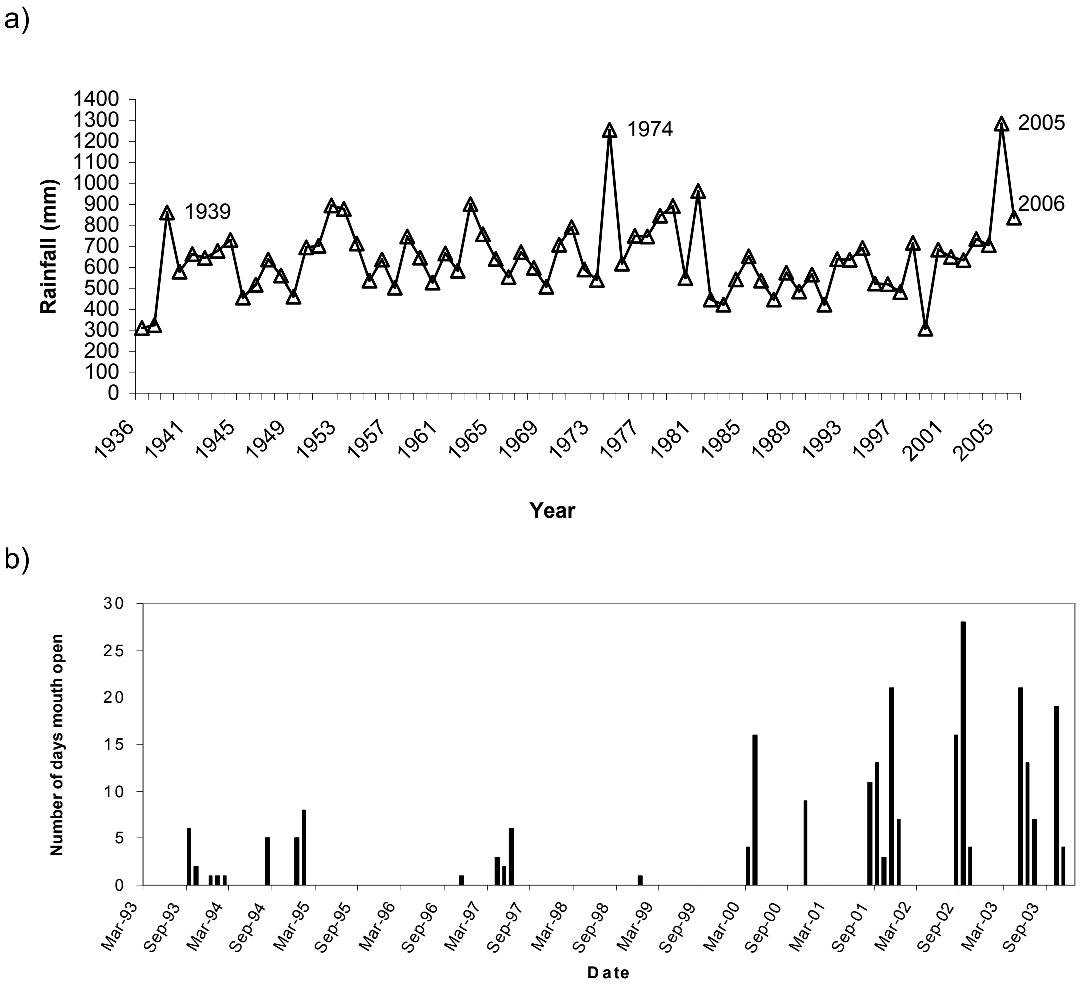
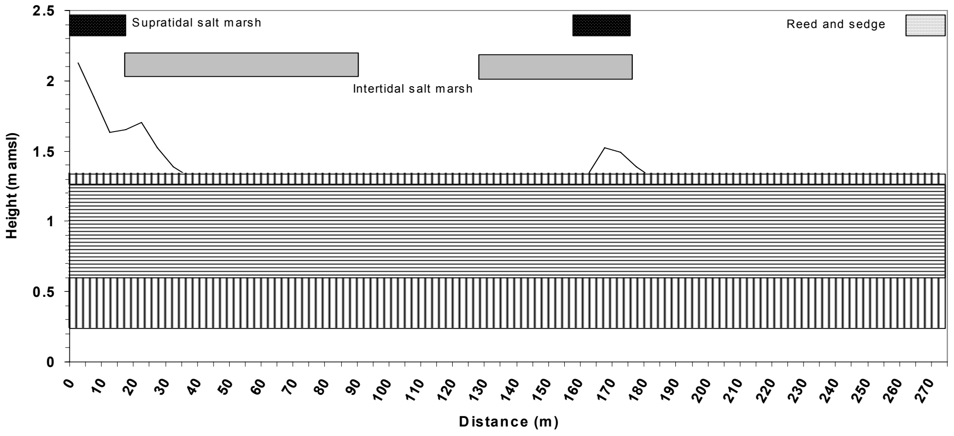


Figure 5. Historical rainfall (1936 to 2006) for Port Alfred (a) and mouth status (b) of the East Kleinemonde Estuary (Source: Cowley pers. comm.).

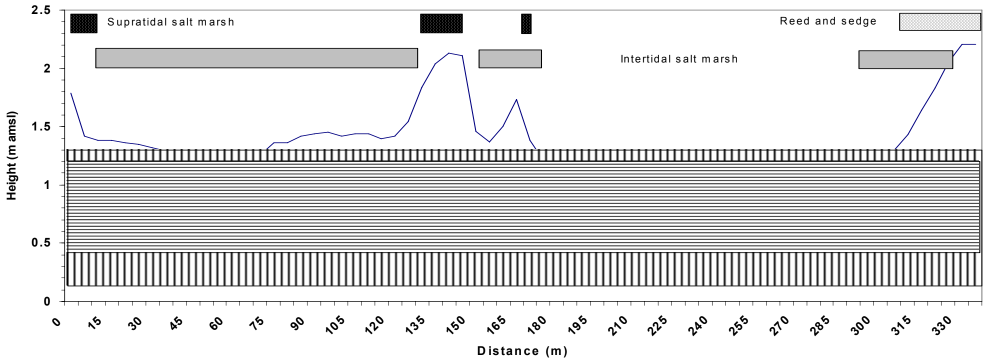
Open, tidal phase

The flood in August 2006 scoured the estuary, reduced the base level and increased tidal amplitude. Intertidal areas are usually characterised by low species richness (three in this study) as only a few species can resist prolonged anaerobic conditions (Silvestri *et al*. 2005). Under the open mouth condition, spring tide ranged from 0.008 to 1.326 m MSL with a tidal amplitude of 1.318 m. During neap tide it was 0.484 to 1.228 m, a tidal amplitude of 0.744 m. Increased water level and salinity fluctuations create unstable conditions with possible losses of salt marsh (Baldwin & Mendelssohn 1998). Permanently inundated areas are bare (Figure 6) and high sediment salinity (45 psu) also prevents recolonisation. Under tidal conditions, salinity ranges from 0.7 to 35.5 psu. *Sarcocornia perennis* is confined to areas above 1 m MSL (Adams & Bate 1994). Seedling emergence takes place within a week of mouth opening and intermittently thereafter. Supratidal salt marsh is only affected by water levels greater than 1.8 m MSL because of elevation. No submerged macrophytes develop because of substrate instability, high water velocity (1 m s^{-1}, Mackenzie pers. comm.) and high turbidity (3-11 cm Secchi depth).

Transect 1



Transect 2



Transect 3

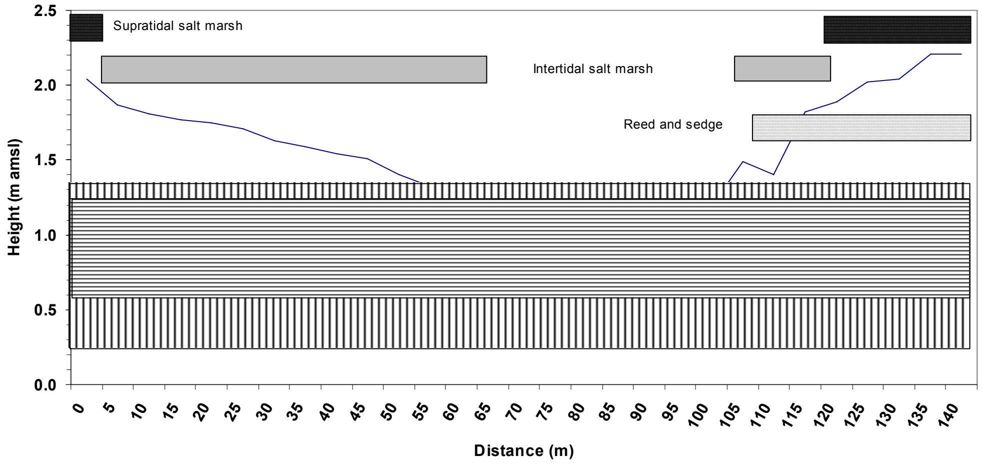
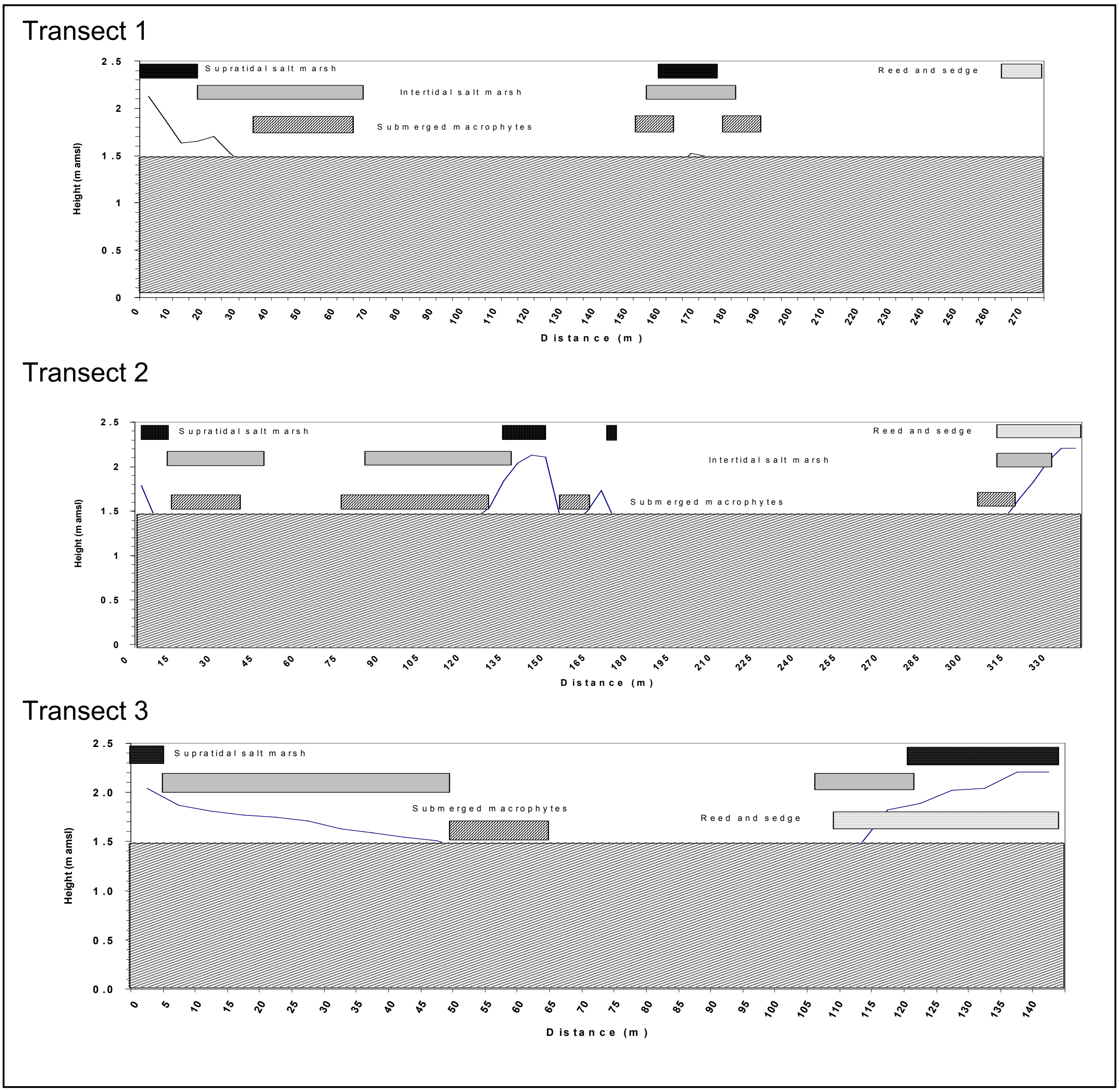


Figure 6. Predictive vegetation model for the East Kleinemonde Estuary under open mouth conditions.

Closed mouth, low water levels (< 1.5 m MSL)

Intertidal salt marsh is negatively affected by inundation for 2 months or longer. Supratidal salt marsh remains unaffected by this low level since it occurs at elevations of > 1.8 m MSL. The submerged macrophytes *Chara vulgaris* and *Ruppia cirrhosa* establish from sediment seed banks after inundation periods of 10-18 days, although more likely only after 30 days. This implies that the mouth needs to be closed for at least this period. Field observations showed that *Chara* did develop in a water depth as little as 10 cm. It can produce oospores within 63 days, providing the water level remains stable (Garcia pers. comm.). Water column salinities of 21 to 23.5 psu are common. Expansion of *Phragmites australis* will take place at 0.74 m^2 yr^{-1} under these low water levels, depending on seasonality (i.e. growth in spring and summer).


Transect 1

Transect 2

Transect 3

Figure 7. Predictive vegetation model for the East Kleinemonde Estuary under closed, low water levels.

Closed mouth, high water levels (> 1.5 m MSL)

Under these conditions intertidal salt marsh is completely inundated and dies. Salt marsh becomes heavily colonised by macroalgal epiphytes with species such as *Cladophora* and *Enteromorpha*. The pioneer species *Chara vulgaris* and *Halophila ovalis* become replaced by the larger more robust species of *Ruppia cirrhosa* (water salinity 0 - 30 psu) and *Potamogeton pectinatus* (water salinity 2 - 15 psu). If not displaced, *Chara* will grow as long as salinity is between 15 - 20 psu (Blindow *et al*. 2003) and *Halophila* is in the 20 - 35 psu range. Since biomass of submerged macrophytes that occupy the whole water column is dependent on depth, as water level increases, so will biomass. *Ruppia* may reach a maximum potential biomass of 2883 g dry mass m^{-2}, *Chara* 599 g dry mass m^{-2} and *Halophila* 101 g dry mass m^{-2} as recorded in this study. If the estuary is perched at the time of mouth breaching and only partial drainage occurs, existing submerged macrophyte beds will supply vegetative material for recolonisation when the water level increases. The expected minimum colonisation rate is 0.75 m^2 yr^{-1}.

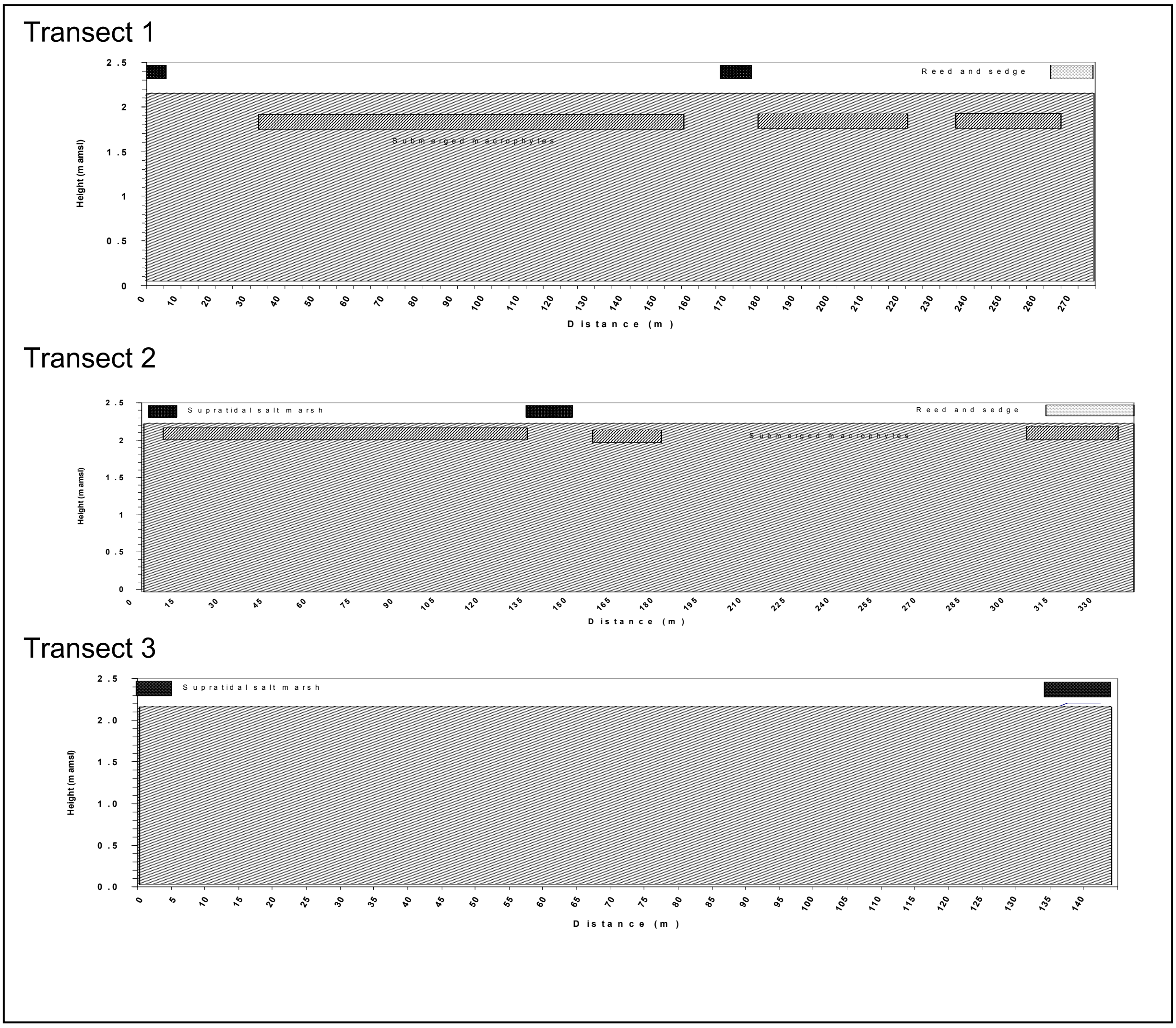


Figure 8. Predictive vegetation model for the East Kleinemonde Estuary under closed, high water levels.

CONCLUSIONS

The original project hypotheses are discussed in relation to the research findings.

Further data are needed to test the following hypotheses:

- *Salt marsh macrophytes die back when completely inundated for 120 days (4 months).*
  These conditions did not occur in 2006 and the hypothesis remains untested.
- *Large floods in TOCEs can completely remove submerged macrophytes and reeds. These recover within a year from remnant roots and rhizomes (reeds) and seeds (submerged macrophytes).*
  Intertidal conditions following the August 2006 flood resulted in exposure, desiccation and die-back of submerged macrophytes. The plants will only return if physico-chemical conditions are favourable. The analysis of the seed bank following flooding is ongoing and therefore this hypothesis stands.
- *For emergent reeds, prolonged (even partial) submergence during spring and summer is more detrimental than that during autumn and winter.*
  Mouth closure did not occur long enough to test this hypothesis.
- *Increased temperatures and nutrient input in spring leads to increased macroalgal abundance*.
  In 2006 the mouth was open in spring, the estuary was tidal and macroalgae were scarce so the hypothesis could not be tested.

Hypotheses not disproven:

- *Salt marsh macrophytes re-establish from their seed bank within 2 months when water level drops and the marsh area is exposed*.
  Intertidal salt marsh grew after 3 days exposure (a drop in water level) and increased in cover to that prior to mouth closure.
- *Submerged macrophyte biomass is highest during low flow, closed mouth conditions, when water clarity is high and physico-chemical conditions are stable.*
  Although the mouth of the East Kleinemonde Estuary did not remain closed for long enough to see if this did occur, historical data show that this is the case.
- *The submerged macrophyte* Ruppia cirrhosa *(Petagna) Grande becomes dominant after mouth closure when salinity ranges from 25 - 45 psu.* Halophila ovalis *co-exists with* Ruppia *at salinity values from 25 - 45 psu.*
  In 2006 *Ruppia* and *Halophila* occurred in the East Kleinemonde at 21 - 23.5 psu.
- *When the mouth is closed and salinity is less than 15 psu then* Potamogeton pectinatus *becomes dominant.*
  Mouth closure did not occur for long enough to test this hypothesis. However, interpretation of aerial photographs showed large beds of *Potamogeton* present in the estuary in the early 1990s when the salinity was probably fresh to brackish.
- Phragmites australis *expands its distribution during low water level conditions and responds to increased freshwater run-off from storm-water and septic tanks.*
  There have been localised increases in reeds below and above the R72 bridge possibly in response to increased freshwater runoff from residential areas.

FUTURE MONITORING AND MANAGEMENT OF VEGETATION

Cover and biomass must be measured to determine abundance and to detect change. A recommendation is that the permanent transects remain in place with monthly monitoring during conditions of rapid change. The scenario that was not captured in this study was an increase in water level during stable conditions. Monitoring of cover abundance should be done in conjunction with aerial photographic analysis on an annual basis.

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Appendix H

Specialist Report: Zoobenthos

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INTRODUCTION

Temporary open-closed estuaries (TOCE's) are the most common generic group of estuaries around the South African coast (Whitfield 1992). Typically, average size decreases from west to east (cool temperate, warm temperate to subtropical, Table 1), with TOCE's about an order of magnitude smaller than permanently open systems in each of the three biogeographical regions (Turpie 2004).

The East Kleinemonde Estuary (35.7 ha) in the warm temperate zone (Turpie 2004) is slightly smaller than the average size for TOCE's in the same biogeographical region (Table 1). Connection with the sea is infrequent (Vorwerk *et al*. 2001) and over the past 10 years (1997-2006), the estuary opened to the sea for brief intervals only (usually <10 days for any continuous period).

Opening events occur at any time during the year, but are more common during the summer rainfall months. Previous information on benthic invertebrates of the East Kleinemonde estuary is restricted to five studies; Forbes (1973) reported on the larval stages of the anomuran *Callianassa kraussi* collected from the East Kleinemonde and Swartkops estuaries, while Hill (1975) investigated the population dynamics of the portunid crab *Scylla serata*. More recently, Teske & Wooldridge (2001, 2003, 2004) reported on the subtidal macrozoobenthos. This report covers the invertebrate zoobenthic component of the East Kleinemonde Estuary project conducted during 2006 and 2007.

Table 1. Average size (ha) of temporarily open-closed and permanently open estuaries in the cool temperate, warm temperate and subtropical biogeographical regions. Data from Turpie (2004) (no information on size for 14 TOCEs).

	Intermittently open estuaries		**Permanently open estuaries**	
	Average size (ha)	n	Average Size (ha)	n
Cool Temperate	295.4	2	2158.4	2
Warm Temperate	39.9	74	177.5	29
Subtropical	13.7	94	100.5	16

MATERIALS AND METHODS

Four field trips were undertaken in 2006 (Table 2). At the time of sampling in March, the mouth had remained closed for about six months. The second field trip in July followed a period of brief mouth opening of approximately one week. Thereafter, the state of the mouth remained mostly open up to the fourth visit in November.

Table 2. Sampling dates and state of the mouth at the time of sampling on each of four trips to the East Kleinemonde Estuary.

Sampling date	State of the mouth	Comments
20 March 2006	Closed	Mouth had remained closed for at least 6 months.
27 July 2006	Closed	Mouth opened for ca 3-4 days a month previously (23 June).
12 September 2006	Open	Mouth opened 3 August. Estuary very shallow above Stn 1 (<0.50 m) at time of sampling.
14 November 2006	Open	Mouth closed between 21-27 September.

Field and laboratory sampling

Benthic community analysis

Subtidal benthic invertebrates were collected from the deck of a flat-bottomed boat using a Van Veen type grab at five sites (Figure 1).

