# DYNAMICS OF MACROPHYTES IN THE EAST KLEINEMONDE, A SMALL TEMPORARILY OPEN/CLOSED SOUTH AFRICAN ESTUARY

By

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# DECLARATION

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# SUMMARY

The East Kleinemonde Estuary is one of 175 temporarily open/closed estuaries (TOCEs) that represent 70 % of estuaries in South Africa. TOCEs are small (mostly less than 100 ha), shallow estuaries (average depth < 2 m) that respond quickly to freshwater inflow events. Their connection to the sea can be highly variable resulting in considerable changes in abiotic and biotic conditions. Mouth status depends on a balance between freshwater inflow and marine influence, which in turn affects ambient abiotic conditions. The objective of the study was to identify the abiotic variables which influence macrophyte growth and habitat availability. It was hypothesised that water level and salinity were the two main drivers of macrophyte change and macrophyte habitat would respond very rapidly, in less than a month, when habitat was available. Macrophyte habitats would also have high sediment seed reserves to ensure persistence under highly variable abiotic conditions. Macrophyte cover was monitored monthly in the East Kleinemonde Estuary along three permanent transects. The dominant habitats were submerged macrophytes, intertidal salt marsh, supratidal salt marsh, reeds and sedges. The following abiotic variables; water level, water column salinity, water temperature, Secchi depth, air temperature and rainfall were also measured between March 2006 and January 2010. Time-lag responses of the macrophytes to water level and salinity changes up to four months prior to the sampling session were also assessed. The analysis of a one year dataset highlighted only water level as a driver of change in macrophyte cover, whereas the five year dataset identified salinity as an additional important abiotic driver. This is because during September 2008 to January 2010 a series of large marine overwash events maintained high salinity (> 30 ppt) and high water level (> 1.6 m amsl) in the estuary. Water level increased by up to 0.33 m due to large volumetric changes and salinity was significantly higher in the 16 month closed euhaline phase after the breach (31 ± 0.9 ppt) compared to 21.9 ± 0.9 ppt in the closed polyhaline phase before the September 2008 breach. This increase in salinity significantly reduced the cover of the submerged macrophytes *Ruppia cirrhosa* and *Chara vulgaris*. They were replaced by macroalgae during this high salinity phase. The cover of supratidal salt marsh and reed habitats was also significantly reduced during the high water level phase, which in turn would lead to the potential for bank destabilisation and erosion. Based on the average elevation above sea level position of the macrophytes in the East Kleinemonde

Estuary, a threshold water level was identified as 1.55 amsl. This was taken to be the height above sea level at which there was a maximum cover change for each macrophyte habitat. Above this water level emergent macrophyte habitat would mainly be inundated. This, together with 30 ppt salinity, was identified as the two thresholds for macrophyte change in the East Kleinemonde Estuary. From these thresholds and the 5 year dataset four biotic states were identified as State A: open and tidal, State B: closed with a water level below 1.55 m amsl and salinity between 18 to 30 ppt, State C: closed and water level above 1.55 m amsl and salinity between 18 to 30 ppt and State D: closed and water level above 1.55 m amsl and salinity above 30 ppt. Intertidal salt marsh, reeds and sedges were dominant during the open phase. Submerged macrophytes were dominant during the closed polyhaline state and macroalgae during the closed euhaline state.

The high variability of abiotic factors common in TOCEs and the response of macrophyte habitat indicated that macrophytes were resilient to changing states provided they were of relatively short (< 3 months) duration. Macrophytes in the East Kleinemonde Estuary were found to have fast growth rates and large seed reserves in the sediment. The seed banks in the East Kleinemonde, as well as the adjacent temporarily open/closed West Kleinemonde Estuary were quantified for the first time in a South African estuary. The averaged data from both estuaries showed that Charophyte öospores represented almost 72 % of the sexual propagules in the sediment with a mean öospore density of 31 306 ± 2 293 $m^{-2}$. This was despite the Charophytes being sparsely located and only representing a maximum of 32.5 % cover in the above ground vegetation. Historically there must have been stands of Charophytes in the East Kleinemonde Estuary, such that öospores could accumulate to such high density found in this study. The second highest seed density was for the intertidal salt marsh plant *Sarcocornia tegetaria* (18 %) (7 929 ± 688 seed $m^{-2}$), followed by the submerged angiosperm *Ruppia cirrhosa* (7 %) (2 852 ± 327 seeds $m^{-2}$). Although seed density did not differ significantly with sediment depth, seeds still occurred at 20 cm below the surface of the sediment providing a regeneration source in the event of sediment scouring during a flood event. Germination studies in the greenhouse showed that most seeds were viable and *Sarcocornia tegetaria* began to germinate after 3 days to a maximum of 82 % after 91 days. Submerged species only germinated after 18 days with a low maximum germination of between 11 and 15 %.