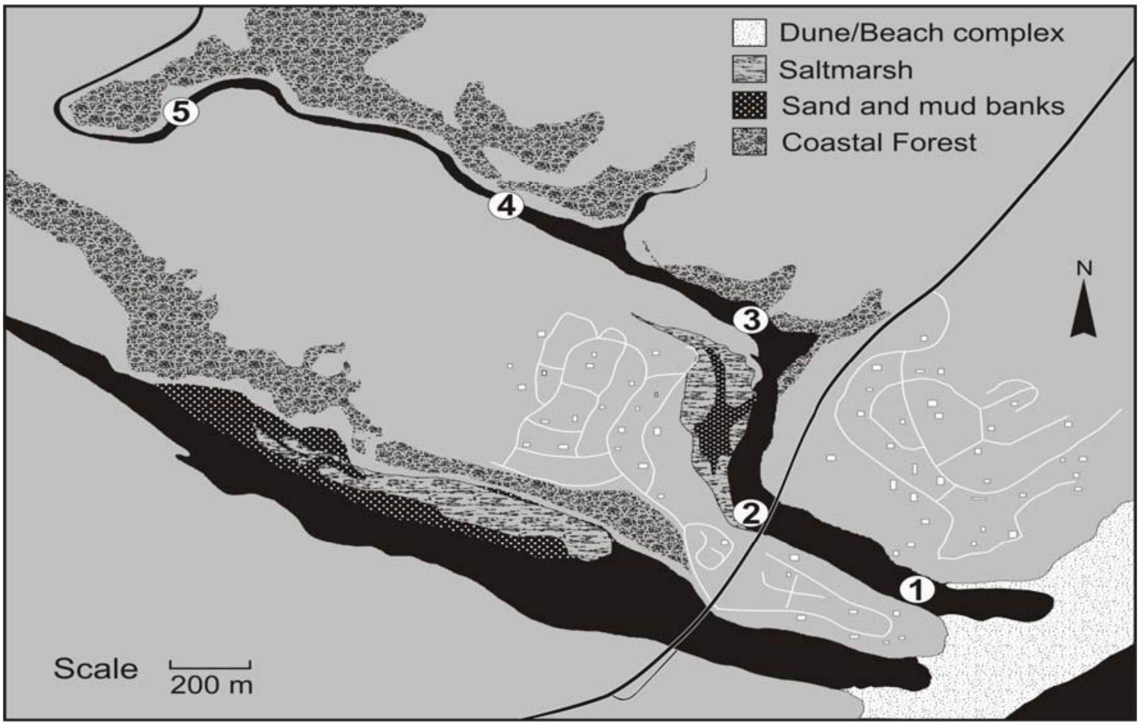

Figure 1. The temporarily open/closed East Kleinemonde Estuary showing the location of the sampling stations. The West Kleinemonde Estuary in the lower section of the map occasionally links with the East Kleinemonde as they meander across the Dune/Beach complex.

Only three sites were sampled in September due to very shallow water after the estuary had opened to the sea. The grab sampler has a 200 cm^2 area that penetrated the sediment down to about 10 cm. Nine replicates were collected over an area of about 10 m^2. Animals retained by the sieve were stored in 500 ml plastic bottles and preserved with 10% formaldehyde solution.

A YSI 6600 multi-parameter sonde was deployed at all stations (Figure 1) to measure abiotic parameters on each sampling occasion. Variables were measured near the surface, at 0.5 m and at 1 m depth (or maximum depth at any site). The following parameters were recorded at every depth sampled: Dissolved oxygen (percent saturation and in mg ℓ^{-1}), salinity (psu), temperature (^{o}C), turbidity and pH. A sediment sample collected at each station provided information on particle size distribution and percent organic content. Dry samples (dried at 60^{o}C for a minimum of 48 h and then weighed) were incinerated at 550^{o}C for 12 h to burn off the organic matter. The difference in mass of the sample after incineration provided information on organic content, expressed as a percentage. Three replicates from each sediment sample were used to obtain a final value. Samples were then soaked in distilled water for 24 h to remove salts. Excess water was carefully siphoned off and the sample again dried at 60^{o}C for 72 h. Dried sediment was then vibrated through a series of metal test sieves (2 mm, 1 mm, 500 µm, 355 µm, 250 µm, 180 µm, 125 µm, 90 µm, 63 µm and <63 µm).

Analysis of samples was completed in the laboratory. Final abundance was expressed as the average number of each species per m^2 of substratum at each site, determined from the nine replicates respectively. Invertebrates were identified to species level wherever possible and the data analysed using multivariate statistics from the statistical package, PRIMER V.6 (Plymouth Routines in Multivariate Ecological Research). If multivariate techniques were not appropriate, other packages using MS Excel or Statistica for Windows were used.

Isotope analysis

A comprehensive set of biotic samples were collected in March at Station 1 (lower estuary) and at Station 5 (upper estuary) for carbon and nitrogen isotope analysis in order to construct a food web for the benthic invertebrates. Five replicates (stored separately) of six grabs each were sampled at each station using the Van Veen type grab. Samples were transferred to a sorting facility, where animals were extracted over the next two days, while stored at 1-4°C. Sand prawns (*Callianassa kraussi*) were collected at Station 1 with a prawn pump and were subsequently frozen.

In the laboratory, unfrozen animal samples were rinsed under running distilled water and their digestive tracts removed under a dissecting microscope. The eggs of gravid females were removed, as these contain high lipid concentrations that might affect the isotope composition of the animals (Sánchez-Paz *et al*. 2006). Whole animals were then frozen, freeze-dried, homogenised using a mortar and pestle and treated with 10% HCl to remove non-dietary carbonates (Fantle *et al*. 1999). The acid was applied in a drop-by-drop fashion, until all the $CaCO_3$ had reacted with the HCl (Jacob 2005). Acid was driven off by warming the samples on a hot plate without rinsing the samples (Bunn *et al*. 1995). Samples were again freeze-dried, homogenised and kept frozen. *Callianassa kraussi* was treated in the same way as described above, but only muscle tissue was used for isotopic analysis. Tissue was then frozen, freeze-dried and homogenized.

Particulate organic matter (POM) was sampled by collecting 2-3 ℓ of water from the upper 30 cm of the water column. It was then filtered through a 50 µm sieve to remove micro-zooplankton and suspended plant debris. The residue was then filtered through a 20 µm nitex mesh and onto a pre-combusted (48 hrs at 110ºC) Whatman GF/C filter (1 µm pore size). Samples taken at Station 1 were filtered *in situ*, while those collected at Station 5 were filtered using a vacuum extractor pump. Filters were kept frozen for further treatment. Three replicates of 340 ml surface water were also collected at Stations 1 and 5 and preserved with Lugols solution (about 10%) for species identification purposes. Identification of dominant groups was made using an inverted microscope.

Microphytobenthos was collected following the method of Riera *et al*. (1996) and Whitfield & Wood (2003). The top 2 mm of intertidal sediment from areas where dense algal growth was evident was scraped off with a spatula or scalpel (100 - 400 ml sample). The sample was transferred to a shallow tray (sediment depth *ca* 1 cm), covered with a 90 µm screen and layered with beach sand (particle size range between 250 - 500 µm) to a depth of 1 - 5 mm. The beach sand was first decalcified with 10% HCl, washed in tap-water and combusted for 8 h at 550ºC. Thereafter, the sediment was kept moist with estuarine water pre-filtered through a precombusted Whatman GF/C filter. Sediment trays were then exposed overnight to fluorescent lighting to encourage microphytobenthic species to migrate from the bottom layer through to the surface. After a few hours, the surface layer of sand became distinctly green. The following morning the sand layer on the 90 µm screen was transferred to an Erlenmeyer flask containing filtered estuarine water and rigorously shaken to separate diatoms from the sediment particles. The supernatant was washed through a 50 µm mesh sieve to remove meiofauna, and subsequently filtered onto a Whatman GF/C filter using a vacuum pump. Small sub-samples (three replicates) of the supernatant were taken and preserved with 10% Lugols solution for identification purposes. Filters were wrapped in aluminium foil and stored in a freezer for later analysis.

All GF/C filters containing POM and microphytobenthos were treated with 10% HCl, but not homogenized afterwards. The acid was evaporated by heating the filters on a hot plate. The filters were then stored wrapped in foil. Nitex filters were rinsed in distilled water, and the then filtered onto GF/C filters and treated as described for GF/C filters.

Dominant macrophytes were collected from the fringing vegetation (ca. 5 - 6 species) and from the subtidal community (7 - 8 species), although some were relatively rare. Plant species sampled included C_4 plants *Pycreus* sp. (sedge) and *Pennisetum clandestinum* (kikuyu grass), and C_3 species including other sedges (*Juncus kraussi*, *Bolboschoenus maritima*), *Phragmites australis* and the salt marsh plant *Sarcocornia* spp. (two species pooled and collected from Station 5 only). Plant material was rinsed in distilled water, freeze-dried and homogenised using a mortar and pestle. Epiphytes were gently scraped off submerged vegetation with a scalpel and frozen. Prior to isotope analysis, epiphyte samples were inspected to insure that no animals were present. They were then treated with 10% HCl in a similar fashion to that of the animal material. At Station 5, five replicates of sediment organic material (SOM, particle size ≥ 0.5 mm) were collected from the grab samples. Insufficient SOM was available in sediments at Station 1. Animals were removed from SOM samples using a dissecting microscope. The SOM samples were treated in the same way as animal material.

Stable carbon and nitrogen isotope analyses were conducted in the Stable Light Isotope Unit of the Archaeology Department, University of Cape Town. Animal and plant material were analysed in a Finnigan MAT-252 isotope ratio mass spectrometer. Stable isotope data were expressed as the heavy to light isotope ratios, in the delta (δ) notation:

$$R‰ = (R_{sample}/R_{standard}) - 1 * 1000‰$$

where R is the heavy to light isotope ratio (eg. $^{15}N/^{14}N$). The standard deviation for analyses was typically 0.1 to 0.2 ‰.

RESULTS

Environmental parameters

Salinity decreased upstream, with the lowest values and the maximum range occurring when the mouth was open to the sea (Figure 2). Salinity at Station 1 ranged between 15.5 and 20.7 over the study period. Corresponding values for the uppermost site sampled over the four visits was 9.3 and 19.6. Vertical stratification was generally associated with the upper reaches, although in September 2006, shallow water at Stations 2 and 3 resulted in a well mixed water column.

Recorded water temperatures (Figure 3) peaked upstream, averaging ca 1.4°C higher when compared to Station 1. The maximum difference between upper and lower reaches occurred in winter (2.2°C). Seasonal range at Station 1 was 11.4°C and at Station 5, 10.4°C.

Fine sand (>0.125 µm - <0.355 µm) dominated the sediment at most sites (Figure 4), with the highest values recorded at Station 1. Mud (<63 µm) constituted less than 20% of the sediment at most sites on all sampling occasions, except in September after mouth opening. The pattern previously recorded in March and July was restored at the time of sampling in November. In July, Station 3 recorded a high proportion of relatively course sand (67.2%).

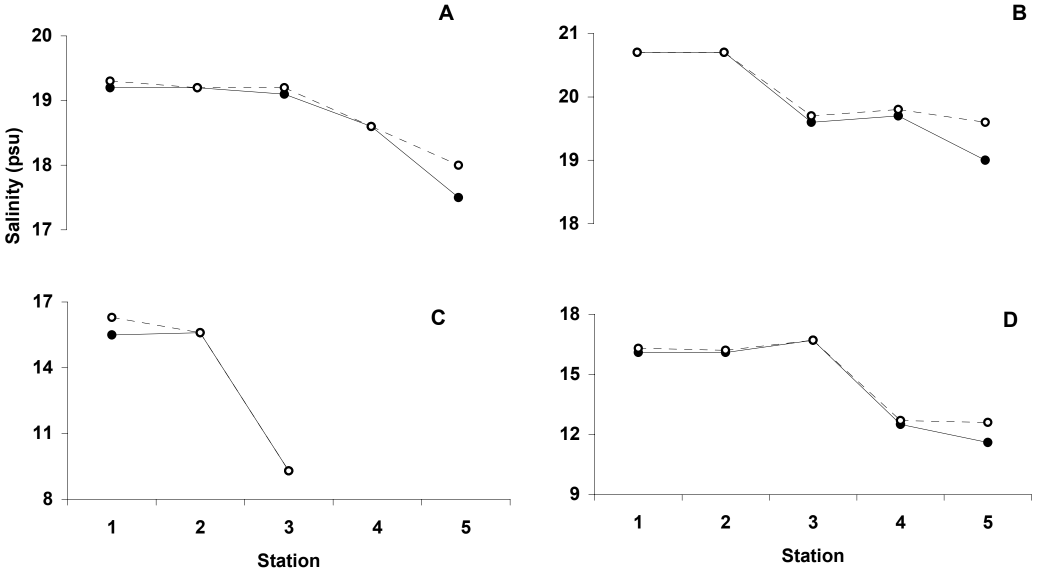

Figure 2. Salinity distribution at the time of sampling in March (A), July (B), September (C) and November (D) in the East Kleinemonde Estuary. Solid lines represent near-surface values, hatched lines represent near-bottom values.

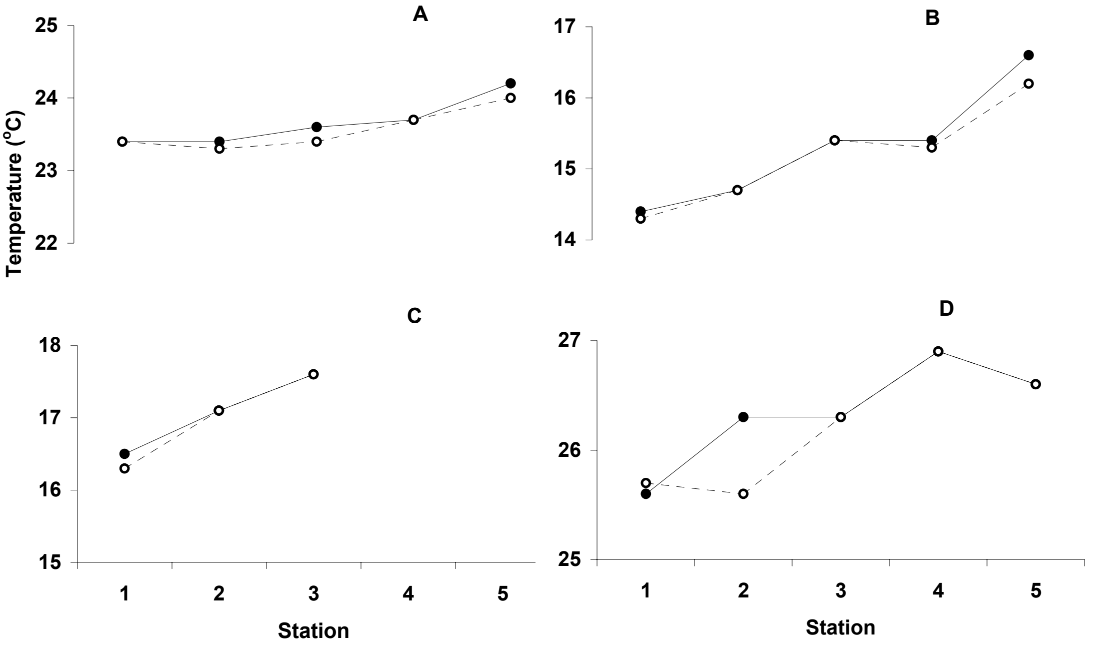

Figure 3. Temperature distribution in March (A), July (B), September (C) and November (D) 2006 in the East Kleinemonde Estuary. Solid lines represent near-surface values, hatched lines represent near-bottom values.

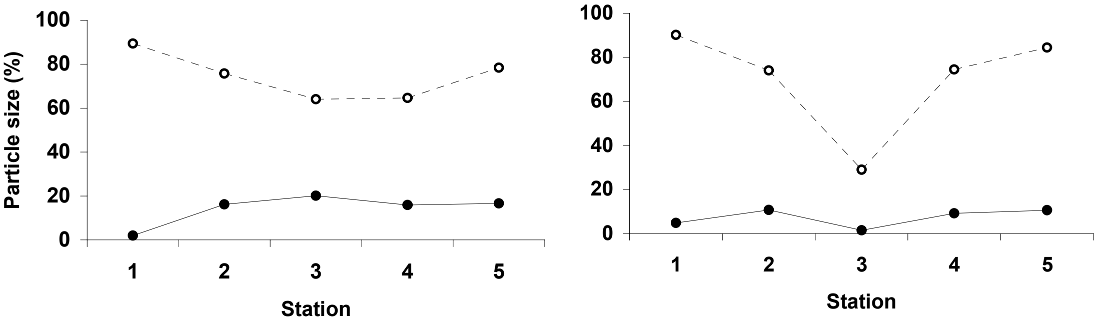

Figure 4. Particle size distribution in March (A) and July 2006 (B) in the East Kleinemonde Estuary. Solid lines represent mud content (<63 μm) and hatched lines percentage fine sand (>0.125 μm - <0.355 μm).

Biotic assemblages

At higher taxonomic levels, amphipods, tanaeids, isopods and polychaetes were the most abundant groups in the benthic community (Figures 5 and 6). Amphipods generally were dominant at all sites, making up a larger proportion of the benthic community after mouth opening. This was not as a consequence of an increase in amphipod abundance because total abundance was approximately 45 000, 13 500, 24 500 and 50 500 m^2 for each of the four sampling sessions respectively, but rather as a result of variable numbers of other groups (polychaetes, tanaeids and isopods particularly). When all groups are combined, the highest total abundance occurred in March 2006 (ca 144 000 m^2) and in November 2006 (ca 70 000 m^2). In summary, benthic invertebrate abundance was highly variable between sampling trips, with no discernable trend in the assemblage composition.

Mean abundance and spatial distribution of the seven most common zoobenthic species at the five sampling sites is shown in Figure 7. Most species were present at all sites. The amphipods *Corophium triaenonyx* and *Grandidierella lutosa* showed no distinct pattern of distribution along the estuary, while others such as the tanaeid *Apseudes digitalis* and the bivalve *Macoma litoralis* were scarce or absent at Station 1. By contrast, the cumacean *Iphinoe truncata* and *G. chelata* were most abundant at the mouth site. The mysid *Gastrosaccus brevifissura,* the amphipod *Urothoe serrulidactylus* and the bivalve *Solen cylindraceus* were only recorded below Station 3.

Group linkages shown by Bray Curtis Similarity clustering based on composition and abundance of the macrozoobenthos using group average mode on fourth-root transformed data indicated two major clusters at a similarity level of 59.7 (Figure 8). Station 1 grouped separately from all other sites for all four sampling trips. The similarity profile test (SIMPROF) indicated that the split between the groups was significant ($P < 0.05$). No significant difference was apparent between the four mouth sites.

Table 3. Correlation between the subtidal benthic species assemblage and environmental variables for each of the four sampling trips to the East Kleinemonde Estuary, 2006.

Date	Variables	Correlation	Comments
March 2006	Integrated temperature, Integrated O_2, % Fine sand and % Organic matter	99%	% mud, % fine sand and % organic matter accounted for a correlation at the 79% level.
July 2006	Integrated temperature, Water depth	87%	Water depth explained 61% of the correlation.
September 2006	Insufficient number of sampling sites		
November 2006	Surface temperature, Depth, % coarse sand	87%	% fine and coarse sand accounted for a correlation at the 58% level.

Communities did not differ in multivariate structure at other sites (Stations 2 - 5) within Group A or Group B (each sampled when the mouth was closed) and the null hypothesis of no difference between Stations 2 - 5 in March or July was therefore accepted ($P > 0.05$). A significant difference was shown between sites C2 and C3 (shortly after the mouth opened in September) and for November (Group D) when community structure at the two lower sites (Stations 2 and 3) was significantly different from the two upper sites ($P < 0.05$).

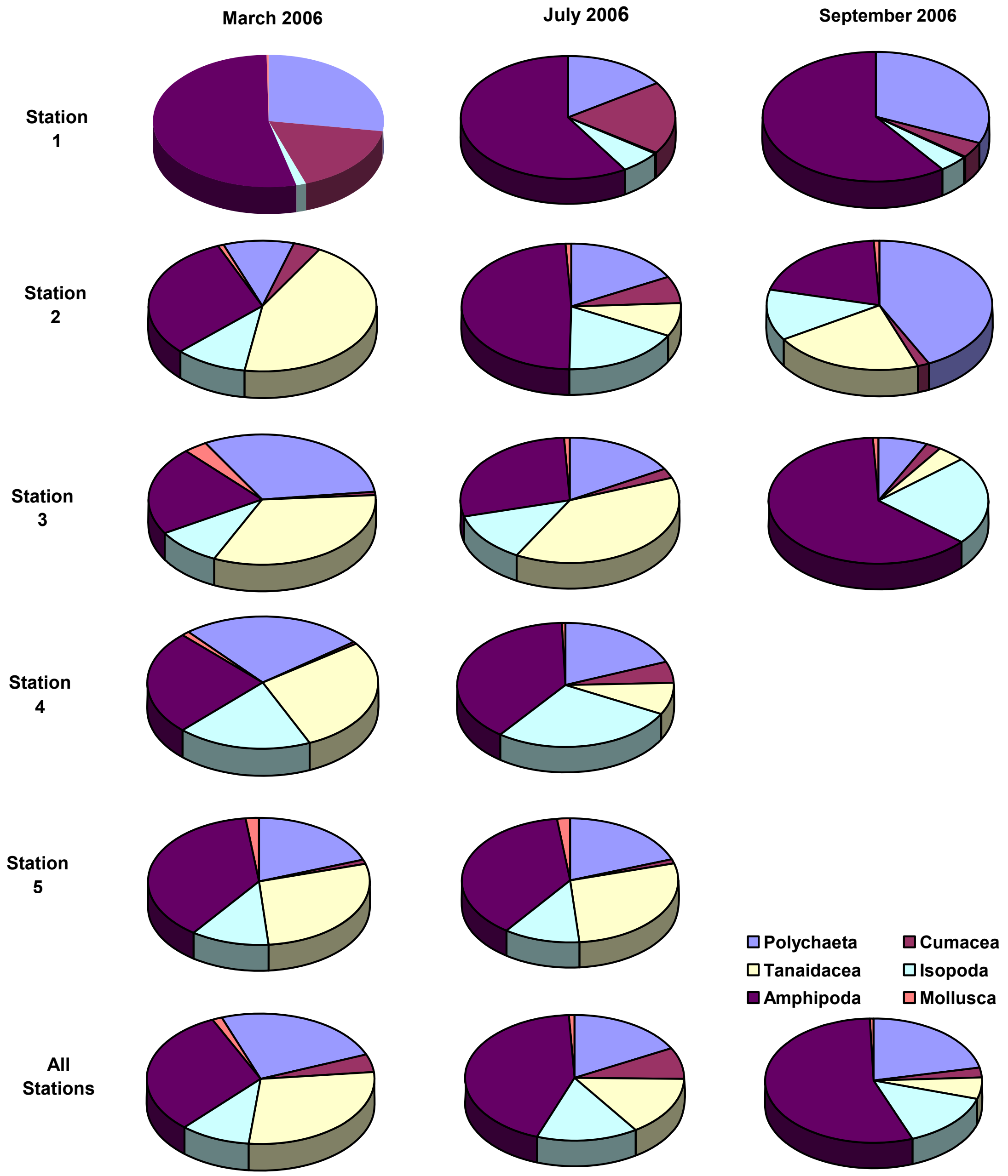

Figure 5. Proportion of the six most important taxonomic groups in the East Kleinemonde Estuary. Data for five stations and three sampling trips. Only three stations were sampled in September.

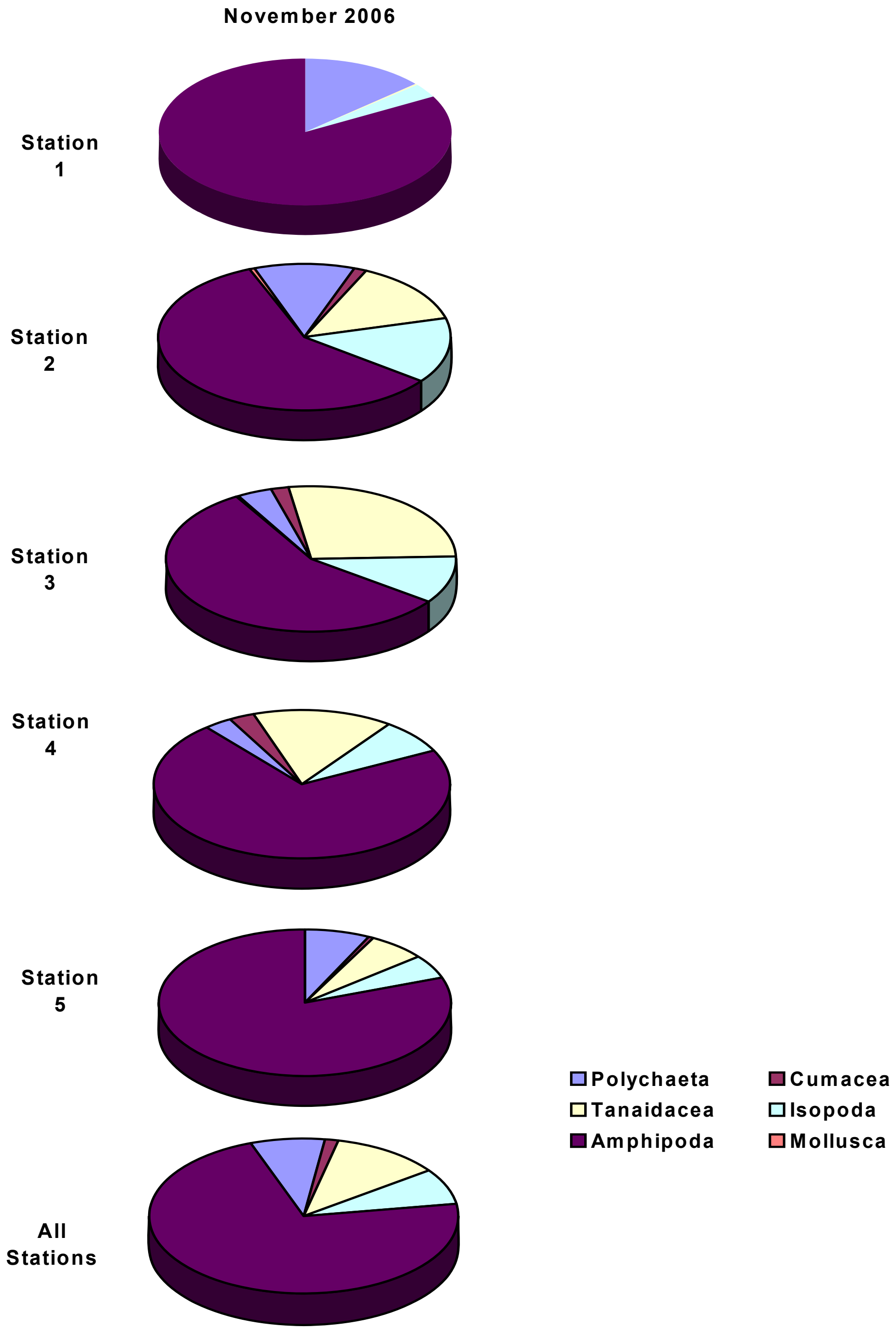

Figure 6. Proportion of the six most important taxonomic groups in the East Kleinemonde Estuary. Data for five stations on the November 2006 sampling trip.

Non-metric multi-dimensional scaling (MDS) of all sites sampled over four sampling sessions (Figure 9) reflects the same pattern shown in Figure 8, with Sites 2 - 5 sampled in March, July and September grouping together at the 70% level. All mouth sites also grouped at the 70% level. The fourth sampling session grouped separately (Group D, November 2006) reflecting small changes in community composition (sites are 67.5% similar to all other sites sampled, Figure 8) after the mouth remained open for approximately 2 months.

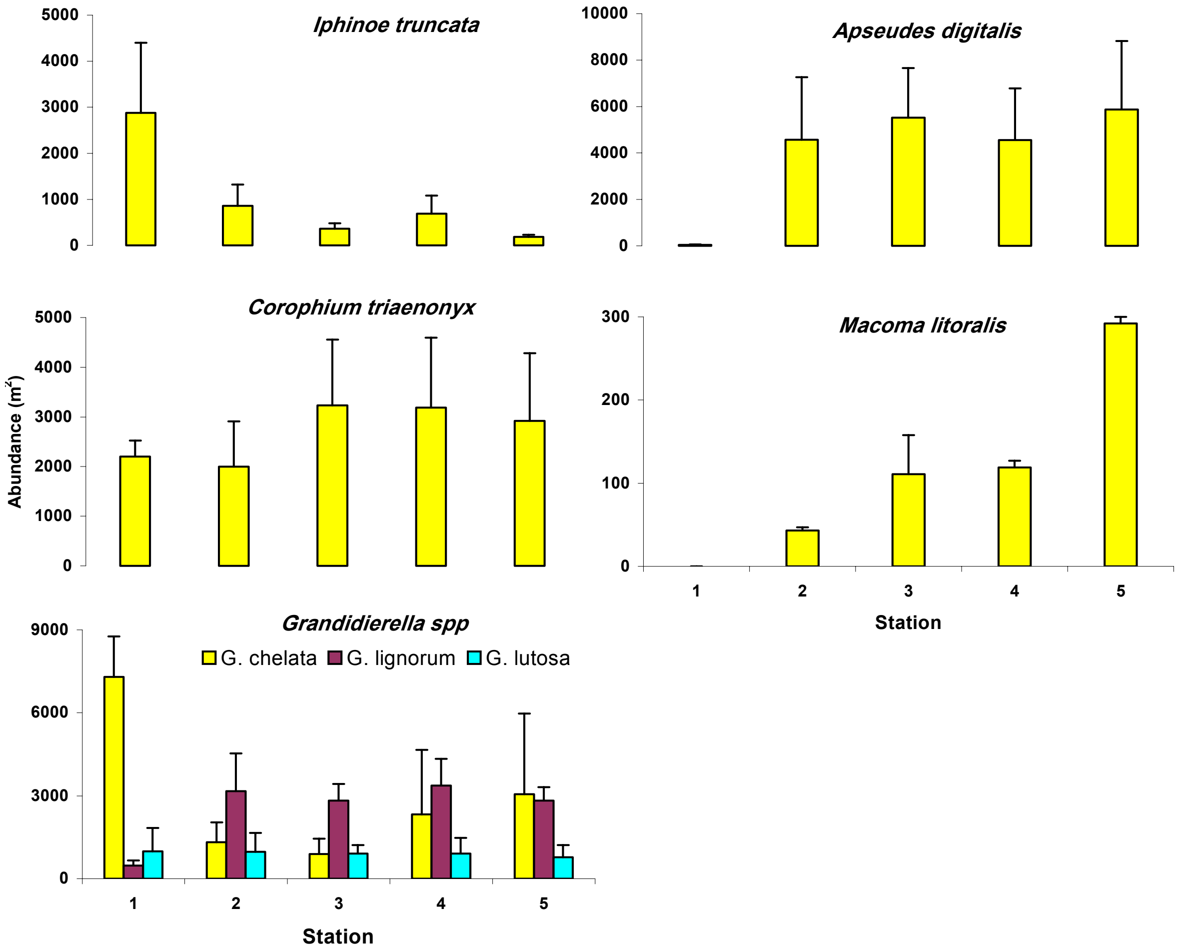

Figure 7. Mean abundance (numbers m^{-2} +1SE) of seven of the most common zoobenthic species in the East Kleinemonde Estuary. Data are for four sampling sessions between April and November 2006.

Environmental variables were then matched to subtidal benthic assemblages for each trip using the BIO-ENV package to best explain biotic patterns. A Draftsman Plot was first used to identify environmental variables that were highly correlated. For example, integrated salinity, surface salinity and bottom salinity were usually highly correlated leading to the exclusion of surface and bottom salinity readings from further analysis. A similar process of potential elimination was undertaken for other variables where multilevel readings were taken. Although up to four variables explained much of the correlation between environmental parameters measured and the subtidal benthic community composition and structure along the estuary (Table 3), sediment characteristics (including percentage organic matter) emerged as the most important on two of three occasions. In September, too few sampling sites were matched against the environmental variables to determine any correlation.

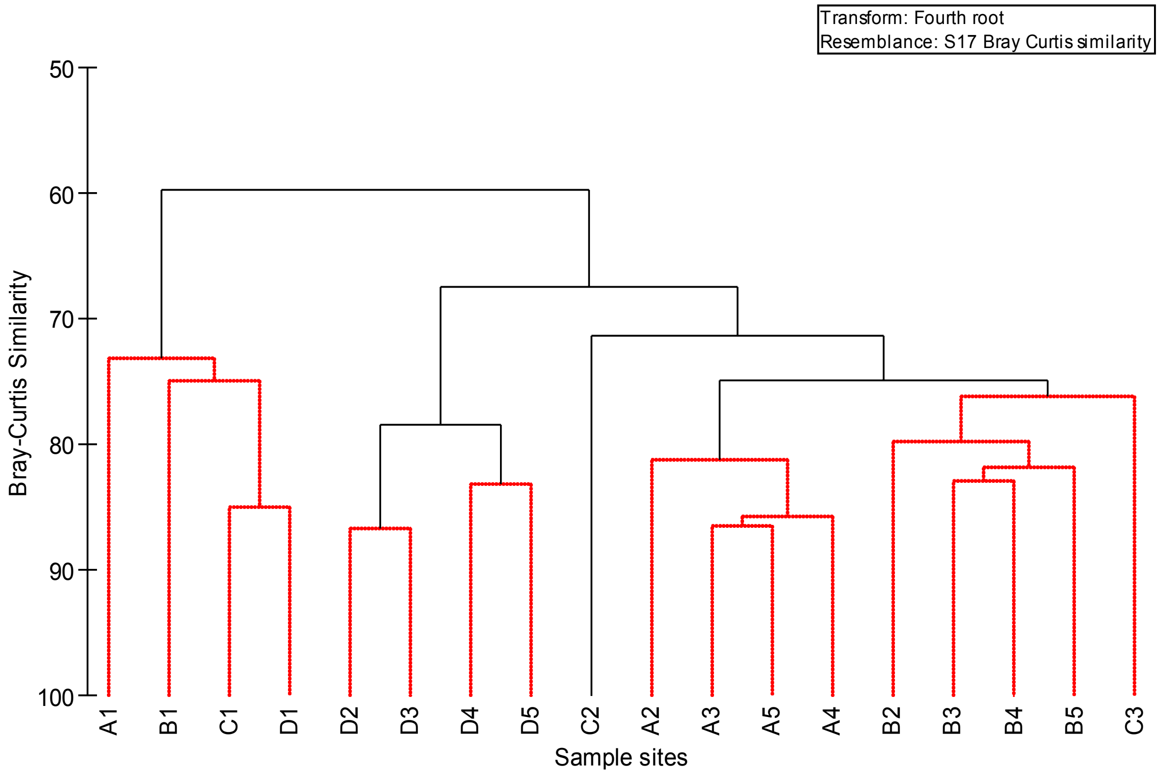

Figure 8. Bray-Curtis Similarity dendrogram based on macrobenthic composition and abundance at each sampling site in the East Kleinemonde Estuary. Data represent four sampling trips (A = March, B = July, C = September, D = November 2006). Red hatched lines indicate those sites that do not vary from each other in multivariate structure (P > 0.05).

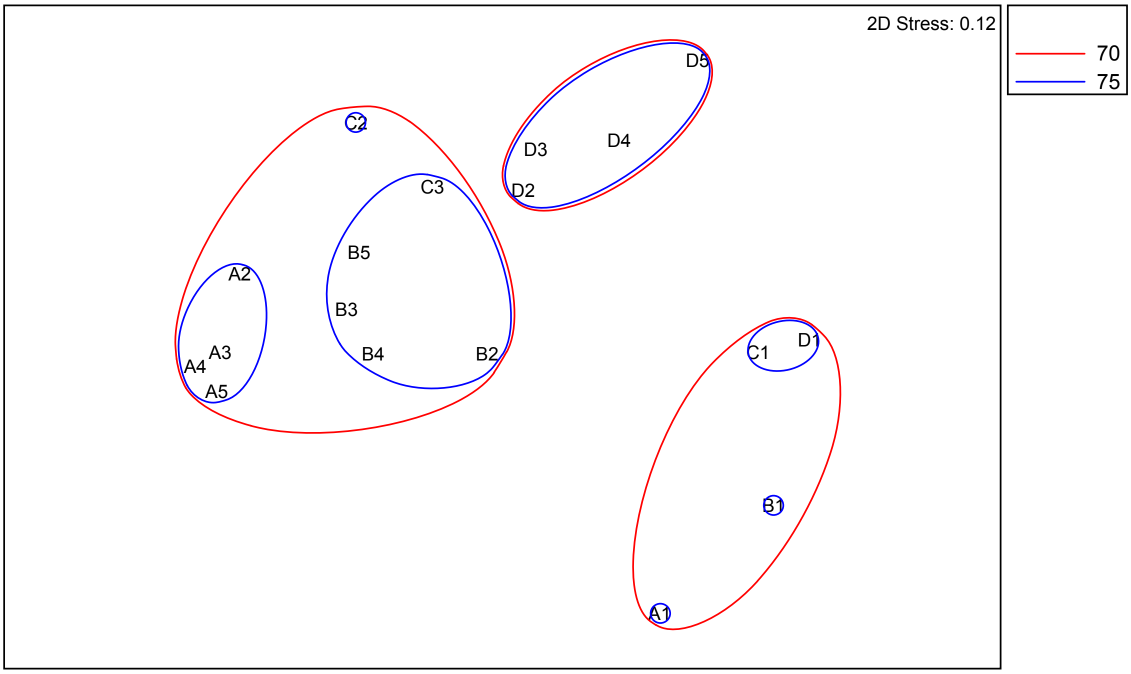

Figure 9. MDS plot based on macrobenthic composition and abundance at each sampling site in the East Kleinemonde Estuary. Data represent four sampling trips (A = March, B = July, C = September, D = November 2006). Red lines group sites that are 70% similar in community structure; blue lines group sites that are 75% similar.

Trophic linkages

Qualitative assessments of POM revealed that diatoms (particularly *Melosira* sp. and *Navicula* sp.) and flagellates were dominant at Station 1, while flagellates were the most important group at Station 5. At the latter station, diatoms (dominated by *Skeletonema* sp.) were of secondary importance. Flagellates were typically small, falling in the 1 - 20 µm size class, while diatoms included both small (<20 µm) and large (>20 µm) species. Dinoflagellates, ciliates, euglenoids and amorphous detritus representing unidentified aggregates of matter presumably organic (origin uncertain), were observed at both stations.

Significant differences between ^{15}N:^{14}N and C:N ratios of primary producers were recorded between Stations 1 and 5 (Figures 10 - 11, Table 4), with higher mean δ ^{15}N values and lower mean C:N ratios at Station 1. This may be due to ^{15}N-rich groundwater or human waste water inputs, which tend to have ^{15}N:^{14}N ratios ranging from 2 - 8 ‰ and 10 - 20 ‰ respectively (McClelland, Valiela & Michener 1997). Fertilizer input from residential gardens on the steep northern bank opposite Station 1 could also contribute nitrogen to the system. These inputs appear to be localized, as enrichment effects were not observed for all producer species sampled at Station 1.

POM <20 µm collected from Station 1 was enriched in δ ^{15}N (8.09 ‰ ± 1.26) and depleted in δ ^{13}C (-28.47 ‰ ± 0.89) compared to Station 5 (6.75 ‰ ± 0.74 and -27.81 ‰ ± 0.70) for δ ^{15}N and δ ^{13}C respectively (Figures 10-11). POM >20 µm was depleted in both nitrogen (4.50 ‰ ± 1.63) and carbon (-26.83 ‰ ± 0.94) at Station 1, compared to the uppermost Station 5 (9.01 ‰ ± 3.82 ^{15}N and -25.94 ‰ ± 0.63 ^{13}C).

Carbon stable isotope ratios of POM differed significantly between stations and particle size groups (Tables 5 and 6). POM >20 µm was typically enriched in ^{13}C compared to POM <20 µm. Nitrogen ratios did not differ significantly between sites and particle size (Table 5). ^{15}N of POM >20 µm, however, differed significantly between stations, while the two particle size fractions at Station 1 also differed significantly (Table 6). C:N ratios showed no difference between stations or particle size groups (Table 5).

No quantitative data are available for the benthic microalgae. However, qualitative observations indicated that flagellates dominated the microphytobenthos at Station 1 (n = 5). Samples included some from areas exposed to nutrient-rich seepage. Diatoms were the more important group at Station 5 (n = 3), with *Cymbella* sp and *Surirella* sp the most important genera.

Trophic levels were determined based on δ ^{15}N enrichments proposed by McCutchan *et al.* (2003) and Minagawa & Wada (1984). Expected enrichments are given as 2.1 ± 0.21 ‰ for whole animals and 2.9 ± 0.32 ‰ for muscle tissue (McCutchan *et al.* 2003), or 3.4 ± 1.1 ‰ (Minagawa and Wada 1984). For example at Station 1 (Figure 10), the difference in δ ^{15}N of 3.39 ‰ between the carnivores (*Ceratonereis keiskama*, *Pontogeloides latipes*, *Cirolana fluviatilis* and *Cyathura estuaria*) and primary consumers represented by *Grandidierella* spp. and *Iphinoe truncata* was greater than the expected enrichment of 2.1 ‰ or 3.4 ‰. These two groups are, therefore, considered as separate trophic levels. There was no significant difference between the males of *Grandidierella chelata*, *G. lignorum* and *G. lutosa* and females of the different species, validating pooled samples for the purposes of isotope analysis (Table 7). The differences between predators (*Ceratonereis keiskama*, *Cyathura estuaria*, *Pontogeloides latipes* and *Cirolana fluviatilis*) mean δ ^{15}N signatures and *Callianassa kraussi* was 0.52 ‰ and between predators and *Urothoe serrulidactylus* the difference was 1.61 ‰. These values were smaller than the nitrogen shifts expected for separate trophic levels (McCutchan *et al.* 2003, Minagawa & Wada 1984). This is supported by the overlap in nitrogen isotope ratios observed between these groups (Figure 10). This overlap with the carnivore group is probably due to the high nitrogen signature of the microphytobenthos (MPB), a possible food source for *U. serrulidactylus*. Similarly, the presence of *C. kraussi*, a suspension feeder, at a similar ^{15}N level as the carnivores is due to the high nitrogen signals of its food source (POM).

The δ ^{13}C signatures of the primary and secondary consumers at Station 1 reflect two broad feeding groups. *Callianassa kraussi* and *Corophium triaenonyx* feed on depleted sources of carbon (particularly 1 - 20 µm size class POM), while primary food sources such as MPB, epiphytes and, to a marginal extend, vascular plants form the base of food chains that include a larger assemblage of species (*Urothoe serrulidactylus, Ceratonereis keiskama, Pontogeloides latipes, Cyathura estuaria, Iphinoe truncata, Grandidierella* spp. and *Cirolana fluviatilis*). Carbon signatures of carnivores indicate the utilisation of a wide range of prey items. Unidentifiable remains of polychaetes were also observed in the stomachs of *C. estuaria* and *C. fluviatilis*.

Carbon and nitrogen signatures of *Grandidierella* spp. and *Iphinoe truncata* overlap with those of MPB collected from the less developed southern bank of the estuary and epiphytes collected from *Phragmites australis*. The consumers are depleted in ^{15}N relative to the primary producers, which may indicate mixed feeding on more than one source, including a source not collected. Based on ^{13}C signatures however, both *Grandidierella* spp. and *I. truncata* preferentially feed on epiphytes and MPB algae not enriched in ^{15}N and therefore not in areas of high seepage (northern shore).

As in the case of Station 1, two distinct trophic levels were identified among the consumers at Station 5 (Figure 11). Predators (*Cyathura estuaria, Dendronereis arborifera, Ceratonereis keiskama* and *Cirolana fluviatilis*) reflected mean ^{15}N values between 12 and 14 ‰. *Macoma littoralis* and *Sanguinolaria capensis* had δ ^{15}N signatures similar to that of the predators. A second trophic level (*Grandidierella spp* and *Apseudes digitalis*), *Corophium triaenonyx* and *Solen cylindraceus* was identified since δ ^{15}N values reflected a difference >3.4 ‰ when compared to the predators (Minagawa & Wada 1984, McCutchen *et al.* 2003).

Mean δ ^{13}C signatures for predators were more spread when compared to the same group at Station 1, ranging between -22 and -26 ‰ (-21.5 and -24 ‰ at Station 1). *Corophium triaenonyx* and *Solen cylindraceus* at Station 5 were more depleted in ^{13}C compared to the group at Station 1, even though POM (1 - 20 µm) signatures were slightly enriched in the former (Figures 10 and 11). *Grandidierella* spp. and *Apseudes digitalis* had carbon signatures that overlapped those of epiphytes, *Phragmites australis* and C_3 sedges, suggesting that these primary producers were the main food source. However, selective feeding on a particular source is also a strong possibility.

Table 4. Spatial variability (Mann-Whitney rank sum test) of isotope signatures of primary producers between Stations 1 and 5 (n.s. = no significant difference).

Isotope	**T**	**P-level**
^{15}N	1271.50	<0.00001
^{13}C	1208.00	n.s.
C/N	739.50	= 0.01

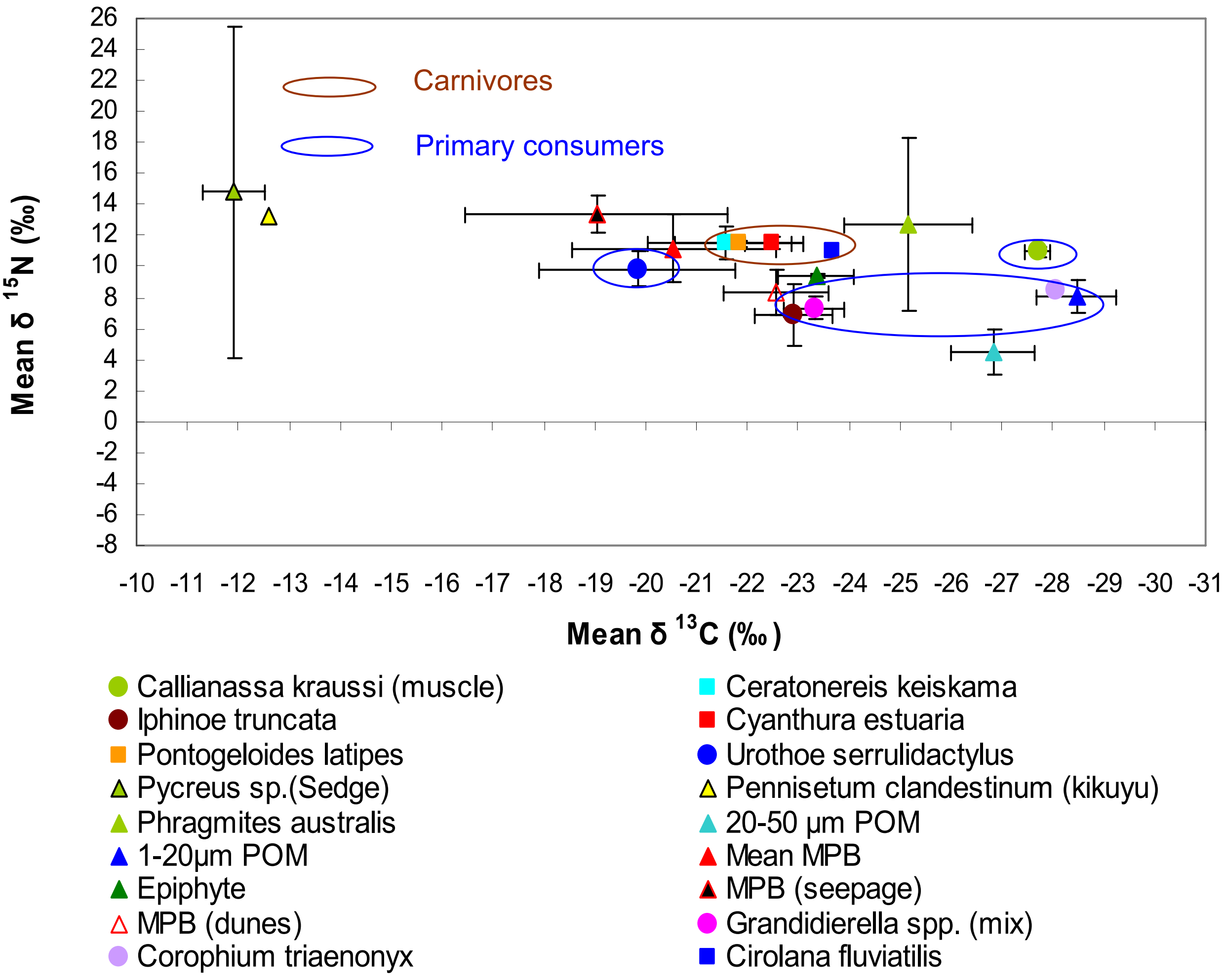

Figure 10. Mean δ ^{15}N and δ ^{13}C signatures (± 1 SD) of primary producers and consumers at Station 1. Triangles indicate primary producers, circles primary consumers and squares the secondary consumers.

Table 5. 2-Way analysis of variance comparing isotope signatures of POM, for particle size groups (1 - 20 µm and 20 - 60 µm) and station (n.s. indicates no significant difference).

Isotope	Factor	F	d.f.	P-level
^{15}N	Particle size (PS)	0.46	1	n.s.
	Station	2.59	1	n.s.
	PS vs Station	8.82	1	< 0.01
^{13}C	Particle size (PS)	24.07	1	< 0.001
	Station	4.75	1	< 0.05
	PS vs Station	0.11	1	n.s.
C:N	Particle size (PS)	2.64	1	n.s.
	Station	0.16	1	n.s.
	PS vs Station	1.92	1	n.s.