This study has made an original contribution to the field of knowledge on macrophyte responses in a small TOCE as it showed that macrophyte habitats in the East Kleinemonde Estuary have a high natural variability in cover over time, they respond quickly after a disturbance event such as a mouth breach and there are large sediment seed reserves that remain viable from 2 to more than 5 years. This ensures habitat persistence even under unfavourable conditions, such as prolonged periods of mouth closure with high water level and flooding which causes loss of salt marsh species.

Given this natural variability it is necessary to predict responses both spatially and temporally in order to manage and maintain ecological functioning in TOCEs. This study identified dominant macrophyte habitat for different abiotic states through the use of water level and salinity thresholds. In the determination of the freshwater requirements of any South African estuary freshwater inflow rates are provided for each estuary's past, present and possible future freshwater inflow scenarios. These flow data are generated by hydrological models and simulated monthly inflow volumes for a period of about 72 years are provided. For the East Kleinemonde freshwater requirement study for any year in that 70-odd year period, the number of high flow and low flow mouth breaches were predicted, as well as the closed state periods. The threshold water level of 1.55 m amsl was also used to filter past, present and future inflow monthly volumes to determine the frequency of the four abiotic states identified in this study. It was based on a water level/water volume equation calculation from a digital elevation model. Results showed that the total closed period in the present state was 83 %, made up of 48 % of the time in a polyhaline state (State C) and 35 % in a euahaline state (State D). A second method was used to quantify available spatial habitat under different water level scenarios. A spatial model was written in Model Builder, an application in ArcGIS that allowed a series of processes to be built. A habitat map was overlaid with a bathymetric map and by selecting water level, available habitat areas were determined and empirical equations of water level versus available habitat were produced. These equations were then used to calculate the available habitat areas for monthly water level conditions from the freshwater requirement study for the past, present and two future inflow scenarios.

Using both the threshold water level method and the spatial availability model method it was possible to assess the effect of the two future inflow scenarios on macrophyte habitat response. Scenario 1 had a 16 % reduction in mean annual runoff (MAR) generating low flows for 88.6 % of the time and a 3.5 % reduction in flood events. In Scenario 2 there would be a 12 % reduction in MAR with low flows occurring for 87.5 % of the year, a 5.3 % reduction in floods and an 11.5 % reduction in the open mouth state. The model showed that Scenario 1 would have the highest submerged macrophyte area (12.56 ha versus 12.48 ha in Scenario 2), whereas Scenario 2 produced the largest mudflat and intertidal salt marsh area (7.11 ha versus 7.34 ha) due to lower water level in conjunction with the bathymetry of the estuary.

A reduction in freshwater inflow to TOCEs either due to anthropogenic influences or natural precipitation cycles is one of the main threats to the optimum functioning of these estuaries. The results from this study and the two methods of assessing the effect of freshwater inflow scenarios on macrophytes in TOCEs can be integrated into the current freshwater inflow assessment methodology in South Africa, as well as adding to our understanding of the ecological functioning of these small, highly variable estuaries. The methods provide a quick assessment of macrophyte habitat associated with abiotic states under past, present and future inflow scenarios. All that is required to predict macrophyte habitat for different freshwater inflow scenarios (present, past and future) is a habitat map, a bathymetric map and the elevation range of macrophytes in the TOCE being assessed. This, together with the knowledge of response rates, provides invaluable information for the management of TOCEs to maintain their ecological functioning under altered freshwater inflow regimes.

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# CHAPTER 1

# GENERAL INTRODUCTION

## *1.1 Background*

Temporarily open/closed estuaries (TOCEs) are estuaries that are blocked off from the sea for varying lengths of time by a sand bar that forms at the mouth (Whitfield, 1992). TOCEs are well represented throughout the world and in South Africa they represent 70 % of all 255 ecologically functioning estuaries (Whitfield and Bate, 2007). This type of estuary is also found in South America (Bonilla *et al*., 2005), parts of North America (Elwany *et al*., 2003), the Mediterranean (Largier *et al*., 1997), India and Sri Lanka (Ranasinghe and Pattiaratchi, 2003). Internationally they are also known as intermittently closed and open lakes and lagoons (ICOLLS), a term mainly used in Australia (Griffiths, 2001; Roy *et al*., 2001; Ryan *et al*., 2003), bar built estuaries, perched estuaries or blind estuaries (Tagliapietra *et al*., 2009). TOCEs have relatively small river catchment areas (< 500 $km^2$) and most South African TOCEs have catchments less than 100 $km^2$ (Whitfield, 1992).

TOCEs are known to fluctuate between different states that depend on a combination of factors such as catchment size, river inflow, evaporation, groundwater and berm seepage, adjacent ocean wave energy, marine current, long shore sediment transport, berm size and marine overwash events (Reddering, 1988; Cooper, 2001; Ranasinghe and Pattiaratchi, 2003; Smakthin, 2004; Hadwen and Arthington, 2006; Rustomji, 2007; Whitfield *et al*., 2008). Based on abiotic variables, TOCEs in South Africa occur in three states, namely open, semi-closed and a closed mouth state (Snow and Taljaard, 2007). During the open state, river inflow to the estuary is sufficiently high to keep the mouth open to the sea and tidal conditions prevail. Longitudinal salinity gradients occur. The open state can last for a few days to a few weeks depending on the minimum freshwater inflow required to maintain an open mouth state. In perched estuaries the base level is above mean sea level and closure usually occurs within days of the mouth being breached (Whitfield, 1992; Stretch and Parkinson, 2005). Most TOCEs along the north eastern coast of South Africa are perched and they are often dominated by fresh to mesohaline conditions. Mesohaline is defined as between 5 to 18 ppt, polyhaline as 18 to 30 ppt and euhaline as 30 to 40 ppt. Breaching is usually seasonal in response to rainfall events (Cooper, 2001). During the semi-closed state low river inflow and an increase in berm height limits seawater intrusion to spring high tides only. The berm is not high enough to prevent water draining from the estuary into the sea (Snow and Taljaard, 2007). Vertical salinity stratification can occur due to the low freshwater input. In the closed state there is extremely low or no river inflow and a gradual increase in the height of the berm prevents seawater from entering the estuary or water draining from the estuary into the sea. During this closed state marine overwash into the estuary can occur, depending on berm height and conditions at sea. Overtopping from the estuary into the sea can also occur if the water level exceeds the berm height. Marine overwash events cause localised high salinity near the mouth but is usually diluted after a few weeks due to direct freshwater inflow or groundwater seepage (Gama *et al*., 2005; Snow and Taljaard, 2007). However, generally in the closed state horizontal and vertical gradients in abiotic variables are absent or are poorly developed (Lukey *et al*., 2006; Snow and Taljaard, 2007; Henninger *et al*., 2009; Whitfield *et al*., 2008). This is because TOCEs generally have an average depth of less than 2 m, wave height is small and wind mixing is common. In the Mtamvuna Estuary along the east coast of South Africa, depth exceeds 2 m and vertical salinity stratification has been recorded (Perissinotto *et al*., 2010). During the closed state salinity can change from polyhaline after mouth closure to mesohaline, or to oligohaline in many of the KwaZulu-Natal TOCEs. However salinity can also range from being fresh immediately after floods to periods of hypersalinity (Whitfield, 1992; Healy, 1997; Largier *et al*., 1997; Bachelet *et al*., 2000). Hypersalinity has been well documented for many Australian and Mediterranean TOCEs (Geddes and Butler, 1984; McComb and Humphries, 1992; Largier *et al*., 1997; Bachelet *et al*., 2000; Dalton *et al*., 2002; Young and Potter, 2002; Barton and Sherwood, 2004; Hastie and Smith, 2006; Deeley and Paling, 2008; Pope, 2008), UK estuaries (Healy, 1997), as well as in African estuaries (Whitfield, 1989; Taylor *et al*., 2006; Lamptey and Armah, 2008). These hypersaline states can occur for varying lengths of time and are linked to annual or decadal climatic variability. Cyclic periods of open and closed states usually occur in areas with seasonal rainfall patterns where the closed state of the estuary coincides with a period of low river inflow while the open state generally coincides with high rates of river inflow and flood events (Cooper *et al*., 1990; Cooper *et al*., 1999; Cooper, 2001; Smakthin, 2004; Van Niekerk *et al*., 2008). This is because estuaries with small catchments less than 100 $km^2$ respond rapidly to rainfall events