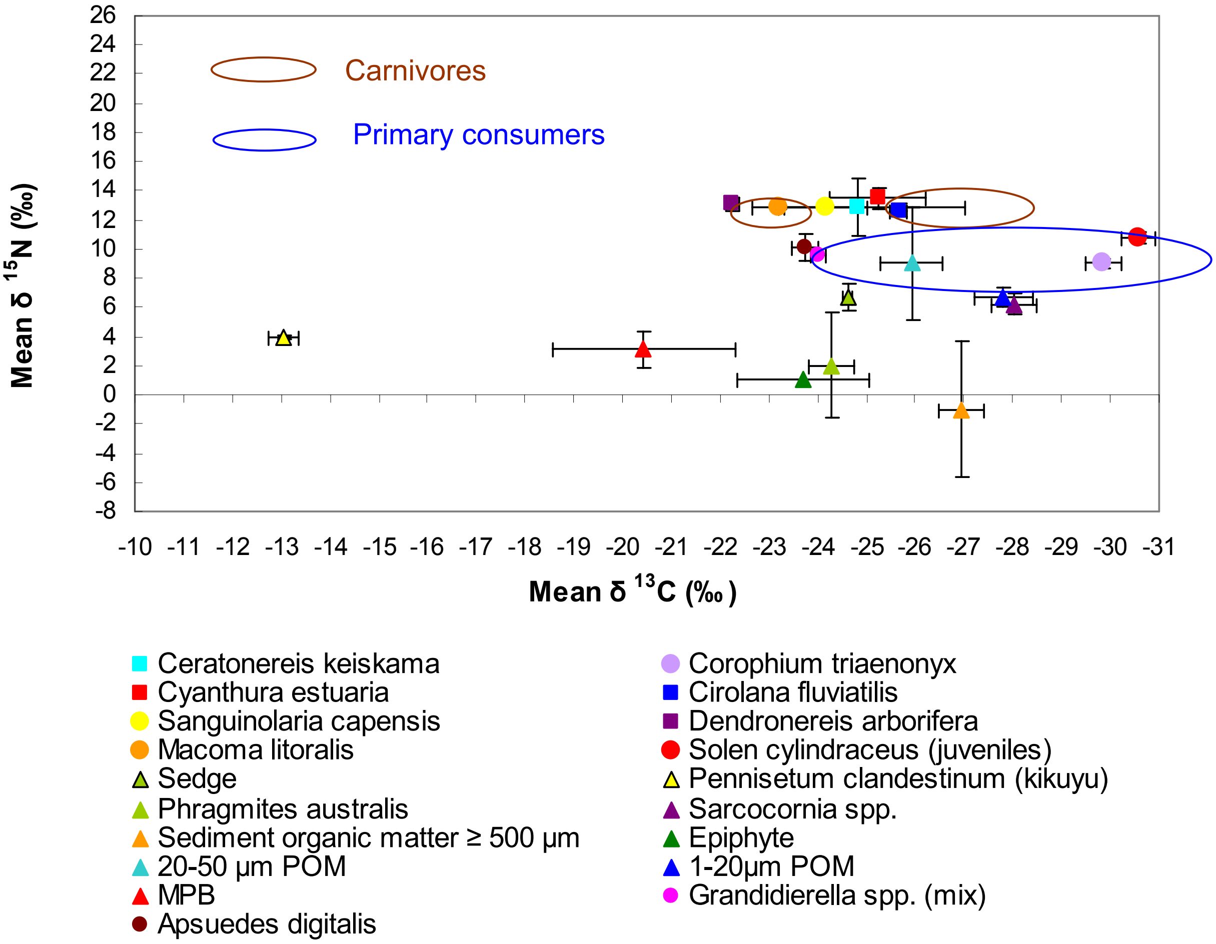

Figure 11. Mean δ ^{15}N and ^{13}C signatures (± 1 SD) of primary producers and consumers (Station 5). Triangles indicate primary producers, circles primary consumers and squares secondary consumers.

Table 6. Tukey *post hoc* tests for the comparison of POM isotope signatures for particle sizes and stations.

Isotope	**Factors**	**Within-factor comparison**	**q**	**P-level**
^{15}N	Particle size vs Station	Stations within 20 - 50 µm POM	4.58	< 0.01
		POM within Station 1	3.65	< 0.05
^{13}C	Particle size		6.94	< 0.001
	Station		3.08	< 0.05

Table 7. Comparison (1-Way ANOVA and Kruskal-Wallis ANOVA on Ranks if not normally distributed) between isotope signatures of male *Grandidierella chelata*, *G. lignorum, G. lutosa* and females of the three species (n.s. = no significant difference).

Isotope	Statistic	D.f.	P-level
^{15}N	F = 0.726987	2	n.s.
^{13}C	H = 3.65976	3	n.s.

DISCUSSION

Typically, temporarily open/closed estuaries support fewer species compared to permanently open estuaries experiencing strong marine influence (Table 8). Strong freshwater influence or river dominance also leads to relatively low species richness. In the present study, 31 macrobenthic species were identified, supporting the model recorded by Teske & Wooldridge (2001). These values on the number of species in the East Kleinemonde Estuary fall in the upper quartile for TOCEs in the region (Table 8). Although fewer species are usually present in TOCE's, the density of species can be higher compared to permanently open systems (Teske & Wooldridge 2001).

Temporal and spatial variation in the horizontal salinity gradient, mouth condition and sediment type all influence estuarine benthic community composition and structure. Salinity variation and sediment characteristics are considered prime external regulatory factors (e.g. Carriker 1967, Wolff 1983, de Villiers *et al*. 1999, McLusky & Elliott 2004), although their relative influence is variable over time and space. In local estuaries salinity is more important as a regulatory factor at the extremes of the horizontal salinity gradient. The degree of marine dominance affects the species assemblage in the lower estuary, while at the head freshwater inflow influences the assemblage in the low salinity zone at the head (Teske & Wooldridge 2003). Both these community types were absent from the East Kleinemonde Estuary during the present study, probably due to the persistence of the closed mouth and low freshwater inflow that lead to euryhaline conditions throughout the estuary.

In the present study, variation in salinity along the East Kleinemonde Estuary did not exceed two during the closed phase (March and July). The recorded range is not unexpected, since periods of estuary mouth closure (months) usually lead to a horizontal salinity gradient that becomes less marked (Day 1981a). After breeching in August, the water level dropped significantly and the salinity difference between Stations 1 and 3 was 6.2, a month after the breeching event. At the time of sampling in September, water depth precluded any sampling further upstream. The mouth again closed ca. three weeks prior to sampling in November, leading to an increase in water depth. On the last sampling trip, the salinity range between the mouth and the upper station was 5.1 psu.

Sediment type is also a key factor that structures estuarine benthic communities (e.g. Boesch 1973, Day 1981b, Bachelet *et al*. 1996, McLusky & Elliott 2004). In a study of 13 South African estuaries having widely different abiotic attributes (state of the mouth, salinity distribution, etc), Teske & Wooldridge (2003, 2004) concluded that composition of the euryhaline subtidal benthic assemblage was mainly influenced by the nature of the substratum. Two groups were distinguished; an estuarine sand fauna and an estuarine mud fauna. Sediment characteristics also include other parameters such as organic content.

Table 8. Number of species recorded in the subtidal benthos in permanently open and temporarily open/closed systems, using a 500 µm mesh sieve. Examples include estuaries from all three biogeographical regions around the coast of South Africa. A description of the degree of marine or freshwater dominance in permanently open systems, based on salinity values, is provided in the references. No sub-categories are shown for TOCE's, as salinity is variable and linked to the state of the mouth.

Estuary:	**Number of species**	**References**
Permanently open estuaries		
Strong marine influence		
Mngazana	61	Thwala (2004)
Kariega	48	Teske & Wooldridge (2001)
Kromme	48	Teske & Wooldridge (2001)
Swartkops	42	Teske & Wooldridge (2001)
Strong freshwater influence		
Great Berg	32	Wooldridge (unpublished data)
Great Fish	22	Teske & Wooldridge (2001)
Keiskamma	23	Teske & Wooldridge (2001)
Olifants	23	Wooldridge (unpublished data)
Sundays	23	Teske & Wooldridge (2001)
Temporarily open estuaries		
East Kleinemonde	30	Teske & Wooldridge (2001)
Gqutywa	29	Teske & Wooldridge (2001)
Kabeljous	28	Teske & Wooldridge (2001)
Mngazi	29	Wooldridge (unpublished data)
Mpekweni	29	Teske & Wooldridge (2001)
Mtati	24	Teske & Wooldridge (2001)
Old Womans	21	Teske & Wooldridge (2001)
Van Stadens	24	Teske & Wooldridge (2001)

In the present study, true estuarine or euryhaline species characterised the benthic assemblage at all stations. Within this assemblage, similarity analysis showed a significant difference between Station 1 and all other sites for all sampling trips. The composition and structure of the community at Station 1 also remained relatively stable, with no significant statistical evidence for any substructures between the four visits. These results reflect community resilience at Station 1, despite changes in the state of the mouth or salinity shifts over the sampling period. Biotic variability increased at sites above the mouth station, particularly during the open phase. In March and July, the community was relatively homogeneous at Stations 2 - 5 with no significant differences between them. After the mouth had remained open for about one month, the community developed very different sub-structures in September at the two upper sites sampled. This was in response to the environmental changes brought about by the draining of the estuary after mouth opening. The estuary mouth remained mostly open for another month and data suggest that a distinct upper estuarine community (Stations 4 - 5) had developed relative to the community at Stations 2 and 3. This was due to a relatively strong inflow of freshwater that also helped maintain the open mouth condition over the two months.

Euryhaline conditions and a weak horizontal salinity gradient persisted during the study period in the East Kleinemonde Estuary. No marine associated assemblage developed, even during the open mouth phase. Instead, the community was composed of mostly estuarine endemic species. These estuarine resident communities have low species richness, high resilience to environmental fluctuations and are structured mainly by sediment characteristics

(Teske & Wooldridge 2001, 2003). A sand associated fauna characterized the assemblage at Station 1 in the East Kleinemonde estuary, while further upstream, a fauna typical of muddy substrata persisted. Low species richness in the estuary is also influenced by specific life history requirements. The mudprawn *Upogebia africana* for example, is probably excluded from the benthic community since it requires an open tidal inlet in order to complete its life cycle (Wooldridge 1999). Other examples include numerous crab species (Hill 1975, Pereyra Lago 1993, Papadopoulos *et al*. 2002).

Although the macrobenthic fauna in the East Kleinemonde Estuary demonstrated spatial and temporal shifts in community structure over the study period, an underlying basic pattern persisted. This would suggest that estuarine types (Whitfield 1992) support assemblages relatively unique to an estuarine type, a concept already described by Teske & Wooldridge (2001) for the macrobenthos. In their study, data were collected over three field trips from 13 estuaries (Teske & Wooldridge 2001).

Characteristic estuarine macrobenthic assemblages described by Teske & Wooldridge (2001) are shown in Figure 12. Hierarchical clustering based on presence/absence of species provided distinct estuarine groups linked to the supply of freshwater and mouth condition. Although the pattern is not precise, broad estuarine groups were identified:

- POEs having a persistent and full horizontal salinity gradient (Great Fish, Sundays and the Keiskamma estuaries – the latter also subject to frequent floods and a high silt input).
- Marine dominated POEs with a weak or reverse horizontal salinity gradient (Kariega, Kromme and Swartkops estuaries).
- Two groups of TOCEs (Kabeljous, Van Stadens and a larger group consisting of the East Kleinemonde, Gqutywa, Mpekweni, Mtati and the Old Womans).

Although two groups were identified within the TOCE group, factors that separated the two groups remain unclear.

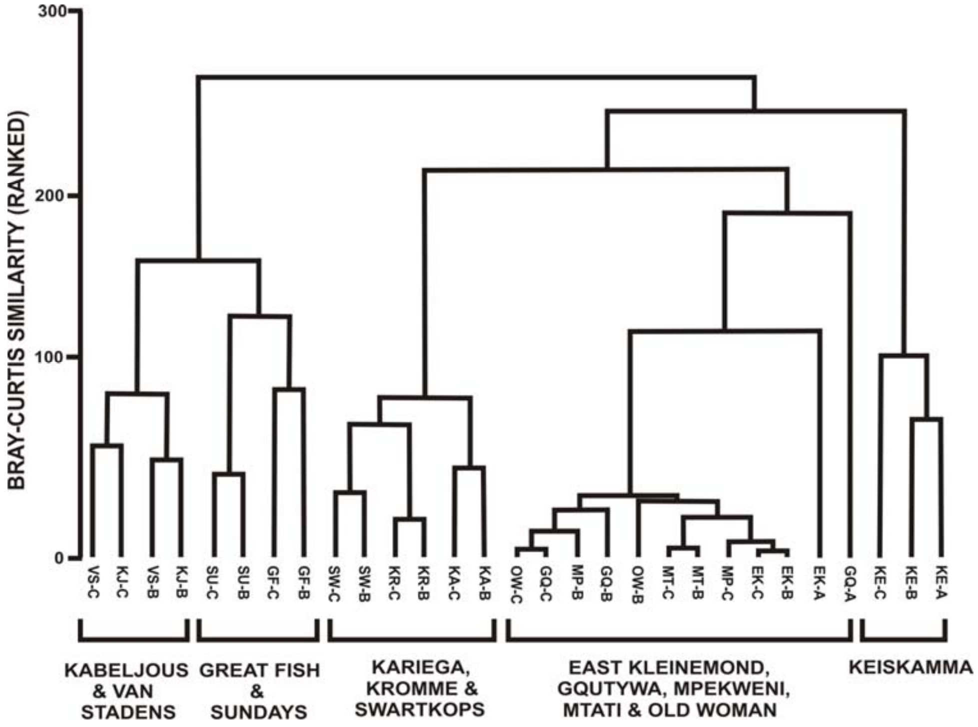

Figure 12. Hierarchial clustering of benthic community structure in 13 estuaries in the warm temperate region. Data based on three field trips, February 1998 (A), June 1998 (B) and December 1998/January 1999 (C). Modified from Teske & Wooldridge (2001).

Although two distinct feeding lineages were identified at both the lower and upper site in the East Kleinemonde Estuary, POM did not appear to represent a major source of carbon for the macrobenthic community. These ^{13}C signatures were depleted relative to typical marine POM signatures and some estuarine systems (Gearing *et al*. 1984, Kibirige *et al.* 2002, Mbande *et al*. 2004), but were similar to values recorded for the Sundays Estuary (Jerling & Wooldridge 1995). The depleted ^{13}C:^{12}C ratios found for POM could be due to the recycling of respired CO_2, which is typically depleted in ^{13}C (Smith *et al*. 1976). This could be expected under closed mouth conditions and/or low freshwater inflow. Other possible reasons include high lipid content (Fry & Wainright 1991), low growth rate (Fry & Wainright 1991) or low available inorganic nutrients in the water column (Simenstad & Wissmar 1985, Bouillon *et al*. 2004).

The amphipod *Corophium triaenonyx*, the sandprawn *Callianasa kraussi* and pencil bait (*Solen cylindraceus*) were the only species identified feeding on POM in the present study. *C. triaenonyx* was recorded at both study sites and was shown to have a more depleted level of ^{13}C in the upper estuary compared to the mouth station, even though ^{13}C of POM <20 µm was slightly but significantly more enriched at station 5. These three species may selectively filter ^{13}C-depleted organic matter from the water column, either based on particle size or organic content. Selective feeding by filter feeders has been demonstrated by authors such as Bouillon *et al*. (2004). The carbon stable isotope ratios for POM differed significantly between stations and particle sizes, although this difference was generally small. This trend was also reported for other systems (Froneman 2002) and may be the result of greater differential enrichment of ^{13}C for various phytoplankton species. In the present study, diatoms dominated at the lower study site and small flagellates at the upper site. Diatoms were shown to be more ^{13}C-enriched relative to flagellates (Gearing *et al*. 1984 and Fry & Wainright 1991), but the opposite was shown in the present study and may be due to other components affecting the signature. Epiphytes and microbenthic algae were the most important primary sources of carbon for the macrozoobenthos in the East Kleinemonde Estuary, supporting the pattern shown for estuarine fauna by other studies (Froneman 2002, Kibirige *et al*. 2002).

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Appendix I

Specialist Report: Zooplankton and Hyperbenthos

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INTRODUCTION

A number of studies have described the zooplankton community structure within temporarily open/closed estuaries (TOCEs) in both the subtropical and warm temperate regions along the South African coastline (Froneman 2002a, 2002b, Walker *et al*. 2001, Perissinotto *et al*. 2000, Kibirige & Perissinotto 2003a, 2003b, Kibirige *et al*. 2006). Results of these investigations suggest that the zooplankton biomass within these systems may attain levels equivalent to those found in the more productive permanently open estuaries (POEs) within the same geographic region (Perissinotto *et al*. 2000). Among the zooplankton, copepods of the genera, *Pseudodiaptomus* and *Acartia* numerically and by biomass dominate the zooplankton community (Perissinotto *et al*. 2000, Froneman 2002a, 2004, Kibirige & Perissinotto 2003b). Locally, mysids may also contribute to the total zooplankton abundance and biomass (Kibirige & Perissinotto 2003a). Temporal changes in the zooplankton community structure and biomass in TOCEs have been linked to a variety of factors including mouth condition, freshwater inflow and water temperature (Perissinotto *et al*. 2000, Froneman 2002a, 2004, Kibirige & Perissinotto 2003b).

The caridean shrimp, *Palaemon peringueyi,* has been identified as the dominant component of the hyperbenthos in both permanently open and temporarily open/closed southern African estuaries (Emmerson 1986, de Villiers *et al*. 1999, Bernard & Froneman 2005). A recent investigation in the temporarily open/closed Grants Estuary on the east coast of southern African indicated that recruitment of juvenile *P. peringueyi* into the estuary occurred mainly during breaching events and that the overwash of marine water into the estuary played only a minor role (Froneman 2006). The study further demonstrated that the abundance and biomass of the shrimp in the temporarily open/closed estuary was lower than that recorded in permanently open systems. The reduced values recorded in the former system could be ascribed to reduced recruitment opportunities and limited habitat availability, mainly submerged beds of macrophytes (Bernard & Froneman 2005).

This report focuses on the more recent work on the zooplankton (Part 1) and the caridean shrimp, *Palaemon peringueyi* (Part 2) in East Kleinemonde Estuary.

PART 1: ZOOPLANKTON

Biological sampling procedures

Zooplankton community structure and biomass at each station was determined from net tows (n = 3 for each station) that were conducted during the day (09:00 - 12:00) and night (19:00 - 21:00) using a WP-2 net (mouth size 0.78 m^2; mesh size 90 µm) towed at the surface (approximately 0.5 m depth). The net was fitted with a flow meter to determine the amount of water filtered during each tow. The volume filtered during the tows varied between 5.3 and 12.8 m^3 and towing speed varied between 1.5 and 3 knots. Upper, middle and lower reaches estuary stations were sampled to assess spatial patterns in the zooplankton.

Biological sample treatment

The samples collected were immediately fixed in 10% buffered (hexamine) formalin. Species composition within each zooplankton group was assessed in the laboratory. Total dry mass of each zooplankton group was determined after oven drying (60^{o}C for 24 h) a sub-sample (1/8 to 1/16) using a Folsum plankton splitter. Data were expressed as mg dry mass m^{-3}.

Statistical analyses

To compare the zooplankton communities during the sampling trips, hierarchical cluster analysis was used in conjunction with the Bray-Curtis Similarity index. Species abundance data were log transformed [log (x+1)] in order to reduce bias due to highly abundant species (Legendre & Legendre 1983). The similarity analysis programs, ANOSIM and SIMPER of the Plymouth Routine in Multivariate Ecological Research (PRIMER) computer package (Clarke & Warwick 1982) were used according to the procedure described by Field *et al*. (1982) to test the significance levels and sources of variance between the various zooplankton assemblages associated with the different groupings identified in the hierarchical cluster analysis.

RESULTS

Zooplankton abundances and biomass

Unfortunately due to the shallow water depth, no samples were collected from the upper reaches of the estuary over the period August to October. Total zooplankton abundance and biomass within the East Kleinemonde Estuary demonstrated a distinct temporal pattern with the daytime values significantly lower than the night-time values ($P < 0.05$ in both cases). The mean total zooplankton abundance and biomass during the daytime ranged from 958 to 4678 ind. m^{-3} and between 8.95 and 16.46 mg dry mass m^{-3}, respectively (Figures 1A and 1B). During the night-time, total zooplankton abundances varied from 3018 to 18116 individuals m^{-3} while the biomass ranged from 18.97 to 40.73 mg dry mass m^{-3} (Figures 1A and 1B). Although there were no significant temporal patterns in total zooplankton abundance and biomass during the study, there was some evidence to suggest that the breaching events were associated with a decrease in both the total zooplankton abundance and biomass (Figures 1A and 1B). Spearman rank correlation analysis revealed significant correlation between total zooplankton biomass and temperature ($R = 0.73$; $P < 0.05$).

Community composition

In the absence of any significant spatial differences in the zooplankton biomass during the study, zooplankton densities for each month were pooled. Results of the numerical analysis revealed the presence of two distinct zooplankton groupings during the study (Figure 2). The first grouping comprised those stations that were occupied during the first seven months (January to July) of the survey. The second grouping comprised those months that were occupied following the major breaching of the estuary in August. ANOSIM indicated that the two groupings were significantly different from one another. ($R = 0.667$; $P < 0.05$). The five most numerical abundant species accounting for up to 95% of the similarity within each grouping identified with the hierarchical cluster analyses is shown in Table 1. SIMPER analyses indicated that the separation of the two-zooplankton groupings could be linked to shifts in the numerically dominant zooplankton species. Within grouping 1, the total zooplankton abundance was numerically dominated by estuarine copepods *Pseudodiaptomus hessei*, *Acartia longipatella* and *Halicyclops* spp, which collectively accounted for > 86% of all zooplankton counted. Following the breaching event in August, the zooplankton community was well represented by both estuarine (*P. hessei* and *A. longipatella*) and marine breeding species including an unidentified chaetognath species of the genus *Eukrohnia,* the copepods *Calanus simillimus*, *C. agulhensis*, *Clausocalanus* sp. and *Oithona plumifera* and juvenile *Palaemon peringueyi*. Collectively the marine species contributed between 18 and 36% of all zooplankton counted.

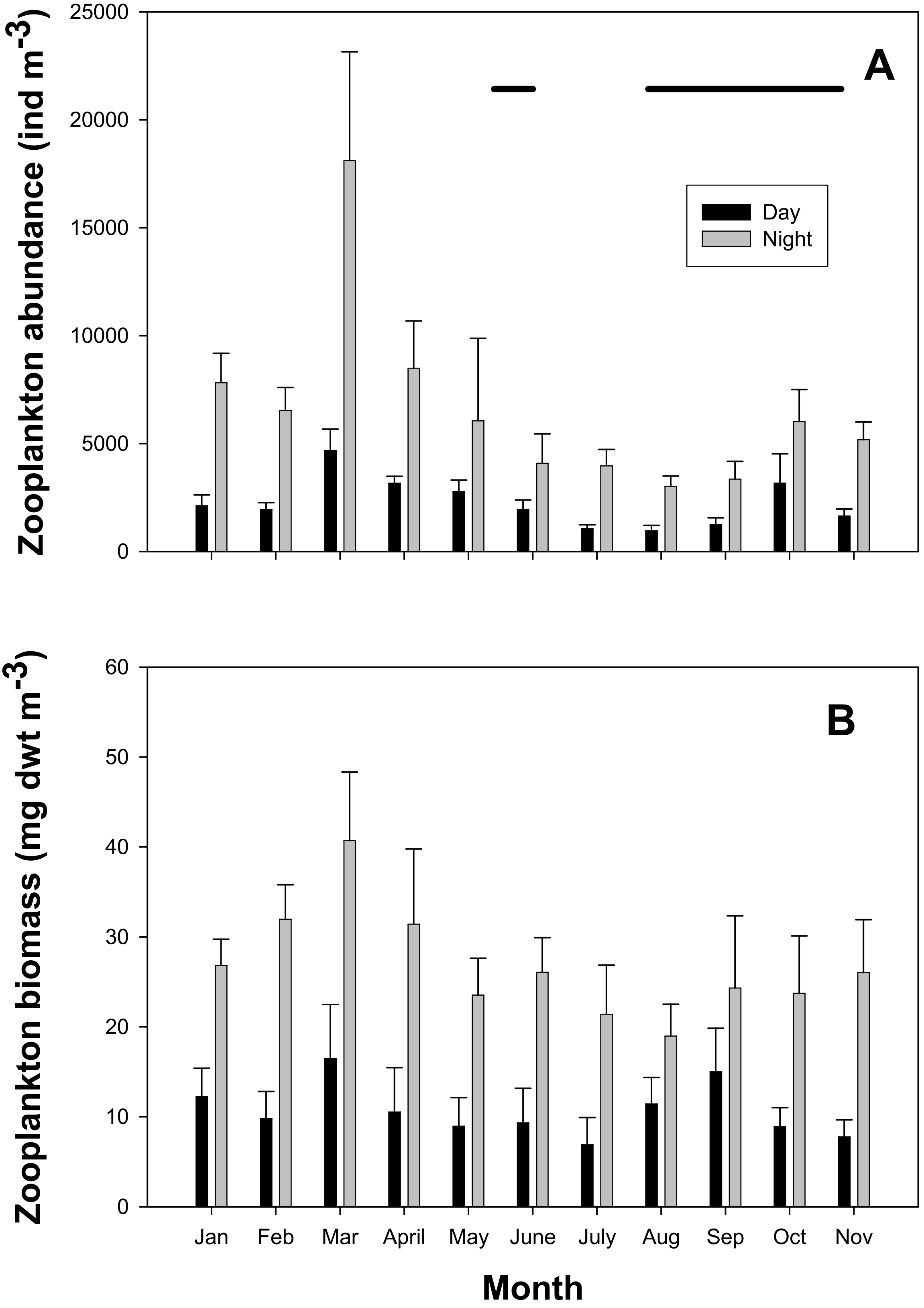

Figure 1. Total zooplankton abundance (A) and biomass (B) within the temporarily open/closed East Kleinemonde Estuary. Error bars are standard deviation. Thick horizontal line indicates when the estuary was open.