(Smakthin, 2004). TOCEs in the Western Cape have seasonal open states during winter because they lie in a winter rainfall area, whereas along the eastern coast (KwaZulu-Natal) the open state is usually during the summer rainfall period. By contrast, estuaries in areas of low rainfall and high evaporation rate, or during periods of severe droughts, experience salinity that may exceed 40 ppt (Perissinotto *et al*., 2010). In the Seekoei Estuary in the Eastern Cape salinity up to 98 ppt resulted in fish kills (Whitfield and Bruton, 1989; Whitfield, 1995). Recently in the Kasouga Estuary, there was an 80 % loss of submerged macrophytes (mainly *Ruppia cirrhosa* (Pentagna) Grande.) due to hypersaline conditions that rose to 40 ppt. This hypersaline event was also brought on by below average rainfall, high evaporation and a number of marine overwash events (Froneman and Henninger, 2009). Cycles of hypersalinity in the St Lucia Estuary occur about every 10 years due to low rainfall (Taylor *et al*., 2006). During "normal" wet periods there is a rise in estuary water level and a net outflow of water from the estuary to the sea. The northern parts of the lake can eventually become almost fresh, with an increasing salinity towards the mouth, where tidal water flows in from the Indian Ocean. Submerged macrophytes are limited to the lake areas when the estuary is tidal. In the southern tidal reaches, mangroves and salt marsh are common. During droughts, the inner part becomes more saline, water level in the estuary drops and there is a net inflow of water from the sea into the estuary. Salinity concentrations above 120 ppt have been measured in the northern parts of the estuary (Taylor, 1991). Submerged macrophytes are lost if salinity exceeds 65 ppt.

The traditional definition of an estuary according to Day (1981) is "a partially enclosed body of water which is either permanently or periodically open to the sea and within which there is a measurable variation of salinity due to the mixture of the sea water with fresh water derived from land drainage". Although this definition recognises that some estuaries can become separated from the sea for periods of time due to the formation of a sand bar across their mouth, researchers drew attention to the fact that estuaries may become hypersaline for long closed periods following very low fresh water inflow, high evaporation and marine overwash (Hadwen and Arthington, 2006). As a result this definition was revised by Potter *et al*., (2010) to "An estuary is a partially enclosed coastal body of water that is either permanently or periodically open to the sea and which receives at least periodic discharge from a river(s), and thus, while its salinity is typically less than that of natural sea water and varies temporally along its length, it can become hypersaline in regions when evaporative water loss is high and tidal input is negligible" This definition has not been adopted yet in South Africa and the definition of Day (1981) is still used, for example in the National Water Act.

There are nine functional macrophyte community types associated with estuaries; supratidal salt marsh, intertidal salt marsh, reeds and sedges, mangroves, intertidal and subtidal benthic microalgae, phytoplankton, macroalgae and swamp forest (Coetzee *et al*., 1996). Macrophytes associated with TOCEs in South Africa are macroalgae, submerged macrophytes, intertidal salt marsh, supratidal salt marsh and reeds and sedges (Coetzee *et al*., 1996). Swamp forest (represented by the freshwater swamp tree *Barringtonia racemosa* and lagoon hibiscus, *Hibiscus tiliaceus*) only occurs in South African TOCEs along the eastern coast from Mngazana Estuary (31º 41' S, 29º 25' E) northwards, because of the sub-tropical climate and higher rainfall (Adams *et al*., 1999). Because there is only an intermittent connection with the sea, mangroves are absent from TOCEs. Estuarine macrophytes provide a number of important functions, from sediment stabilisation, to improving water quality (Dekker *et al*., 2005; Estevez *et al*., 2008; Obrador and Petrus, 2010). They are a large source of primary production; they provide faunal habitats, food sources and are important as nursery and refuge areas (Able, 2005; Veiga *et al*., 2006; Becker and Laurenson, 2008). When habitats change as a result of macrophyte disappearance the result is a reduction in abundance and even loss of associated biota. The macrophyte habitats and dominant species considered in the East Kleinemonde study were the submerged macrophytes (*Ruppia cirrhosa* (Pentagna) Grande, *Stukenia pectinata* L. and *Chara* spp.) and emergent habitats which consisted of supratidal salt marsh (*Sporobolus virginicus* (L.) Kunth, *Juncus kraussii* Hochst, *Sarcocornia decumbens* (Tölken) A.J. Scott, *Stenotaphrum secundatum* (Walt.) O. Kuntze, *Limonium scabrum* (Thunb.) Kuntze), intertidal salt marsh (*Sarcocornia tegetaria* (L.) A.J. Scott, *Salicornia meyeriana* (Moss.)) and reeds and sedges (*Phragmites australis* (Cavinelles) Trinius ex Steudel, *Bolboschoenus maritimus* (L.) Pallas). Although *Juncus kraussii* is an intertidal species in other South Africa estuaries, it occupied a supratidal position in the East Kleinemonde Estuary. *Sporobolus virginicus* also occupied both supratidal and intertidal habitat in the East Kleinemonde although it is traditionally considered a supratidal species.