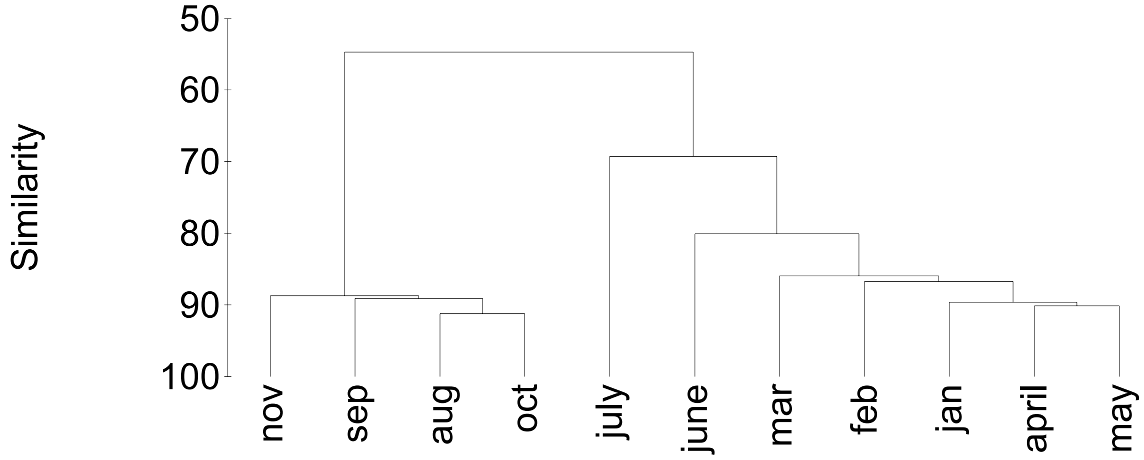

Figure 2. Results of the hierarchical cluster analyses performed on the mean zooplankton abundance data for a period of 11 months within the temporarily open/closed East Kleinemonde Estuary.

Table 1. The average abundances of the five most numerically dominant zooplankton species within each grouping identified with the hierarchical cluster analysis depicted in Figure 2. Collectively, the five zooplankton species contributed up to 95% of the similarity within each grouping. The results are from the SIMPER routine in PRIMER.

Group 1 (before major mouth breaching)		**Group 2 (after major mouth breaching)**	
Species	**Average abundance (individuals m^{-3})**	**Species**	**Average abundance (individuals m^{-3})**
Pseudodiaptomus hessei	4316	*P. hessei*	1116
Acartia longipatella	314	*A.longipatella*	113
Nauplii	397	*Oithona plumifera*	87
Oithona nana	172	*Palaemon peringueyi*	12
Halicyclops sp.	68	*Calanus agulhensis*	8

The estimates of total zooplankton abundance and biomass during the present study are in the range reported for other TOCEs both within the same geographic region (Froneman 2002a, 2002b, 2004) and within the warm temperate region of the South African coastline (Kibirige & Perissinotto 2003, Kibirige *et al*. 2006). Breaching events in the East Kleinemonde were associated with a decline in the total zooplankton abundance and biomass within the estuary (Table 1). The observed decrease is consistent with the published literature and can be related to the export of the biomass rich estuarine waters into the marine environment (Froneman 2004).

Results of the numerical analyses indicate that mouth phase plays a critical role in determining the zooplankton species composition within the East Kleinemonde estuary (Figure 2). In the absence of any direct link to the marine environment, the zooplankton community was numerically dominated by typical estuarine copepod species of the genera, *Pseudodiaptomus*, *Acartia* and *Halicyclops* which collectively contributed > 86% of all the zooplankton counted (Table 1).

The establishment of a link to the marine environment following the breaching of the estuary in August coincided with a decrease in the estuarine copepod species and an increased contribution of marine breeding zooplankton species within the estuary (Table 1). The absence of any distinct change in the zooplankton community structure following the breaching event in June, however, suggests that the influence of these events on the zooplankton community structure demonstrates a high degree of temporal variability. The absence of any direct response can likely be related to both the magnitude of the breaching event as well as the availability of recruiters within the surface zone adjacent to the estuary (Froneman 2002*a*, Kemp & Froneman 2004).

PART 2: HYPERBENTHOS (*Palaemon peringueyi*)

Biological sampling procedures

To investigate the spatial and temporal variability of the shrimp in the estuary, samples of *Palaemon peringueyi* were collected from the lower, middle and upper reaches of the system using a dip net (mouth area 0.3 X 0.3 m) according to the method described in Bernard & Froneman (2005). Five replicate samples were collected at each station by pushing the dip net perpendicular to the shore over a distance of 1.5m. The total area sampled on each occasion was thus equivalent to 0.45 m^2.

Biological sample treatment

Samples collected were preserved in 5% buffered formalin (hexamine) solution for later analysis in the laboratory. In addition, during each sampling trip, a visual estimation of the submerged macrophyte cover at each station was made in a 1x1m quadrant. Study was conducted over the period January to November.

Statistical analyses

In the laboratory, all shrimp collected were counted, sexed (individuals >9mm) and the carapace length (CL) (defined as the region from the anterior tip of the carapace to the dorsal midline of the posterior margin of the carapace) measured and the wet weight determined. The sex of shrimps >9 mm were determined by the presence or absence of an appendix masculine on the endopod of the second pleopod (Bernard & Froneman 2005). Individuals <9mm (CL) were assumed to be juveniles (Bernard & Froneman 2005). The carapace length measurements were made using electronic Vernier callipers and the wet weights using a Sauter AR microbalance. Mean abundance and biomass data at each station were standardised and expressed as ind m^{-2} or mg wet mass m^{-2}, respectively.

RESULTS

Abundance and biomass of *P. peringueyi*

Mean total abundance and biomass of the caridean shrimp, *Palaemon peringueyi*, during the study ranged from 0 to 9.1 individuals m^{-2} and from 0 to 8.8 mg wet mass m^{-2}, respectively. A distinct spatial pattern in the total abundance and biomass of *P. peringueyi* was evident with highest values generally recorded in the middle reaches of the estuary while the lowest values were consistently recorded in the upper reaches of the system (Figure 3A and 3B). Exceptions were recorded in April, May and October where the highest abundance and biomass values of *P. peringueyi* were recorded in the lower reaches of the estuary. There were no apparent seasonal trends in the total abundance or biomass of *P. peringueyi* evident (Figures 3A and 3B). Rather, changes in the abundance and biomass of the shrimp were linked to breaching events that were recorded in late May and August.

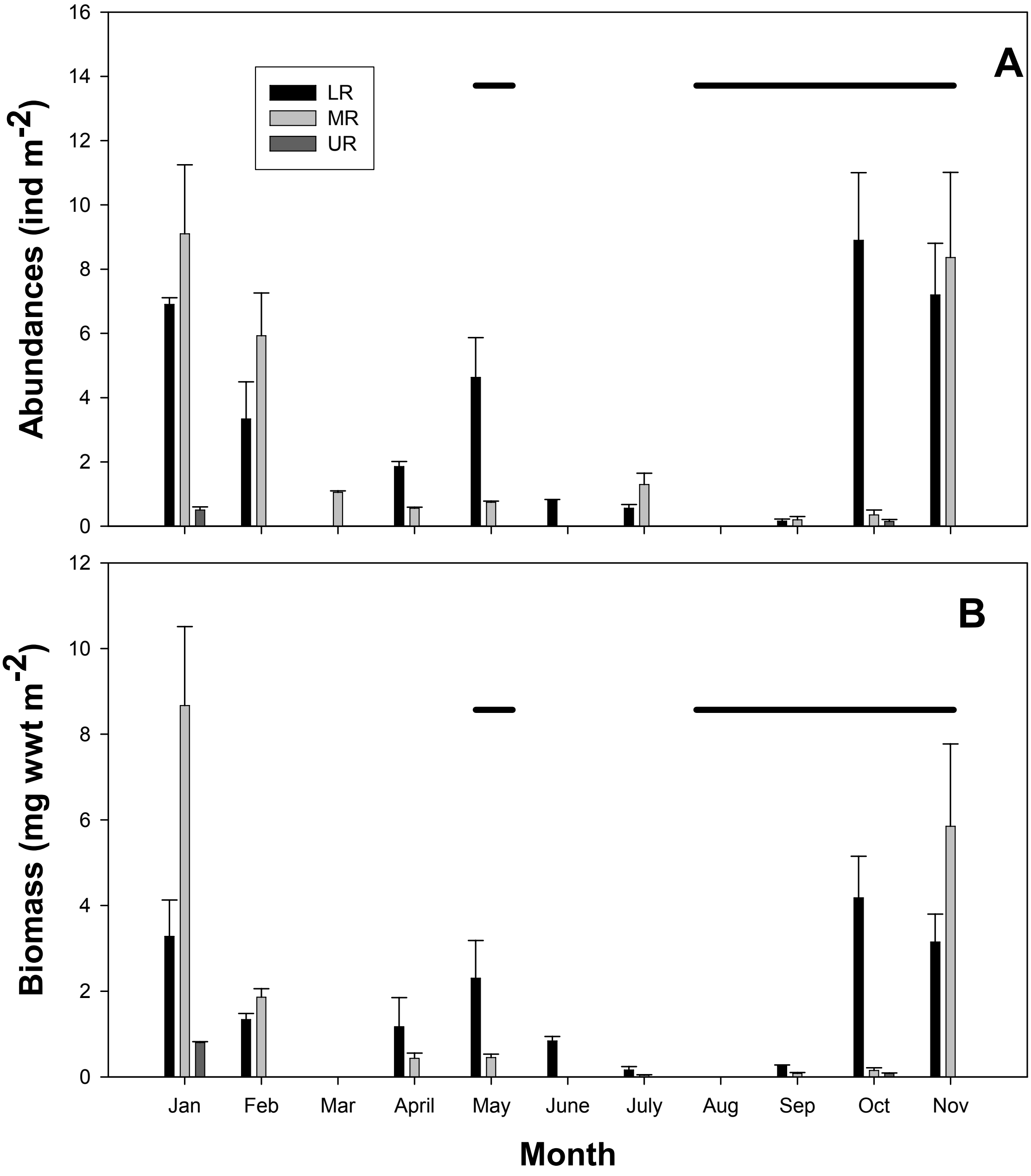

Figure 3. Spatial and temporal pattern in the total abundance and biomass of the caridean shrimp, *Palaemon peringueyi* in the temporarily open/closed East Kleinemonde Estuary. Thickened horizontal line indicates when the estuary was open. UR = upper reaches; MR = middle reaches; LR = lower reaches.

Population demographics of *P.peringueyi*

The population demographics of *P. peringueyi* in the East Kleinemonde Estuary exhibited a distinct temporal pattern. During the first seven months of the survey, the total abundance of the shrimp was dominated by adults, which comprised between 72 and 87% of all shrimp counted (Figure 4). The breaching event in August was associated with a dramatic increase in the contribution of juveniles to the total counts. During October and November, juveniles accounted for between 68 and 73% of all *P. peringueyi* collected (Figure 4).

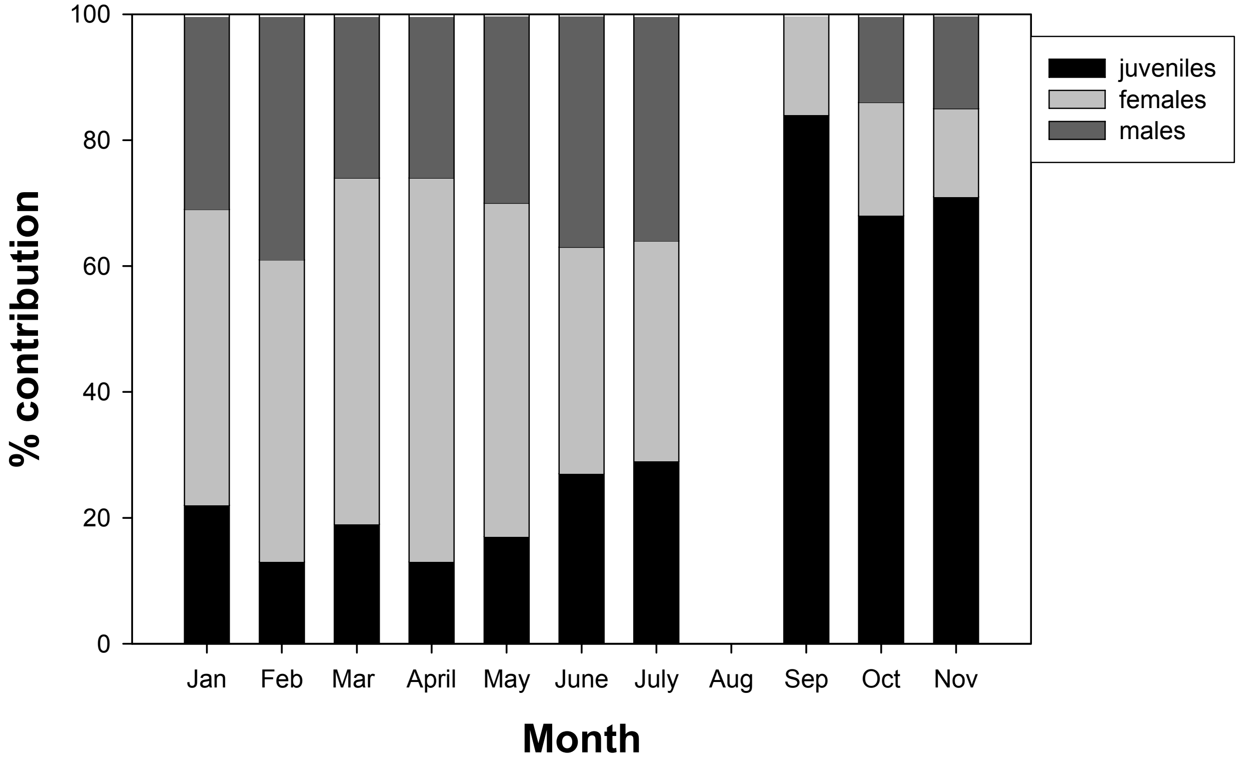

Figure 4. Population demographics of the caridean shrimp, *Palaemon peringueyi* in the temporarily open/closed East Kleinemonde Estuary. Data presented are the mean values from three stations.

In agreement with studies conducted on zooplankton and fish within temporarily open/closed estuaries (Vorwerk *et al*. 2003, Froneman 2004, Kemp & Froneman 2004, Bernard & Froneman 2005), the breaching event in late August was associated with a dramatic decrease in the total abundance and biomass of the caridean shrimp, *Palaemon peringueyi*, within the East Kleinemonde Estuary (Figures 3A and 3B). The observed trend can likely be associated with the outflow of biomass rich estuarine waters into the marine environment (Froneman 2004), and the decline in water level within the estuary would have been associated with a loss of habitat, mainly submerged littoral vegetation.

Recent studies conducted within TOCEs within the same geographic region have demonstrated that breaching events coincide with the recruitment of juvenile *P. peringueyi* into these systems (Bernard & Froneman 2005, Froneman 2006). The breaching event in August 2006, was, however, not associated with the recruitment of *P. peringueyi* into the estuary. A peak in the recruitment of *P. peringueyi* into permanently open estuaries occurs during summer, with a secondary minor peak, taking place in early winter (Emmerson 1986). The breaching event in late August therefore coincided with period when the shrimp is reproductively less active. It is worth noting that numerical dominance of juveniles observed in October and November 2006 suggests that recruitment had occurred by late spring.

The estimates of total abundances and biomass of *P. peringueyi* during this study are substantially lower than those recorded in permanently open estuaries within the same geographic region. For example, in the Swartkops and Kromme river estuaries (Eastern Cape), *P. peringueyi* attained abundance levels of 200 - 400 individuals m^{-2} with a dry biomass equivalent to between 3 and 6 g dry mass m^{-2} (de Villiers *et al*. 1999). It is worth noting, however, that the estimates of the shrimp abundance and biomass within the estuary are in the range reported in other temporarily open/closed estuaries within the same region (Bernard & Froneman 2005, Froneman 2006).

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Appendix J

Specialist Report: Fish

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INTRODUCTION

The fishes associated with the temporarily open/closed East Kleinemonde Estuary have been subject to considerable research attention over the past decade. Published findings of this research effort include descriptions of the fish community (Cowley & Whitfield 2001a, Strydom *et al*. 2003, Vorwerk *et al*. 2003), quantification of fish population sizes (Cowley & Whitfield 2001b), fish biomass and production estimates (Cowley & Whitfield 2002), composition of larvae in the marine environment adjacent to the mouth of the estuary (Cowley *et al*. 2001) and the recruitment strategies by marine-spawning species (Bell *et al*. 2001). This report summarizes the major findings from the above studies as well as the more recent work on larval and early juvenile fish ecology in the East Kleinemonde Estuary.

PART 1: LARVAE AND EARLY JUVENILES

Biological sampling procedures

Spring, summer, autumn and winter samples were collected, with additional sampling during mouth open phases. On each sampling trip, the littoral zone was sampled during daylight hours at seven sites (S1 - S7, Figure 1) along the lower, middle and upper reaches of the East Kleinemonde Estuary using a 1.5 m deep × 4.5 m long modified beach-seine net with a mesh aperture of 0.5 mm. Each littoral site was divided into three 25 m stretches and sampled by pulling the net across this distance parallel to the estuary bank. In total, 75 m consisting of three replicates was sampled at each marginal water site. The approximate volume of water filtered by each haul was 150 m^3, as the entire net was submerged during hauls.

At night, mid-channel plankton netting was conducted at five sites (P1 - P5, Figure 1) throughout the estuary using two slightly modified WP2 plankton nets (570 mm mouth diameter and 0.2 mm mesh aperture size) fitted with Kahlsico 005 WA 130 flow meters. The nets were held approximately 20 cm below the water surface for the duration of the tow. Towing speed ranged between 1 - 2 knots and each tow lasted for three minutes.

Biological sample treatment

Samples were preserved on site with 10% formalin in seawater. Whole samples were processed in the laboratory and larval and juvenile fishes removed and identified to the lowest possible taxon. Positively identified fishes were categorized according to the degree to which the species is dependent on South African estuaries (Whitfield 1998). All terminology pertaining to larval fish followed that of Neira *et al.* (1998). The term “larva” was used to designate all stages in the early life history from hatching to the attainment of a full fin ray complement, squamation and the subsequent loss of all larval characteristics, at which stage the “larva” becomes a “juvenile”. The term “larva” was further divided into yolk-sac, preflexion, flexion and postflexion stages. Larval fish density was expressed as numbers of larvae 100 m^{-3} (Strydom *et al.* 2003) and the modified beach seine-net catch was expressed as catch per unit effort (CPUE) in terms of numbers. One unit of effort refers to a single seine pulled parallel to the estuary bank over a distance of 25 m in littoral waters.

Statistical analyses

Variations in CPUE and density of larval and early juvenile fishes were tested using Mann-Whitney U-Tests. The Kruskall-Wallis ANOVA and Median test were used to assess differences in CPUE and density between the sites and seasons, as well as for differences in physical data between sites and seasons within the estuary. Multiple linear stepwise regressions (MLR) were used to ascertain whether environmental variables displayed any significant relationships with fish density and CPUE. MLR was used on total CPUE (all species included) and also individually on dominant species. In all statistical analyses, a significant level of $P < 0.05$ was used.

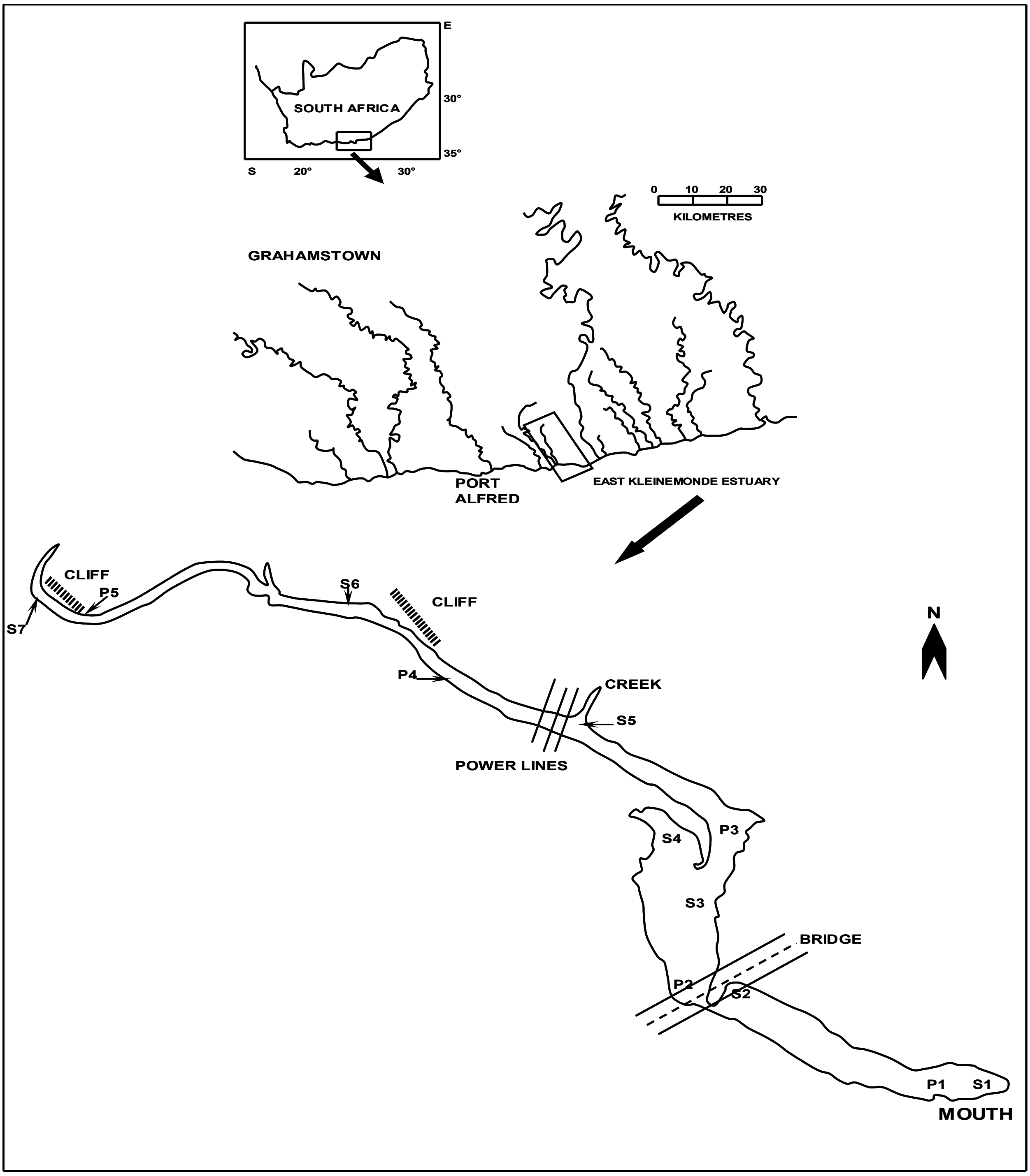


Figure 1. Geographic location of the East Kleinemonde Estuary, showing the positions of sampling sites. Seine net sites are numbers S1 - S7 and plankton towing sites are numbered P1 - P5.

RESULTS

Larval and early juvenile fish composition and assemblage dynamics

A total of 13 742 fishes were caught during this study of which 11 450 were caught along the margins and 2 013 in the mid-channel plankton tows. Seine catches at marginal sites along the estuary yielded 8 fish families and 13 species, mid-channel tows yielded 8 fish families and 16 species and surf seine catches yielded 11 fish families and 17 species.

Marginal sites

Seasonal CPUE trends

A significant ($P < 0.001$) difference in CPUE (all species combined) was recorded across seasons. The highest mean CPUE occurred during the summer of 2005 (911 fish haul^{-1}), coinciding with an opening event, which was significantly different from all other seasons sampled. CPUE showed no significant differences between sites during each season sampled in this study. The fish families Atherinidae, Clupeidae and Gobiidae were found to be dominant in margin catches. *Atherina breviceps*, *Gilchristella aestuaria* and *Glossogobius callidus* were the dominant species within each of these families, with mean total CPUE values of 28, 75 and 14 fish haul^{-1}, respectively.

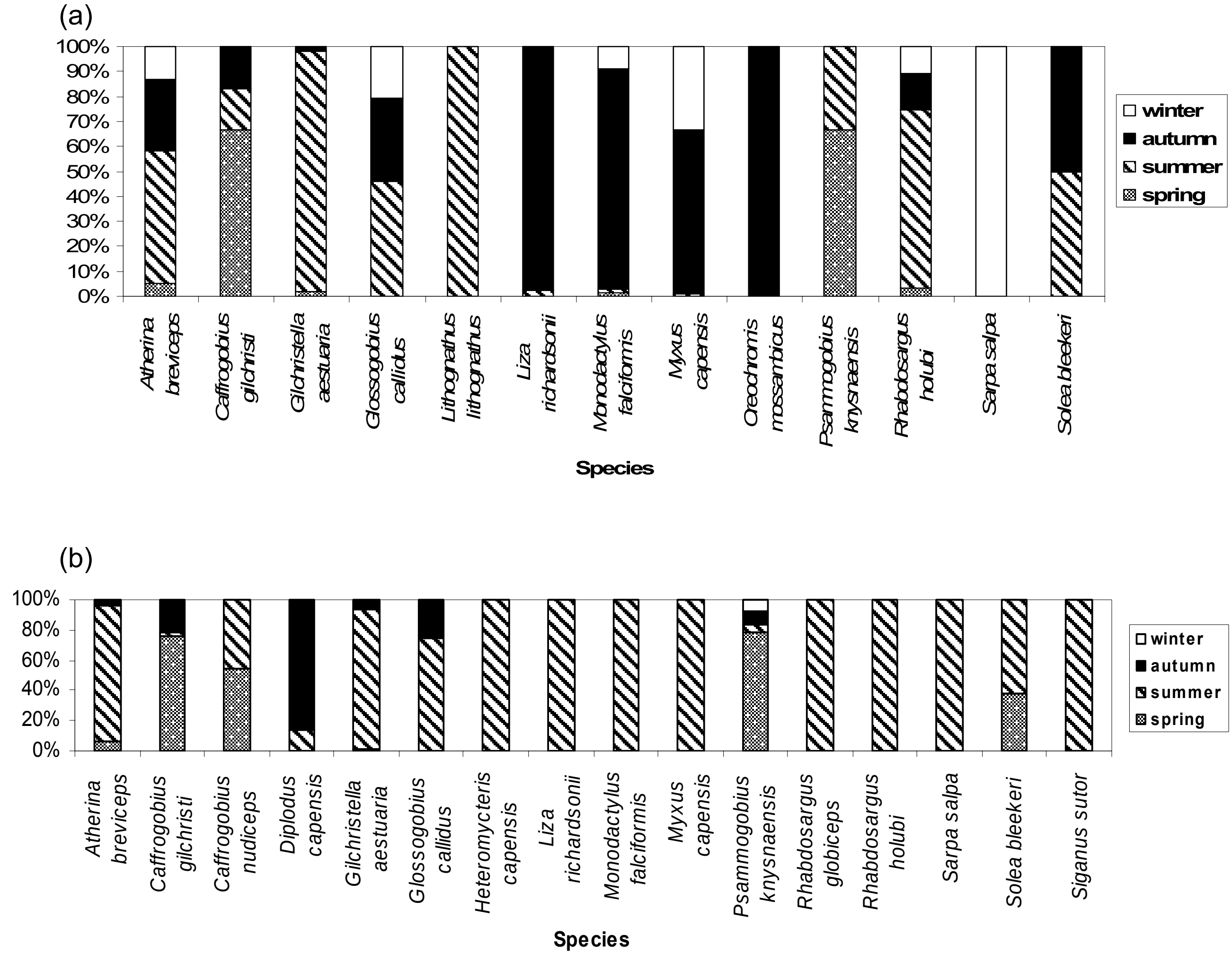


Figure 2. Seasonal contribution (percentage) of each species to the total early stage fish CPUE for seine catches (a) and density for mid-channel tows (b).

Species composition

A clear temporal variation in fish species composition was evident seasonally for seine net catches (Figure 2a). Estuary-resident fish species were present throughout all seasons, but dominated significantly (P<0.05) in summer. *Gilchristella aestuaria* was abundant in summer, while *Atherina breviceps* was present in all seasons sampled, but more so in summer and autumn. The gobiids *Caffrogobius gilchristi* and *Psammogobius knysnaensis* showed higher presence in spring, with some autumn occurrences. Estuary-dependent species, particularly the two mullet species *Myxus capensis* and *Liza richardsonii*, as well as *Monodactylus falciformis* were caught in autumn. *Rhabdosargus holubi* were significantly more abundant (P<0.05) during summer. The freshwater species *Oreochromis mossambicus*, was present in seine catches only in the upper reaches of the estuary during autumn. No marine straggler species were sampled in the marginal seine net catches.

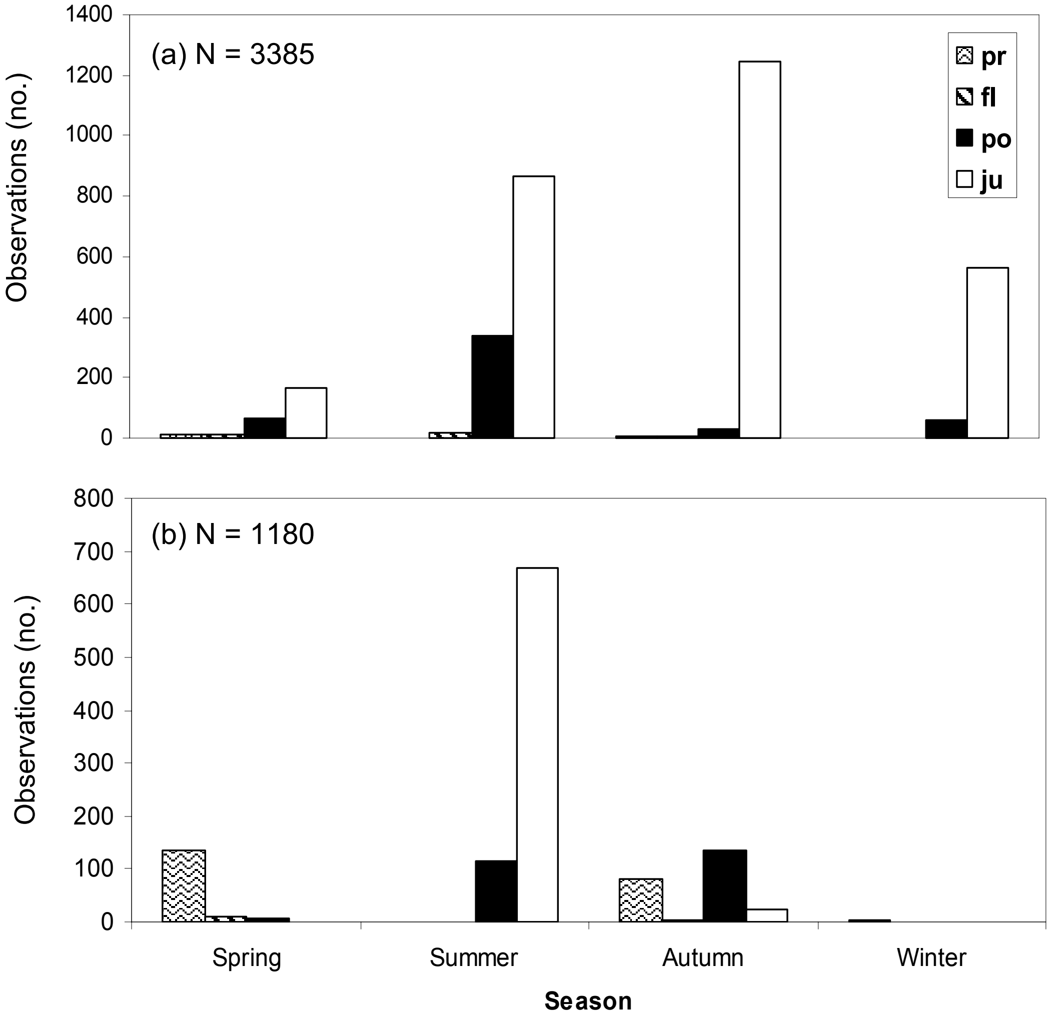


Figure 3. Frequency of occurrence of developmental stages for fishes (seasonally) in marginal seine net catches (a) and mid-channel tows (b) (pr = preflexion, fl = flexion, po = postflexion, ju = juvenile).

Developmental stages

Early juvenile stages dominated at marginal sites (Figure 3a), with abundance peaking in autumn after a summer mouth opening event. Flexion and postflexion larvae were most numerous during summer, while preflexion larvae peaked in abundance during spring. The juvenile stage was the dominant developmental stage for *Gilchristella aestuaria* and *Atherina breviceps* along the margins, while the preflexion stage was dominant for *Caffrogobius gilchristi* and the postflexion larval stage was dominant for *Rhabdosargus holubi*.

Estuary association

Seasonal presence of the different estuarine association categories is represented in Figure 4a. Estuary-resident (category I) species dominated in summer (mouth opening) ($P<0.05$), especially the species *Atherina breviceps*, *Gilchristella aestuaria* and *Glossogobius callidus*. Estuary-dependent (category II) species also showed a significant ($P<0.05$) increase in abundance in summer (mouth opening) and autumn. No marine straggler (category III) species were sampled in the marginal seine net catches. A single freshwater species (category IV), *Oreochromis mossambicus*, occurred during autumn.

Mid-channel sites

Seasonal CPUE trends

There was a significant ($P<0.001$) difference in catches of larval and early juvenile fishes between seasons. Highest catches were again recorded in the summer of 2005 ($P<0.05$), with a mean density of 1258 100m^{-3} (Table 1). There was no difference in fish density between sites, per season ($P>0.05$). The same fish families (Clupeidae, Atherinidae and Gobiidae) dominated both mid-channel and marginal sites sampled. There were significantly ($P<0.05$) higher densities of *Atherina breviceps*, *Gilchristella aestuaria*, *Glossogobius callidus* and *Rhabdosargus holubi* during summer compared to winter.

Table 1. Descriptive statistics for larval and early juvenile fishes occurring in different seasons and salinity zones in the East Kleinemonde Estuary during the study period.

	CPUE (no. haul^{-1})			Density (no. 100m^{-3})		
	Margin sites			**Mid-channel sites**		
	Mean	Median	Range	Mean	Median	Range
Total	159	29	0-3742	272	43	0-1895
Season						
Spring	12	6	0-77	104	28	6-600
Summer*	911	247	44-3742	1258	1271	293-1894
Autumn	98	62	0-366	118	89	16-332
Winter	45	12	0-254	2	0	0-10
Salinity zone						
Polyhaline	220	59	0-3742	336	67	0-1894
Euhaline	12	6	0-77	104	28	6-600

*Estuary mouth open

Species composition

Estuary-resident species and estuary-dependent marine species were abundant during summer (Figure 2b) when the mouth of the estuary was open. There were significant seasonal differences in density of *Atherina breviceps* ($P<0.05$), *Gilchristella aestuaria* ($P<0.05$) or *Glossogobius callidus* ($P<0.001$), with higher catches occurring in summer. *Myxus capensis* and *Rhabdosargus holubi* also showed significantly higher densities in summer ($P<0.05$) during the open phase. The two gobiids, *Caffrogobius gilchristi* and *Psammogobius knysnaensis*, however, displayed their highest densities in spring. The larvae of the sparid *Diplodus capensis* occurred in mid-channel tows during autumn sampling, while *Sarpa salpa* was recorded in summer tows while the estuary was open.

Developmental stages

Larval fishes were recorded through all developmental stages in the mid-channel waters (Figure 3b) when compared with those observed along the margins (Figure 3a). Spring densities were the highest for preflexion and flexion larvae, while postflexion larvae and juvenile fishes reached their highest densities in summer (mouth opening event) and autumn. More juveniles were recorded during the opening event in summer. Few juveniles were observed in spring and winter. The same developmental stages for resident species that were dominant along the margins such as *Atherina breviceps* (juvenile), *Gilchristella aestuaria* (juvenile), *Caffrogobius gilchristi* (preflexion) and *Rhabdosargus holubi* (postflexion) were dominant in the mid-channel catches. *Atherina breviceps* and *Gilchristella aestuaria* were represented by all four developmental stages.

Estuary association

Catches in the mid-channel showed a similar pattern to that observed along the margins. Estuary-resident species were dominant during all four seasons, but showed significantly ($P<0.05$) higher densities during summer when the mouth of the estuary was open. Marine species that are dependent on estuaries, like the mugilids *Liza richardsonii* and *Myxus capensis* as well as *Monodactylus falciformis* and *Rhabdosargus holubi*, were also present in significantly ($P<0.05$) higher densities during summer. No freshwater species were present in the mid-channel catches.

PART 2: MOUTH DYNAMICS AND FISH COMMUNITY STRUCTURE

Larval community comparison: 1999 versus 2006

No mouth opening events were recorded in the East Kleinemonde Estuary during earlier sampling conducted between July 1998 and April 1999. However, an estuary opening event was observed in the summer of 2005. Mean salinity recorded seasonally was generally lower in 1999, except for winter when salinity values were more similar. Temperatures were similar between the two sampling years. Water clarity also shows little fluctuation between the two datasets, although the mean water clarity for summer 2006 during the opening event was several times higher than that recorded in 1999 when the estuary was closed. Mean fish density (1621 individuals 100 m^{-3}) for the 1999 dataset was significantly higher ($P<0.05$) than the mean fish density (570 individuals 100 m^{-3}) for the 2006 study. However closer examination of the densities of estuary resident and estuary-dependent marine species showed different mouth conditions affected these groups differently (Table 2). Density of estuary resident species was significantly higher ($P<0.001$) during the closed phase and the density of estuary-dependent marine species peaked during the sampling year characterized by an opening event, although this increase in mean density from 0.13 to 2.18 individuals 100 m^{-3} was not statistically significant. The maximum density of larval fishes recorded in the estuary occurred in 2006 (Table 2).

Table 2. Comparative descriptive statistics for larval fish density (individuals 100 m^{-3}) recorded during two plankton surveys of the East Kleinemonde Estuary under different mouth conditions.

	1999			2006		
	Mean	**Median**	**Range**	**Mean**	**Median**	**Range**
All species	1621	104	0 – 13881	570	4	0 – 17431
Estuary dependent species	0.1	0	0 – 13	2	0	0 – 220
Estuary resident species	202	0	0 – 9399	42	0	0 – 1641

Can larvae distinguish water from different sources?

Although the recruitment of larvae and juveniles into estuaries has been well documented, little is known about the factors governing the immigration of marine-spawning fishes into estuaries. Several physical factors have been proposed as stimuli that could elicit a recruitment response in estuary-associated fish species, including current speed, salinity, temperature and turbidity. Furthermore, numerous researchers have suggested that olfaction is the primary sense responsible for the successful location of estuarine environments (e.g. Whitfield 1994, Cowley *et al*. 2001b, Strydom 2003).

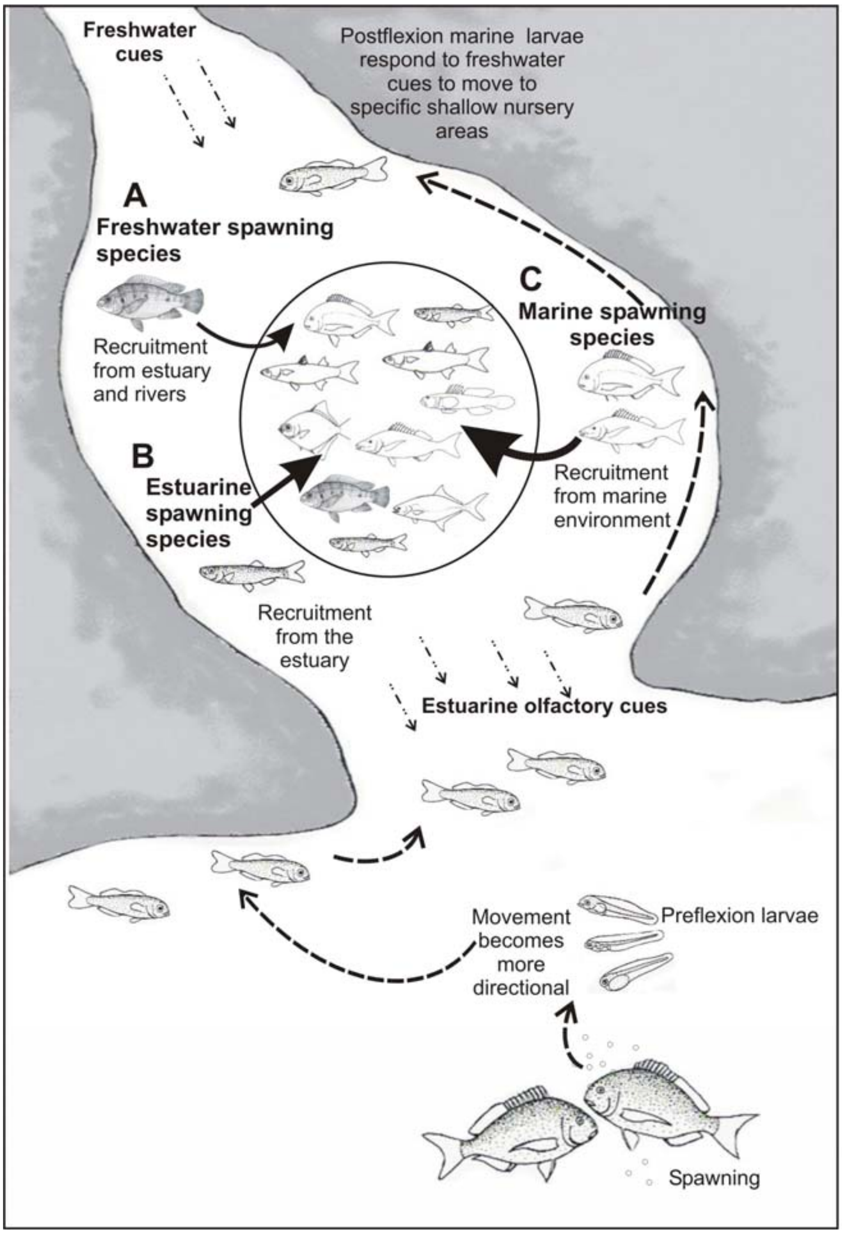
A series of laboratory experiments were conducted using a choice chamber to test the attraction of *Rhabdosargus holubi* larvae to different water types (riverine, estuarine, coastal and marine). Prior to each experiment the estuarine and river water was adjusted to a salinity of 35 psu using artificial sea salt (Red Sea Salt® and Dopan Marine Salt®) to eliminate salinity as an influencing factor and also to prevent mixing of different water types in the experimental choice chamber (James 2006).

In the first experiment, larvae that had recently recruited into an estuary showed a preference for estuary and river water over seawater. However, when given a choice between estuary and river water, or estuary and river water and a mixture of the two, the results were inconclusive. In the second experiment, larvae that were smaller in size (10 - 11 mm SL) and had not yet recruited into an estuary were found to be significantly attracted to estuary water and weakly attracted to river water. These results suggest that migrating larvae in the sea respond primarily to cues in estuary water, and to a lesser extent river water, when recruiting into estuaries and probably develop a stronger attraction to river water once in the estuary, as observed in the first experiment. It is also important to note that estuarine water may contain cues of riverine or catchment origin. Furthermore, when given a choice between seawater and surf-zone water, *Rhabdosargus holubi* larvae selected surf-zone water over sea water (James 2006).

The findings of this study are in agreement with the pattern proposed by Whitfield (1994), with additional refinements. Based on the available evidence the proposed recruitment pattern adopted by marine-spawning estuary-dependent species is as follows. Initially, larvae are attracted to surf zone water and maintain position within this zone but may move laterally along the coast. Only later, when accumulating and migrating into estuaries, are larvae attracted to estuarine water as opposed to other water types. Once in the estuary, larvae usually continue to move up the system towards littoral nursery areas and are then mainly attracted to river water cues. Larvae and early juveniles probably use olfaction, along with other cues such as salinity, temperature and turbidity gradients to ‘fine tune’ their movements to specific nursery areas within estuaries. Olfactory cues may be particularly important in orientating towards upstream nursery areas in TOCEs where strong horizontal gradients are often absent.

Fish assemblage comparisons: 1995-2005

A large mesh seine net (50 m x 2 m) with a 15 mm bar mesh was used to sample the juveniles and sub-adults of marine-spawning species biannually (summer and winter) from 1995 to 2005 (James 2006). The number of large mesh seine net hauls per sampling trip usually varied between 8 and 18, with the final number dependent on access to sampling sites, as well as a flattening of the species: sampling effort curve. On some occasions when the mouth was closed several sites were completely submerged and, during the open mouth phase, some sites were completely exposed. On each sampling occasion all possible littoral habitat types in the lower, middle and upper reaches of the estuary were sampled. The seine net was laid in a semi-circle from the bank using a small motorised boat and then hauled ashore. All fish caught were identified and measured to the nearest millimetre standard length (SL) before being released.



Community structure determined by the recruitment of (A) freshwater, (B) estuarine and (C) marine spawning species

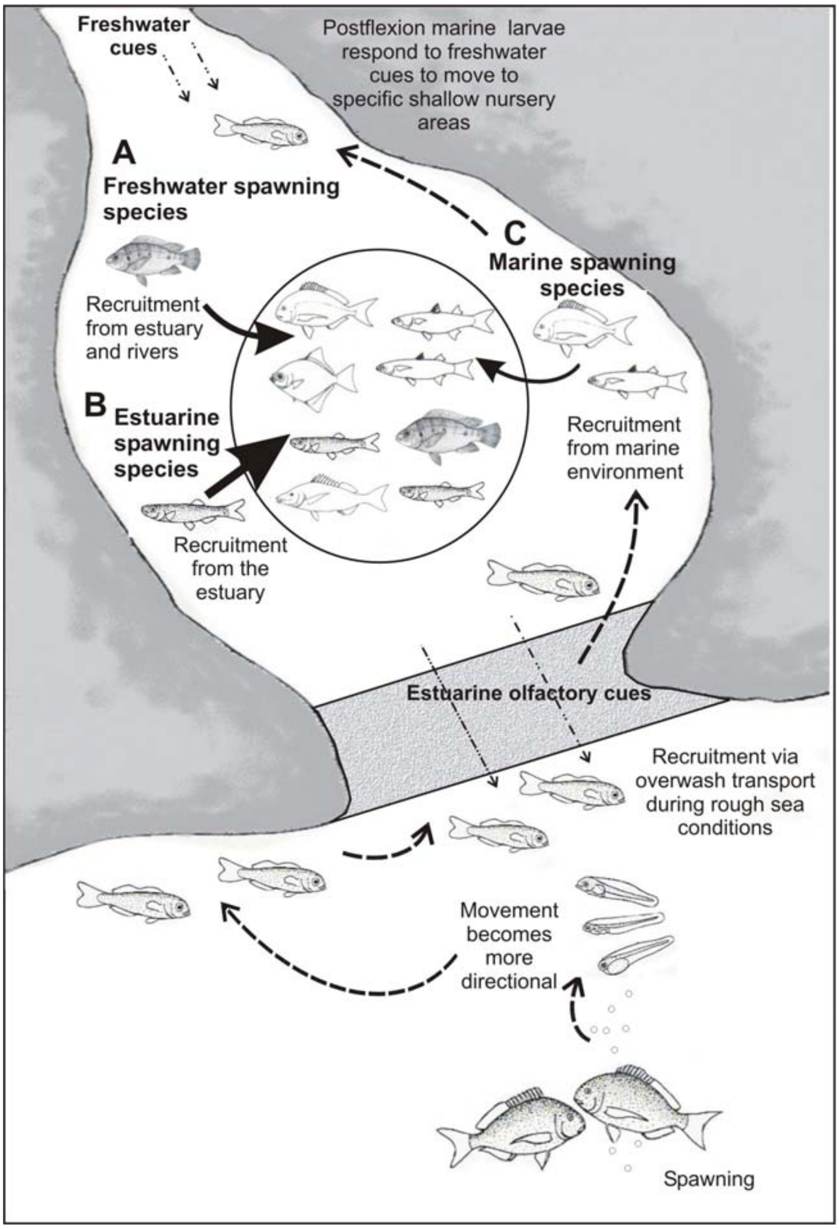
A) Spawning and nest building by *Oreochromis mossambicus* is disrupted when the mouth opens during peak breeding season resulting in a decrease in recruitment.