The large abiotic variability in TOCEs can lead to a shift in the dominant primary producers, for example from a submerged macrophyte dominated clear water state to a turbid water state dominated by phytoplankton (Pope, 2008; Rosqvist *et al*., 2010). During the closed phase of a TOCE extensive submerged macrophyte beds of *Ruppia cirrhosa* and *Stukenia pectinata* can develop depending on the salinity regime. If salinity falls below 15 ppt for an extended period of time (> 6 months) then *Ruppia cirrhosa* is replaced by *Stukenia pectinata* (Howard-Willliams and Liptrot, 1980; Adams and Bate, 1994). Since mouth state in some TOCEs is not seasonal, estuarine macrophytes must be able to respond rapidly and be resilient and persistent under changing abiotic states. Resilience is defined here as "*the ability of an ecosystem to withstand change while still retaining function and structure, without collapsing into a different state with a different set of processes*" (Gunderson *et al*., 2010). Persistence is defined as "*the ability of a population to return after a disturbance and/or regime shift without extinction occurring*". This means macrophytes in TOCEs must establish and grow quickly and have adequate seed reserves to endure adverse abiotic conditions, similar to species in ephemeral systems such as lakes and pans (Arendt, 1997).

The East Kleinemonde Estuary is a small temporarily open/closed estuary that lies on the south east coast of South Africa, almost midway between Port Elizabeth and East London (33°32′ S; 27°03′ E). The estuary is approximately 3.7 km in length with a maximum surface area of 35.7 ha and a catchment area of 43.5 $km^2$ (Badenhorst, 1988). The estuary has a water depth of between 1 to 2 m, except when the mouth of the estuary has been closed for an extended period of time and the sand berm has built up. Water level can then reach 2.5 m amsl. The estuary breaches on average 2.6 times a year (Van Niekerk *et al*., 2008). Mouth breaching is erratic and unpredictable and is usually in response to rainfall events above 100 mm (Cowley and Whitfield, 2001). Base flow into the estuary is not large enough to keep the mouth open and it usually closes within 24 hours. Only when river inflow is relatively high (> 0.04 $m^3$ $s^{-1}$) does the mouth stay open while the flow persists (Van Niekerk *et al*., 2008). Although rainfall may occur at any time of the year, long-term records demonstrate an autumn-spring bimodal pattern with a spring peak (Kopke, 1988, in James, 2007). Analysis of historical rainfall patterns shows that there also appears to be a 1:30 year flood cycle in the geographical area (Van Niekerk *et al*., 2008), although Jury and Levy (1993) state there are drought cycles that occur every 3.45 to 18.2 years for the Eastern Cape (Figure 1.1). Flood and freshette events have been suggested to be the drivers of abiotic changes in the East Kleinemonde Estuary with baseflow only influencing the system to a lesser extent. These flood events reset the estuary by lowering the base level due to scouring out of large quantities of sediment and the open mouth state is maintained for a longer period until the berm begins to build up and the mouth closes.

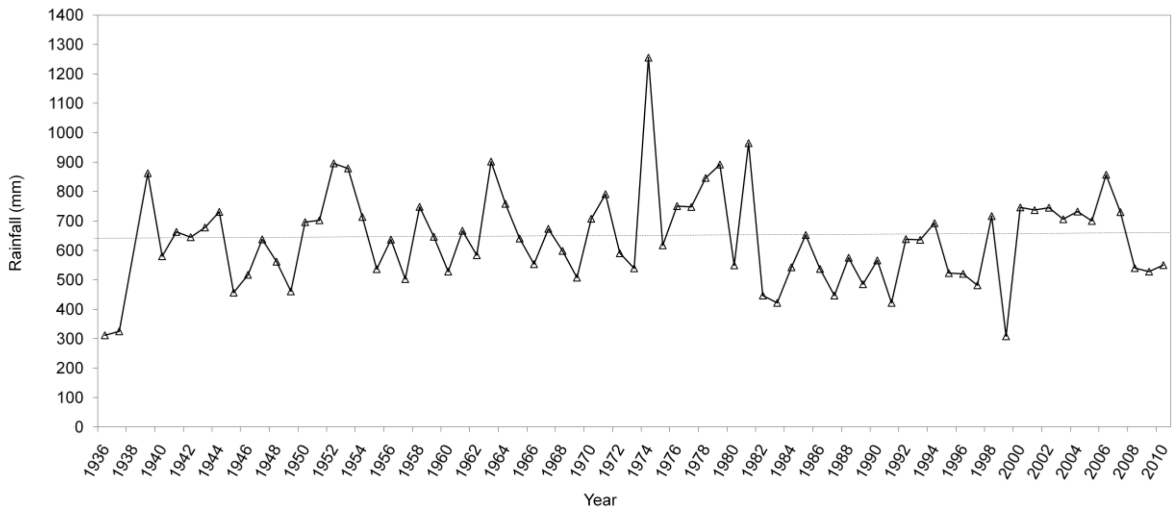


Figure 1.1. Historical rainfall records for the East Kleinemonde Estuary. Data obtained from South African Weather Bureau. The dotted line represents the average rainfall for 1936 to 2010.