B) During periods of high river flow eggs and larvae of estuarine spawning species may be swept out to sea resulting in a decrease in recruitment.

C) The recruitment of marine spawning species is enhanced when the mouth is open (particularly during spring), thus leading to higher species richness of marine spawning species and a more stable marine community.

Figure 4. Major contributors to fish community structure in the East Kleinemonde Estuary during following an open mouth phase. The movement of estuary-associated marine fish larvae from marine spawning grounds into the estuary is also shown (after Cowley *et al.* 2008).

Numerous authors (e.g. Bell *et al*. 2001, Vorwerk *et al*. 2003) have suggested that the timing, duration and frequency of mouth opening events play an important role in determining fish species composition, diversity and seasonality within temporarily open/closed estuaries (TOCEs). According to Potter *et al*. (1993) researchers have not yet determined whether long-term changes in fish communities of TOCEs reflect the period that these estuaries were isolated from the sea. This study provided the first opportunity to test this hypothesis because it combined a long-term dataset of fish catches under different mouth states (e.g. see Figures 4 and 5) over different time scales.



Community structure determined by the recruitment of (A) freshwater, (B) estuarine and (C) marine spawning species

A) Closed conditions result in inundation of marginal vegetation, stable water levels and an absence of water currents, thus providing favourable conditions for breeding of *Oreochromis mossambicus* and enhanced recruitment.

B) Recruitment of estuarine spawning species is greater during the closed phase as eggs and larvae are retained in the estuary, conditions are physically more stable, and marginal vegetation is inundated. Freshwater input leading to mouth opening may also enhance populations through nutrient input.

C) The recruitment of marine spawning species is lower when the mouth is closed (particularly during spring). Species such as *Rhabdosargus holubi* and various mullet are able to recruit via overwash transport but other species such as *Lithognathus lithognathus* and *Pomadasys commersonnii* are unable to recruit, resulting in lower marine species richness and a less stable marine community.

Figure 5. Major contributors to fish community structure in the East Kleinemonde Estuary during a prolonged closed phase. The movement of estuary-associated marine fish larvae from marine spawning grounds into the estuary is also shown (after Cowley *et al.* 2008).

The timing of mouth opening has a significant effect on marine species composition in the estuary, with multi-dimensional scaling grouping years into two distinct groups (Figure 6). More species were recorded during years that succeeded spring (September - November) mouth opening events than in years following no mouth opening events in spring. Mean annual CPUE for species that recruit predominantly in spring were higher in ‘spring opening’ years. Species that are known to recruit during both overwash and open mouth conditions were consistently recorded each year, irrespective of a ‘spring opening’.

Two different periods of stability were identified in the East Kleinemonde Estuary, viz. 1995 - 2000 and 2001 - 2005. During the 2001 - 2005 period a higher value for Kendall’s coefficient of stability (*W*) was recorded and there was also an increase in inter-annual community stability (IMD) and seriation (IMS) from the ‘other’ years to the ‘spring’ years. These results suggest that optimum mouth opening for marine fish recruitment is during spring (September - November) in Cape TOCEs and the predominance of spring opening events between 2001 and 2005 probably allowed for the regular recruitment of a wide variety of marine species into the estuary.

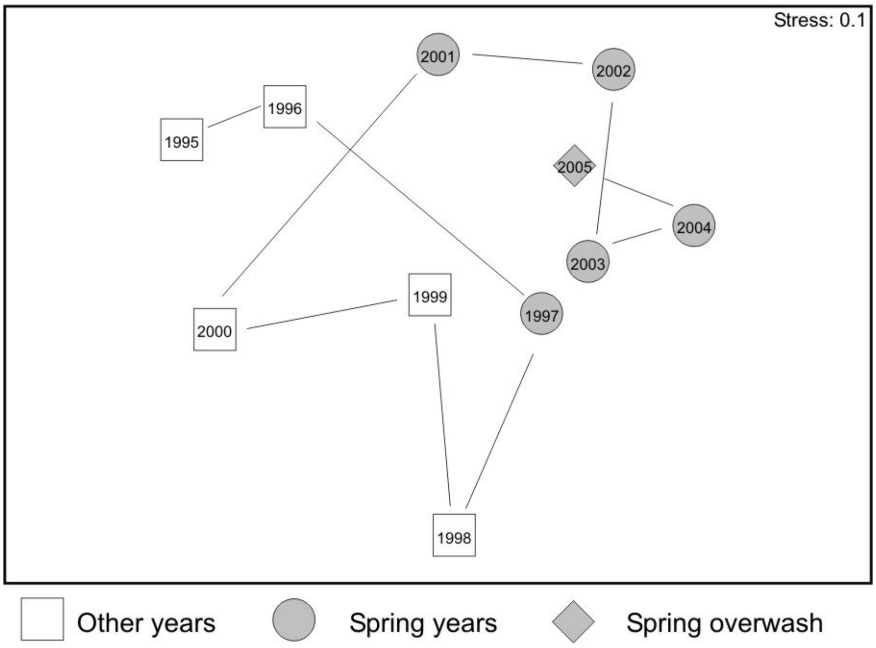


Figure 6. Ordination of annual seine net fish CPUE data from the East Kleinemonde Estuary between 1995 and 2005.

Individual fish populations in the East Kleinemonde Estuary were dynamic as the abundance of all species varied markedly between years. For example, maximum abundance for *Rhabdosargus holubi* (264 fish haul^{-1} in 2003), *Myxus capensis* (83 fish haul^{-1} in 2004) and *Gilchristella aestuaria* (1662 fish haul^{-1} in 2005) was several times greater than in the years of least abundance (21 fish haul^{-1} for *R. holubi* in 2001, 0.2 fish haul^{-1} for *M. capensis* in 1995 and 97 fish haul^{-1} for *G. aestuaria* in 1996). This was because the recruitment strengths of juveniles varied between years and the years when recruitment was greatest varied between the different species.

Similarly, Cowley & Whitfield (2001b) found that overall populations of marine migrant species associated with the East Kleinemonde Estuary are characterised by a high degree of inter-annual variability. For example, the total population size of all marine fishes in the estuary increased by almost eight-fold from a mark-recapture experiment conducted between October 1994 and December 1994 (~18 000 individuals) to a mark-recapture experiment conducted between October 1995 and February 1996 (~133 000 individuals). The large inter-annual variability was attributed to both abiotic (estuary mouth state) and biotic conditions such as spawning success and larval survival.

In the East Kleinemonde Estuary the timing of mouth opening events (abiotic factors) and life-history pattern (biotic factors) together influenced species composition and abundance. Species with extended breeding seasons that recruit during overwash and open mouth conditions or breed in the estuary dominated catches numerically (Table 3). In contrast, species with restricted spawning seasons, most of which only recruit into estuaries during open mouth conditions, were found in low numbers within the estuary.

Cowley & Whitfield (2001a) also found that mouth state is important for the reproductive success of estuarine-spawning species in the East Kleinemonde Estuary. Reproductive activity is halted during low level water conditions following a mouth opening event. On the other hand, closed mouth conditions result in more stable physical conditions, elevated water levels and habitat inundation, which results in an increase in breeding success. Submerged macrophyte beds, which expand during extended closed mouth phases, are an important habitat for the estuarine pipefish *Syngnathus watermeyeri*, a species which disappeared from the estuary when aquatic macrophytes died back between 2004 and 2006.

Table 3. Reproductive seasonality and recruitment behaviour of the 10 most abundant species caught in the East Kleinemonde Estuary.

Species	**Spawning period***		**Recruit during overwash events**	**Life history**	**Rank**
Gilchristella aestuaria	Breeds all year round	Extended	NA	Ia	1
Atherina breviceps	September - February	Medium	NA	Ia	2
Rhabdosargus holubi	July - February	Extended	Yes	IIa	3
Glossogobius callidus	October - November	Short	NA	Ib	4
Myxus capensis	March - November	Medium	Yes	Vb	5
Monodactylus falciformis	October - February	Medium	Yes	IIa	6
Oreochromis mossambicus	September - February	Medium	NA	IV	7
Liza richardsonnii	September - March	Extended	No	Iic	8
Liza dumerili	December - February	Short	Yes	IIa	9
Lithognathus lithognathus	June - August	Short	No	IIa	10

* Short 1 – 3 months; Medium 4 – 6 months; Extended > 7 months

Analysis of the long-term small seine dataset suggests that inter-annual changes in the abundance of certain estuarine-spawning species were related to rainfall and thus river pulses. The filter-feeding *G. aestuaria* is the most abundant species in the East Kleinemonde Estuary and forms an important link in the food chain in South African estuaries because it is preyed upon by various predatory fish. Martin *et al*. (1992) recorded an order of magnitude increase in the abundance of *G. aestuaria* in the St Lucia Estuary following a cyclone and attributed this response to the phytoplankton bloom and increased zooplankton stocks associated with the river flooding. However, major flooding in estuaries can also result in a decrease in estuarine resident species. Strydom *et al*. (2002) recorded an absence of young *G. aestuaria* in the water column of the Great Fish Estuary during periods of high flow and attributed this to large numbers of eggs and larvae being swept out of the estuary.

Timing and duration of mouth opening events can influence the recruitment success of *O. mossambicus* in the East Kleinemonde Estuary. In 2004 the mouth opened in late December, thus allowing successful breeding, while in 2005 the mouth opened during November before the cycle could be completed. Recruitment success was consequently very poor in 2005/06 (Ellender 2006).

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Appendix K

Specialist Report: Birds

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INTRODUCTION

Little historic information is available on the avifauna of the East Kleinemonde Estuary. One single count of all waterbirds exists for 1980. Cowley (1998) and Terörde (2005a) studied the population dynamics of the piscivorous component of the avifauna in 1995 and 2005 respectively. The avifauna of the temporarily open/closed East Kleinemonde Estuary has now been studied in detail as part of an MSc thesis from March 2005 to November 2006. This review will focus on this recent work conducted.

STUDY AREA AND METHODS

Sampling procedure

Seventy-two bird counts were conducted on the East Kleinemonde Estuary from March 2005 to November 2006. Counts were carried out from a motorised boat or on foot, depending on water levels, from the mouth to the head of the estuary. The location of every encountered waterbird was noted on a GPS referenced map (Figure 1). Only non-terrestrial estuary-associated species were counted. Terrestrial feeding birds such as Egyptian Goose, Black-headed Heron and Brown-hooded Kingfisher were not included in the study. A total of 29 counts were carried out in summer (October - March) and 43 in winter (April - September). Fifty-one of these counts were conducted while the estuary mouth was closed and 21 while the mouth was open to the sea.

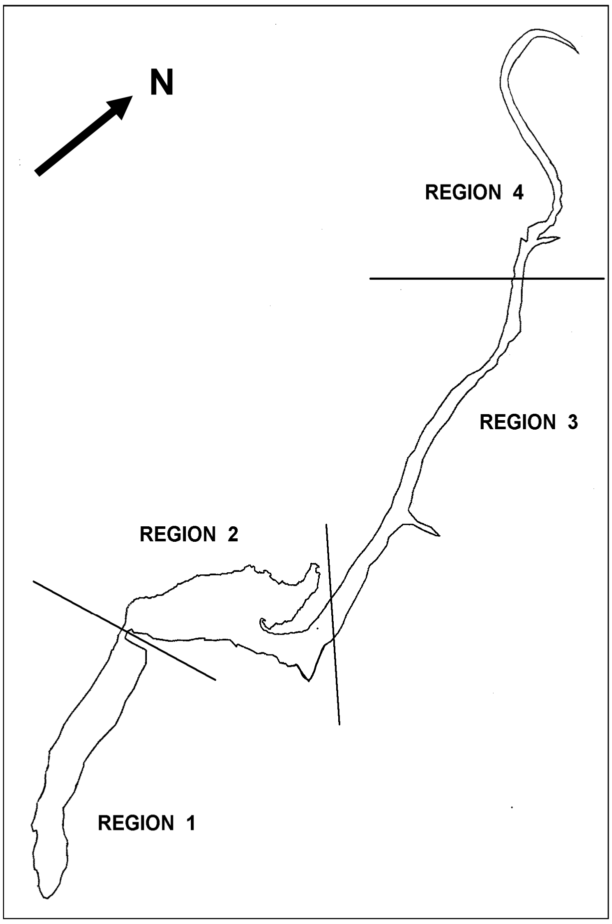

Figure 1. Map of the East Kleinemonde Estuary, showing the divisions between the main counting areas.

Division of the estuary into regions

The estuary was divided into four regions (Figure 1) according to habitat characteristics, as follows:

Region 1 extends from the mouth to below the bridge and is characterised by human habitation and disturbance, little natural vegetation on the riverbanks and relatively deep water. It has a maximum surface area of 8.3 ha and a maximum width of 115 m.

Region 2 comprises the area above the bridge where the estuary becomes very wide. The maximum surface area of this region is 11.8 ha with a maximum width of approximately 200m. This region has an extensive shallow salt marsh area and few human settlements on both banks. The vegetation is dominated by *Euphorbia* and *Aloe* species, on the western bank above the salt marsh, and *Phragmites* reeds on the eastern bank.

Region 3 of the estuary becomes narrower with degraded veld on the western slope and valley bushveld on the steeper eastern slope. The maximum width of this region is approximately 55 m with a maximum surface area of 5.5 ha.

Region 4 comprises the upper reaches of the estuary which have relatively undisturbed valley bushveld on the steep eastern slopes and degraded veld used for cattle farming on the flatter western slopes. This region has a maximum surface area of 2.7 ha and a maximum width of 30 m. In the upper reaches of Region 4, the western bank becomes very steep and rocky and valley bushveld dominates.

Statistical analyses

Abundance

The mean number of birds was calculated for summer, winter, closed and open estuary mouth conditions. The data were not normally distributed, therefore nonparametric analyses were used. Differences between the mean number of birds during summer and winter months, as well as during open and closed mouth conditions were tested for significance using the Mann-Whitney U-test. The frequency of occurrence for each species was calculated by dividing the number of times a species was present by the total number of counts (n = 52). The species were then classified according to an encounter rate ranking. Species present less than 25% of the time were classified as rare, those present 25 - 49% of the time as occasional, 50 - 74% as frequent and 75 - 100% as regular users. The mean number of individuals per count was calculated for all species.

Taxonomic composition and diversity

All recorded birds were identified to species level. A summary of represented orders and families was compiled.

Community composition and dietary guilds

Species were grouped according to feeding methods into aerial diving piscivores (terns, gulls, kingfishers, raptors), wading piscivores (herons, egrets, spoonbills, ibises, hamerkops), pursuit swimming piscivores (cormorants, grebes, darters), waders (oystercatchers, thick-knees, greenshanks, sandpipers, plovers, lapwings, etc) and herbivores (ducks, teals). The contribution of each dietary guild to the total population was determined from the number of individuals of a guild recorded during the entire study period. Summer and winter community composition and spatial distribution across the four regions in summer and winter were also compared. The mean number of individuals per count was calculated for every guild for summer, winter, open and closed estuary mouth. The results were compared using the Mann-Whitney U-test.

Spatial distribution patterns

The distributions of individuals across the four regions in summer and winter were compared.

Correlations of feed guild composition with water level

The change in the abundance of feed guilds in response to changes in water level was compared.

PRESENT STATUS OF THE AVIFAUNA

Species richness, abundance and frequency of occurrence

The mean number of birds recorded per count at the East Kleinemonde Estuary (Table 1) was 63.7 (SD ± 28.5). There was no significant difference between summer and winter mean abundances. There was, however, a significant difference between the mean number of birds recorded during open and closed conditions (Mann-Whitney U-test: $Z = 2.53$; $U = 331$, $P < 0.05$).

Table 1. Descriptive statistics for bird counts conducted on the East Kleinemonde Estuary under open and closed estuary mouth conditions and in summer and winter months.

	Mean number of individuals	**Standard deviation**	**Range**	**Significant difference**
Total	63.6	28.5	21-144	-
Summer	66.8	24.6	35-120	No
Winter	61.6	31.0	21-140	
Open	76.4	29.8	32-144	Yes
Closed	58.4	27.0	21-127	

Three species were encountered in more than 75% of counts and were classified as regular users (Figure 2). Only Grey Heron was recorded during every count. Eight species were encountered frequently, nine species occasionally and a further 27 species rarely.

The three most frequent species were also the most abundant (Table 2), with Reed Cormorant having the highest mean number of individuals per count. Of the 20 most abundant species only three can be regarded as full time residents (Table 2). Other breeding species, such as Great Egret and Grey Heron, which are present year round, often fly elsewhere to feed, mostly to the adjacent larger West Kleinemonde Estuary. Reed Cormorant, African Spoonbill, Little Egret and others use the estuary mainly for feeding and roosting, but breed elsewhere.

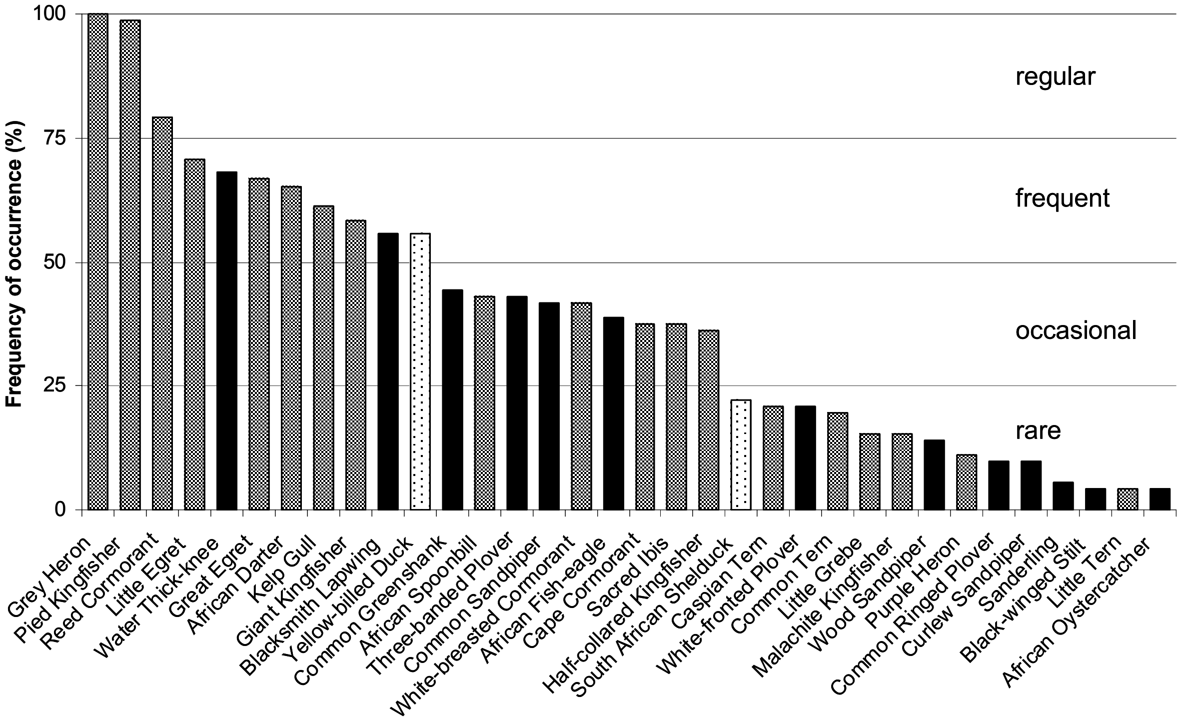

Figure 2. Frequency of occurrence of birds recorded at the East Kleinemonde during the study period. Grey bars represent piscivorous species, black bars invertebrate-feeding species and white bars herbivorous species. Species with a frequency of occurrence of less than 3% are not shown.

Table 2. Descriptive statistics for the 20 most abundant species recorded at the East Kleinemonde Estuary during the study period.

Species	Mean	Standard deviation	Range	Estuary use
Reed Cormorant	9.2	11.7	0-60	Feeding, roosting
Pied Kingfisher	6.8	3.4	0-16	Resident
Grey Heron	6.6	4.2	1-23	Breeding, roosting, some feeding
Water Thick-knee	5.9	5.8	0-20	Resident
Great Egret	3.7	5.9	0-24	Breeding, roosting, some feeding
African Spoonbill	3.6	6.5	0-33	Roosting, feeding
Little Egret	3.1	4.6	0-29	Roosting, feeding
Yellow-billed Duck	2.7	4.4	0-24	Feeding, roosting
African Darter	1.7	2.4	0-13	Feeding, roosting
Curlew Sandpiper	1.6	6.1	0-34	Irregular feeding
Common Greenshank	1.6	2.9	0-13	Feeding
White-breasted Cormorant	1.4	3.8	0-29	Roosting, feeding
Blacksmith Lapwing	1.3	1.6	0-9	Feeding, breeding
Three-banded Plover	1.2	1.9	0-9	Feeding
Kelp Gull	1.1	1.1	0-5	Feeding, roosting
Cape Cormorant	1.1	1.7	0-7	Roosting, feeding
Sacred Ibis	0.9	1.4	0-6	Roosting, feeding
Common Sandpiper	0.8	1.2	0-6	Irregular feeding
Giant Kingfisher	0.8	0.8	0-3	Resident
White-fronted Plover	0.8	2.1	0-14	Feeding (mouth area only)

Taxonomic composition and diversity

A total of 48 non-passerine waterbird species were recorded on the East Kleinemonde Estuary during 2005 - 2006, with 44 of these species being recorded during summer and 38 in winter. A total of 17 families of waterbirds in six orders were represented on the estuary, with Charadriiformes, making up the majority (48%) of recorded species (Table 3). Twenty-nine percent of recorded species belonged to the Order Ciconiiformes, 10% to Anseriiformes and 6% to Coraciiformes. Gruiformes and Falconiformes were only represented by one species (2%). A total of 16 species were long-distance migrants.

Table 3. Numbers of species of different groups that have been recorded on the East Kleinemonde Estuary (non-passerine waterbirds, excluding vagrants).

Order	**Family**	**Common names**	**Total**	**Migratory**
Anseriformes	Anatidae	Ducks, geese	4	
Coraciiformes	Alcedinidae	Alcedinid kingfishers	2	
	Dacelonidae	Dacelonid kingfishers	2	
Gruiformes	Heliornithidae	Finfoots	1	
Charadriiformes	Scolopacidae	Whimbrels, greenshanks, stints, ruff, sandpipers, turnstones	10	10
	Burhinidae	Thick-knees	1	
	Haematopodidae	Oystercatchers	1	
	Recurvirostridae	Stilts	1	
	Charadriidae	Plovers, lapwings	6	3
	Laridae	Gulls, terns	4	3
Falconiformes	Accipitridae	Raptors	1	
Ciconiiformes	Podicipedidae	Grebes	1	
	Phalacrocoracidae	Cormorants	4	
	Ardeidae	Herons, egrets	5	
	Scopidae	Hamerkop	1	
	Threskiornithidae	Ibises, spoonbills	2	
	Ciconiidae	Storks	1	

Community composition and dietary guilds

Piscivorous birds were numerically dominant at the East Kleinemonde Estuary, making up 70% of all recorded individuals. Invertebrate feeding waders formed the second most numerous component (24%). Waterfowl were particularly scarce in the estuary, making up only 6% of all recorded individuals.

In summer, the piscivorous component was predominently wading piscivores (Figure 3). The percentage of pursuit swimming piscivores (cormorants, grebes, darters) increased considerably from summer (17.3%) to winter (27.5%), mostly due to cormorants that arrive in winter to utilise the estuary for feeding. Their abundance increased significantly ($P<0.05$) from summer to winter (Table 4).

Resident wader numbers increased in winter, while migratory waders were virtually absent from the estuary during this time (Table 4). Therefore the overall wader component remained relatively constant throughout the year. Waterfowl were only present in small numbers, but their numbers did increase slightly in the winter months.

Table 4. Mean, standard deviation and range of numbers of individuals of different groups of birds recorded per count during the study period in summer (October - March) and winter (April - September).

	Summer (N = 29)			Winter (N = 43)			
	Mean	Standard deviation	Range	Mean	Standard deviation	Range	Significance
Aerial diving piscivores	11.4	5.0	1-25	10.5	4.9	3-23	Not significant
Pursuit swimming piscivores	10.1	10.2	0-34	16.3	16.9	1-75	P<0.05
Wading piscivores	21.8	16.3	3-60	15.6	11.0	1-48	Not significant
Resident waders	6.6	4.5	0-14	12	10.2	0-50	P<0.05
Migratory waders	10.4	13.1	0-50	0.9	1.5	0-8	P<0.001
Herbivores	1.9	2.0	0-6	4.3	5.6	0-24	Not significant

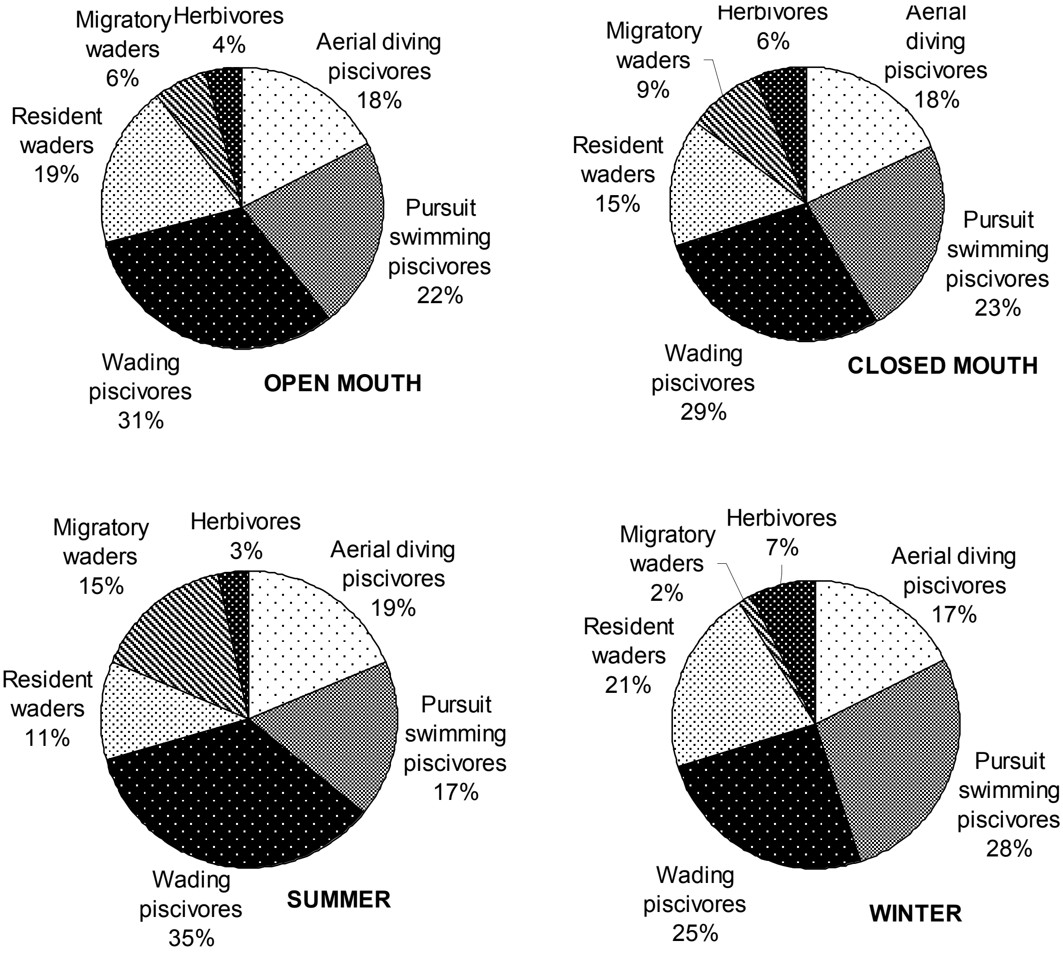

Figure 3. Community composition of the avifauna at the East Kleinemonde Estuary during open and closed estuary mouth conditions and in summer and winter months.

The estuary mouth was open to the sea mainly in the winter months of the study period, with only six out of 21 bird counts conducted in summer under open conditions. Therefore a comparison between open and closed conditions is largely a within-winter issue. There was a significant increase in the mean number of aerial diving piscivores, wading piscivores and resident waders from closed to open mouth conditions (Table 5).

Spatial distribution patterns

In summer, only 13% of birds were found between the estuary mouth and the bridge (Region 1). Region 2 was the most frequented area of the estuary with 53% of birds occurring there. Twenty percent and 13% of birds were found in Regions 3 and 4 respectively. In winter, 24% of birds were in Region 1, 49% in Region 2, 16% in Region 3 and 11% in Region 4.

Table 5. Mean, standard deviation and range of numbers of individuals of different groups of birds recorded per count during the study period under open and closed estuary mouth conditions.

	Closed (N = 51)			**Open (N = 21)**			
	Mean	Standard deviation	Range	Mean	Standard Deviation	Range	Significance
Aerial diving piscivores	10.1	4.6	1-19	12.9	5.3	4-25	P<0.05
Pursuit swimming piscivores	12.9	13.4	0-66	16.1	17.9	2-75	Not significant
Wading piscivores	16.0	14.4	1-60	23.1	10.4	6-48	P<0.01
Resident waders	8.3	9.1	0-50	13.6	6.7	2-27	P<0.01
Migratory waders	4.8	10.0	0-50	4.5	8.6	0-40	Not significant
Herbivores	3.5	5.3	0-24	3.1	2.5	0-9	Not significant

Wading piscivores and invertebrate feeding waders preferred the large expanse of shallow water in Region 2 throughout the year. Pursuit swimming piscivores (cormorants, grebes, darters) also occurred in greatest numbers in this region, as it has a wide, deep channel and suitable roosting sites. Overall, aerial divers occurred relatively evenly along the length of the estuary, but kingfishers and African Fish Eagles preferred the upper reaches of the estuary, while terns and gulls were most common in the lower two regions. In general, birds do not aggregate near the mouth in this estuary but instead prefer the region above the bridge where there is less development and human disturbance, together with a large area of shallow water. There was no noteworthy change in overall distribution of birds between open and closed estuary mouth phases.

Correlations with water level

There was a significant positive correlation (r^2 = 0.13; $P<0.05$) between water level and the number of pursuit swimming piscivores recorded under closed estuary mouth conditions (Figure 4). Wading piscivore numbers decreased with increasing water levels (r^2 = 0.16; $P<0.001$; Figure 5). Non-migrating invertebrate feeding waders were more abundant in the winter months (Mean = 13.3 ± 9.89) than in the winter months (Mean = 6.8 ± 4.87) and their numbers decreased with increasing water levels (Figure 6). When separating the data by season a significant correlation (r^2 = 0.20; $P<0.01$) was found between water level and number of recorded individuals in the winter months. The correlation between water level and abundance was not significant in the winter months. Aerial diving piscivore numbers decreased significantly with increasing water levels (r^2 = 0.19, $P<0.001$; Figure 7). Migrating wader numbers did not show any significant correlations with water level. This is probably because they tended to visit the estuary for short periods and there was little site-faithfulness.

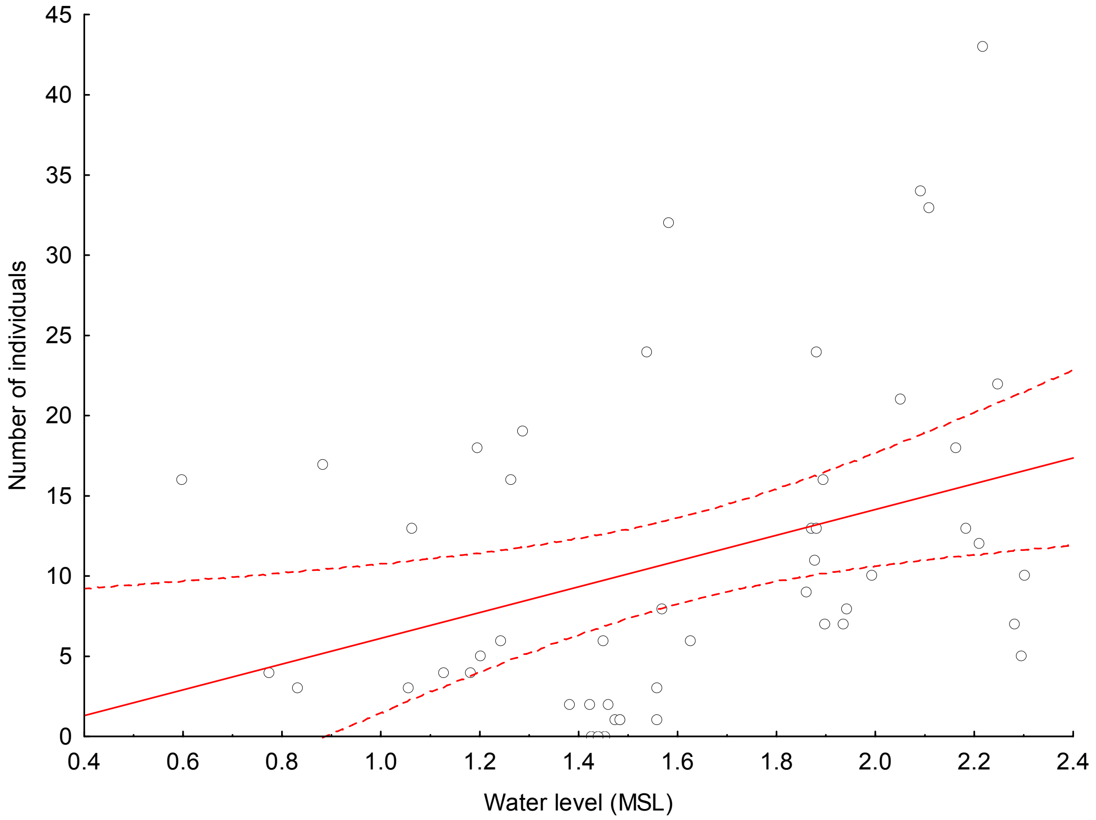

Figure 4. Pursuit swimming piscivore numbers recorded at different water depths under closed mouth conditions in the East Kleinemonde Estuary between March 2005 and November 2006.

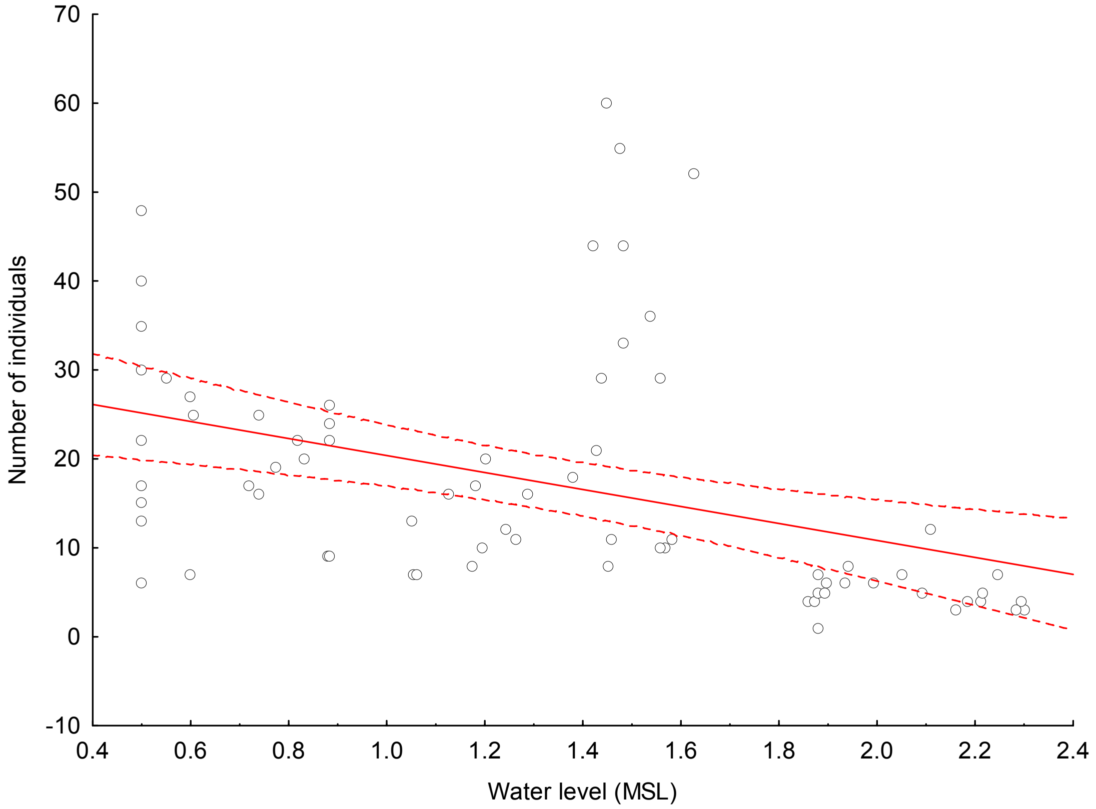

Figure 5. Wading piscivore numbers recorded at different water levels in the East Kleinemonde Estuary between March 2005 and November 2006.

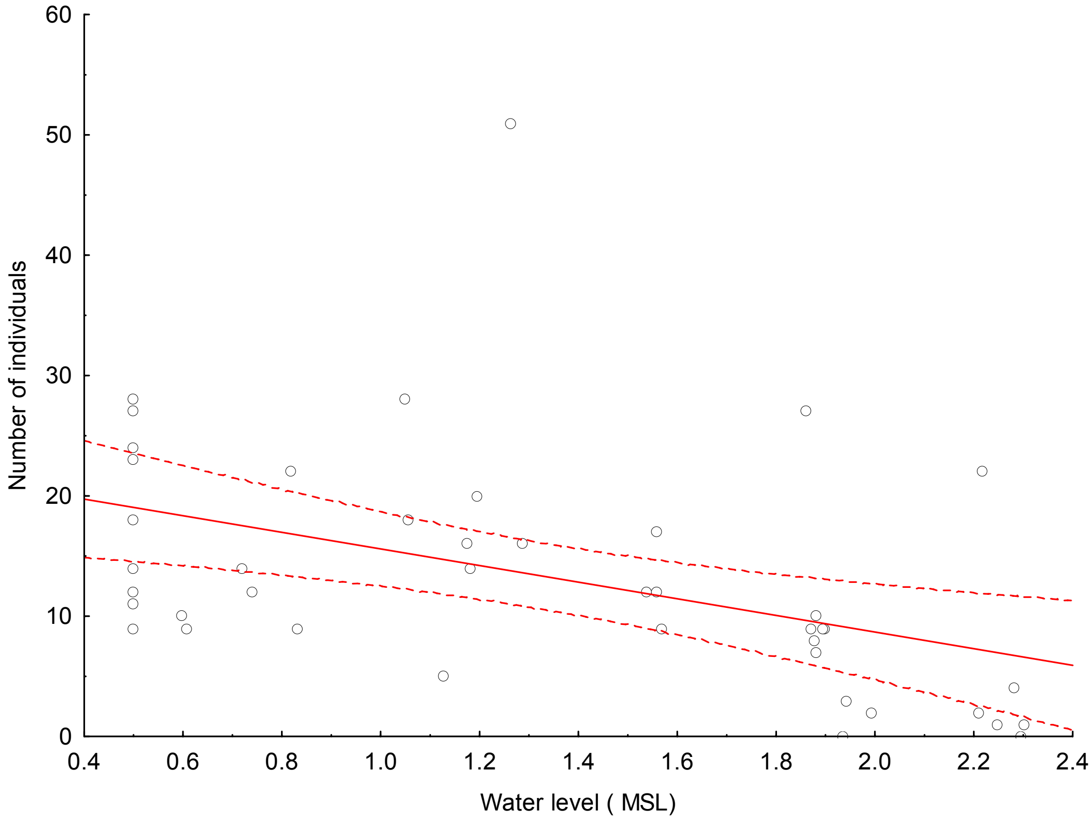

Figure 6. Non-migrating invertebrate-feeding wader numbers recorded at different water levels during the winter months (April-September) in the East Kleinemonde Estuary (2005-2006).

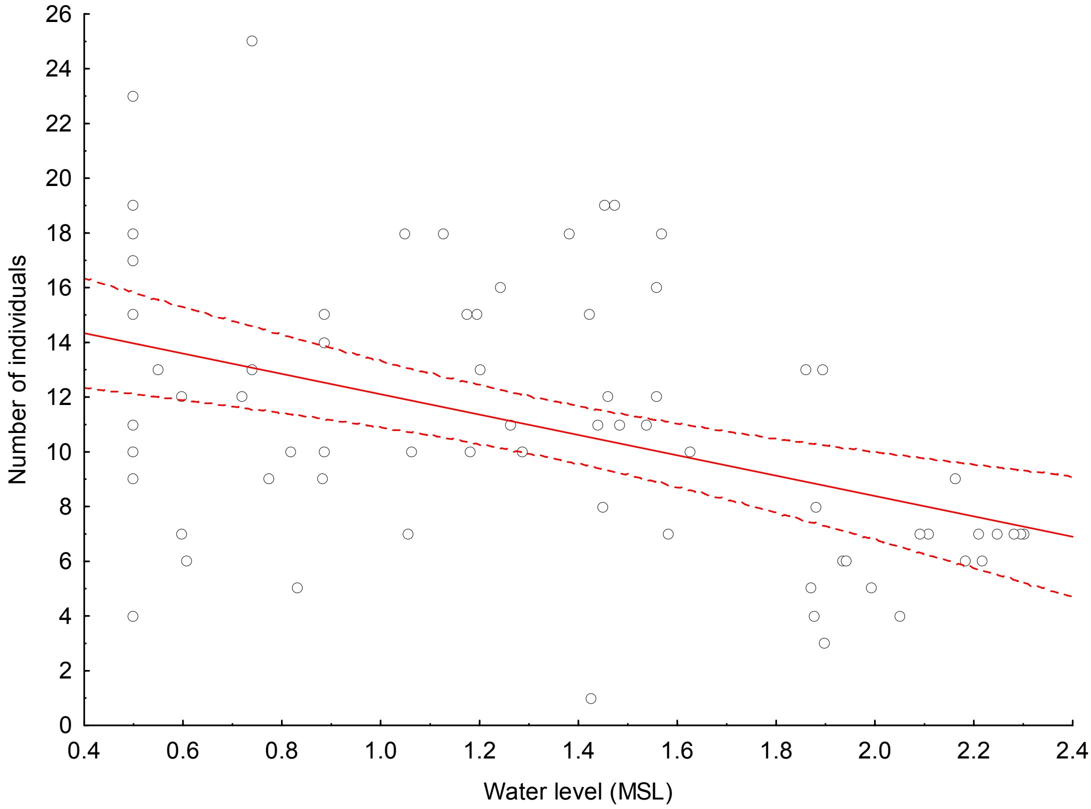

Figure 7. Aerial diving piscivore numbers recorded at different water levels in the East Kleinemonde Estuary between March 2005 and November 2006.

FACTORS INFLUENCING AVIFAUNAL COMMUNITY STRUCTURE AND ABUNDANCE

As discussed in Turpie (2000), it is difficult to accurately predict the impact of changes in estuarine characteristics on bird assemblages (Hockey & Turpie 1999a, 1999b) and predictions often have to be made on the basis of qualitative assessments of the relationships between the main variables that influence bird community structure and abundance in estuaries (Figure 8). These relationships may vary seasonally, from estuary to estuary, or between biogeographical zones. Difficulties in prediction are also compounded by variability due to external factors such as breeding success on distant breeding grounds. Very few quantitative studies have been made of the influence of abiotic and biotic factors on bird community structure and abundance in South African estuaries.

A complex array of variables is expected to influence the bird community (Figure 8, Hockey & Turpie 1999a, 1999b). Apart from external influences, different trophic groups of birds are probably influenced primarily by the availability (or catchability) of food (plants, invertebrates or fish), in turn influenced by its abundance and size class distribution. In addition to the relationship between food groups, the availability of food is in turn expected to be influenced by salinity, nutrients and relative availability of different habitat types (e.g. mudflats, sandflats, vegetated habitats). The latter variables are influenced by freshwater inputs to the estuary. Certain groups or species are liable to be more responsive to changes in system variables than others, depending on their ability to adapt to a range of circumstances (e.g. Turpie & Hockey 1997).

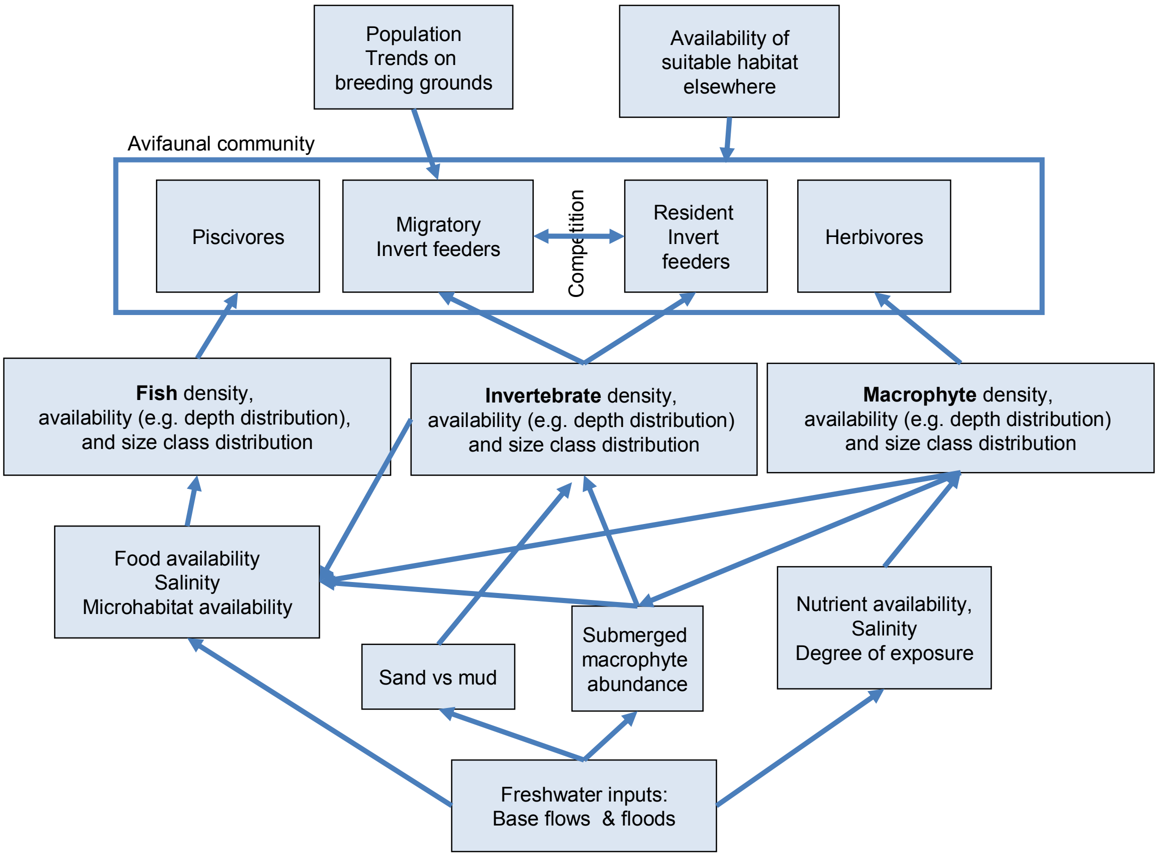

Figure 8. A simplified example of the types of relationships used in predicting the structure of an estuarine avifaunal community. Assumptions are made about the nature of relationships for each of the arrows depicted (from Turpie 2000).

Figure 8 depicts the trophic interactions within the estuary. Other variables, such as availability of roosting, perching or breeding sites, may feature strongly in determining the presence and abundance of certain species.

Where the composition and productivity of a food group is determined by abiotic factors such as salinity or sediment particle size, these variables may indirectly influence the nature of the avifaunal community. For example, a broad assumption applied to invertebrate feeding waders is that wader densities are negatively correlated with sediment sand fraction, because the latter is negatively correlated with invertebrate density/availability.

The above relationships have largely been considered for permanently open estuaries. In the case of systems such as the East Kleinemonde, it appears that water level is a major factor determining avifaunal abundance, since it affects habitat availability (e.g. deep channels for diving, shallow areas for wading) and food availability directly (e.g. sandprawn density; Terörde 2005b) or indirectly (e.g. through affecting availability of submerged macrophyte beds suitable for certain fish species).

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Appendix L

Proposed changes to RDM methodology for estuaries

The following recommendations were made for future RDM studies:

- TOCEs with small catchments react at relatively short time scales to changes in river inflow. The RDM methods currently analyse changes in runoff on a monthly time step. Daily flows may be needed to determine the ecological flow requirements of smaller systems to quantify the high degree of variability in small systems.

- Recent studies on epiphytes indicate that they are an important component of the estuarine ecosystem. The RDM methods need to be reviewed in the light of these findings and a decision made on the inclusion of epiphytes in future studies.

- Similar to water quality, future RDM studies should also measure the end-points for microalgae (i.e. marine algae) to understand the fluctuations in biomass in the estuary.

- The guidelines for the **Importance Rating** of an estuary are ambiguous and need to be refined:

    - Should the evaluation be on a national or regional scale?

    - Refine guidelines to capture the needs of rare and endangered species (e.g. estuarine pipefish).

- A number of changes are recommended to the wording of the **Functional Importance** scoring system to assist in guiding workshop participants**:**

    a) Export of organic material generated in the estuary (regional scale).
    b) Nursery function for fish and crustaceans (marine/riverine).
    c) Movement corridor for river invertebrates and fish breeding in sea.
    d) Roosting area for marine or coastal birds.
    e) Catchment detritus, nutrients and sediments to sea.