During 2005/2006 an intensive multidisciplinary research programme was conducted on the East Kleinemonde Estuary as part of a Water Research Commission research study on the freshwater requirements of warm-temperate TOCEs (van Niekerk *et al.*, 2008). Aspects studied ranged from hydrodynamics, sediment dynamics, nutrients, microalgae, macrophytes, zoobenthos, hyperbenthos, zooplankton, ichthyoplankton to fish and birds. The findings of these can be found in Van Niekerk *et al*. (2008). The freshwater requirement of any estuary in South Africa is assessed by means of the Resource Directed Measures Programme (RDM) of the South African Department of Water Affairs and forms part of the National Water Act (No. 36 of 1998). The RDM procedure assesses the amount of freshwater required to maintain the estuary in an acceptable ecological condition. It is a holistic approach that includes all the abiotic and biotic parameters associated with an estuary. The method gives an indication of how a system differs from its natural state in its present condition and under different future inflow scenarios. Abiotic and biotic parameters are determined for the present state or condition of the estuary, as well as for a reference or natural state. The reference state is based on hydrological models that simulate monthly inflows volumes for a period of up to 72 years. The description of the present state, together with the reference condition, forms the basis for the preliminary Determination of the Ecological Water Requirement study (Taljaard *et al*., 2004). Specialist scientists describe and document their understanding of the characteristics and functioning of an estuary (backed by appropriate field measurements and scientific expertise). The East Kleinemonde Estuary has long term data on mouth state dating back to 1993 (Cowley *et al*., unpublished data). These data together with other physical data captured in 2005/2006 formed the basis of the RDM study for the East Kleinemonde Estuary. In the East Kleinemonde Estuary three abiotic states were recognised based on present and simulated freshwater inflow volumes. A simple basin model was developed in which river inflows into the estuary were accumulated to estimate the volume in the system (Van Niekerk *et al*., 2008). The volume, in turn was used to evaluate probable mouth conditions and the salinity regime of the system. Based on these volumes the abiotic states were:

State 1 Intermittently open/closed driven by high flow events > 0.3 x $10^6$ $m^3$

State 2 Intermittently open/closed driven by persistent low flow periods < 0.3 x $10^6$ $m^3$ and cumulative inflows > 0.3 x $10^6$ $m^3$

State 3 Closed mouth with flow volume < 0.3 x $10^6$ $m^3$ and cumulative inflows < 0.3 x $10^6$ $m^3$.

For 78.4 % of the year the estuary is in State 3 and breaching occurs either due to a slow increase of water level which builds up a large volume of water that scours out the berm, or due to a freshet (> 0.3 x $10^6$ $m^3$) which would fill up the estuary and trigger a breach event. A storm event could also trigger a breach event due to increased marine influence. During State 1 the mouth can remain open for 1 to 28 days and there will be an initial flushing of the estuary and a strong marine influence. Salinity above 30 ppt can occur. For State 2 the mouth only remains open for one to three days and salinity is in the region of 15 ppt. During State 3 salinity can drop from > 25 ppt to 15 ppt if the estuary remains closed for long enough.

The RDM study also assessed estuary health for a number of future freshwater inflow scenarios. For each state, past, present and future, an Estuarine Health Index was calculated in which abiotic variables account for 50 % of the score (habitat health score) and biotic variables the other 50 % (biotic health score). Each of these is in turn made up of scores for aspects ranging from hydrodynamics to birds. RDM studies of estuaries can take place on three different levels; comprehensive, intermediate or rapid level of assessment. Rapid level studies rely on the available data and existing knowledge of estuarine processes and functions, whereas intermediate and comprehensive studies depend on the extent of data available or required. For example, in an intermediate RDM, at least a once-off survey of macrophytes is required, preferably during the open mouth phase, whereas for a comprehensive level, sampling of macrophytes is required during both an open and a stable closed phase. The RDM study for the East Kleinemonde Estuary was at an intermediate level based on the available information and expertise (Van Niekerk *et al*., 2008).

For the present, past and future states the response of biota are assessed based on the percentage change from the present state. For example if under a future inflow scenario mouth closure state is calculated to increase by 10 % from the present state, then macrophyte species richness, abundance and composition may change because intertidal salt marsh will be replaced with submerged macrophytes. Within the submerged macrophyte habitat there may be a species composition change from *Ruppia cirrhosa* to *Stukenia pectinata* as salinity decreases with closure. Scoring is based on expert understanding of biotic responses to abiotic variables but results are given as a percentage change from present state. No quantification of interactions is given. Turpie *et al*. (2008) attempted to quantify the interactions between mouth status and biota in the East Kleinemonde Estuary using Stella®. Stella® is a computer modeling package that allows users to construct dynamic models that realistically simulate biological systems. The East Kleinemonde model was made up of a series of sub-models, each of which dealt with a different physical, biotic, economic or management aspect of the estuary. A daily time-step was used to model simulation periods of up to four years. For the macrophyte sub model a number of assumptions were used based on species response to salinity changes, growth rates based on Riddin and Adams (2008) and germination inputs based on Riddin and Adams (2009). This systems model method provided an excellent platform for the integration of relevant socio-economic variables, which are important because management decisions increasingly need justification in economic terms, for example tourism value, recreation, aesthetics and property value of adjacent areas (Turpie *et al*., 2008).

The research presented in this thesis is unique as no other data set on monthly responses of macrophytes to abiotic changes is available. There is only a single long-term monitoring data-set on estuarine macrophytes in South Africa for the Great Brak Estuary. An assessment using fixed line transects was established in this estuary in 1989 and observations and counts were made regularly to determine the influence of reduced freshwater inflow on macrophytes following the construction of the Wolwedans Dam (Adams, 2008, in DWAF, 2008). Although results document the changes in spatial cover of salt marsh, no data exist on how fast these changes occurred. This is due to relatively infrequent monitoring. Rates of change are not always considered when managing mouth states in estuaries. In the Great Brak Estuary a mouth management plan which, providing water volume in the Wolwedans Dam is above a certain level, ensures that water is released during spring for ecological reasons. The released water serves to breach the mouth at a time correlated to the flowering periods of halophytes in the estuary, as well as for invertebrate and fish recruitment (CSIR, 1990; Ematek, 1992; Slinger *et al*., 2005; DWAF, 2008). This allows seed reserves in the sediment to be replenished. Data on how quickly macrophytes recover in response to disturbance are lacking.

Given this lack of information the objectives of this study were to:

1. determine the abiotic drivers of macrophyte habitat change in the East Kleinemonde Estuary,
2. quantify the rates of macrophyte habitat change,
3. quantify sediment seed banks in order to determine whether macrophyte habitats are resilient and can persist under the highly variable and unpredictable abiotic conditions associated with TOCEs,
4. identify the dominant macrophyte habitats associated with different abiotic states and the causes or thresholds responsible for shifting from one state to another and
5. model habitat availability under different water level scenarios.

The hypotheses were that water level and salinity were the two main drivers of habitat changes and that macrophytes respond by having rapid growth rates, recovering within a month or less, as well as having large seed reserves to ensure persistence of habitats under unpredictable abiotic disturbances.

Chapters 2, 3, and 4 have been published as scientific papers. A literature review of macrophytes in TOCEs was not included in this thesis but can be found in Whitfield and Bate (2007). Chapter 2 identified the abiotic drivers that resulted in macrophyte habitat change in the East Kleinemonde Estuary over a 13 month period (March 2006 to March 2007). Chapter 3 evaluated the effects of a series of marine overwash events on the abiotic and biotic components that led to a closed marine-dominated state. Chapter 4 provided detail on the sediment seed banks of the East Kleinemonde Estuary, as well as the neighbouring West Kleinemonde Estuary. Seed viability and germination rates in response to flooding and salinity were tested. The West Kleinemonde Estuary was included as a study site because the mouth had remained closed for longer (2 years) than the East Kleinemonde Estuary providing the opportunity to study the viability of macrophyte seed under these extended conditions. Chapter 5 assessed the drivers of macrophyte habitat change over the full study period (March 2006 to January 2010) and compared these data with those reported from the short-term study (Chapter 2). This comparison was to determine the presence (or absence) of specific states and the cause of change from one state to another. Long term changes and habitat availability were predicted in Chapter 5 for two freshwater inflow scenarios. Chapter 6 provided the final conclusions. Shortcomings of the study were identified and suggestions were made for further studies in order to understand and manage TOCEs better. Results from this research have been published in the following papers;

Riddin, T., Adams, J.B., 2008. Influence of mouth status and water level on the macrophytes in a small temporarily open/closed estuary. Estuarine and Coastal Shelf Science 79: 86-92.

Riddin, T., Adams, J.B., 2009. The seed banks of two temporarily open/closed estuaries in South Africa. Aquatic Botany 90: 328-332.

Riddin, T., Adams, J.B., 2010. The effect of a storm surge event on the macrophytes of a temporarily open/closed estuary, South Africa. Estuarine and Coastal Shelf Science 89: 119-123.

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# CHAPTER 2

# INFLUENCE OF MOUTH STATUS ON HABITAT DEVELOPMENT: SHORT TERM

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## *Abstract*

The monthly responses of macrophytes in the East Kleinemonde Estuary were examined in relation to changes in physical factors between March 2006 and March 2007. The East Kleinemonde is a small temporarily open/closed system where the mouth breaches in response to high water level (> 2 m amsl) or following high river inflow. On breaching there is a rapid drop in water level that causes the submerged macrophytes to be exposed and they die as a result of desiccation. Salt marsh habitat then establish in the vacant habitat. Correlation analysis showed that water level and duration of inundation influenced macrophyte cover abundance. Inundation for three months caused die back of intertidal salt marsh. Under open and tidal conditions, intertidal salt marsh increased at a maximum monthly expansion rate of 25 % change in cover. Supratidal salt marsh expanded at maximum monthly rates of 33 % change in cover. Because of its position at a relatively high elevation compared to other vegetation, supratidal salt marsh was only affected by water level of > 1.8 m amsl and only after being inundated for one to two months. Submerged macrophytes developed in inundated areas when stable water level was present for longer than two months at a monthly maximum expansion rate of 23% cover change. In this study macrophytes responded quickly to water level fluctuations and indicate that monthly monitoring is needed to provide an understanding of macrophyte response. This is the first study that reports on rates of macrophyte habitat development in temporarily open/closed estuaries. These data can be used in mouth management plans and freshwater

requirement studies to predict the growth and establishment of a diversity of macrophyte habitats.

Keywords:
Estuary; macrophytes; response; inundation; expansion; breaching

## *2.1 Introduction*

The East Kleinemonde Estuary is one of 175 temporarily open/closed estuaries (TOCEs) that occur along the South African coastline (Whitfield, 1992). TOCEs are small (< 100 $km^2$), shallow estuaries (< 2 m) that can either be predominantly open or closed (more or less than 50 % of the time respectively). Their connection to the sea can be highly variable resulting in considerable changes in the value of abiotic variables. In the East Kleinemonde Estuary, low or no river inflow results in the formation of a sand bar that can isolate the estuary from the sea for as long as two years (Cowley and Whitfield, 2001). Under these extended periods of mouth closure large beds of submerged macrophytes were observed to develop. The mouth breaches in response to high water level (> 2.3 m amsl) following increased river inflow, but the marine connection usually only remains in place for a few days. Under open conditions, the estuary can drain completely resulting in the loss of submerged macrophyte beds through exposure and desiccation. This has important ecological implications for associated biota that use these habitats are refugia and nursery areas. Over the past decade studies on TOCEs in South Africa and in similar systems worldwide have concentrated on these responses of faunal communities to changes in the physical variables as a result of water level fluctuations (Carruthers *et al*., 1999: Australia; Bell *et al*., 2000: South Africa; Cowley and Whitfield, 2002: South Africa; Perissinotto *et al*., 2003: South Africa; Froneman, 2004: South Africa; Dye and Baros, 2005: Australia; Jones and West, 2005: Australia; Nozias *et al*., 2005: South Africa; Gladstone *et al*., 2006: Australia; Hastie and Smith, 2006: Australia). Botanically, only a few studies have related changes in microalgal communities to mouth status and water level fluctuations (Gobler *et al*, 2005: USA; Skinner *et al*., 2006: South Africa; Anandraj *et al*., 2007: South Africa). There is a substantial database on large scale macrophyte distribution in relation to abiotic variables in lakes and big estuaries (Gafny and Gasith, 1999: Israel; Kunii and Minamoto, 2000: Japan; Havens *et al*., 2005: USA; Appelgren and Matilla, 2005: Baltic; Turner and Schwarz, 2006: New Zealand; Selig *et al*., 2007: Germany; Deegan *et al*., 2007: Australia). However, only a few studies have quantified or attempted to model the response of macrophyte cover to water level fluctuations in TOCEs (Healy, 1997: Ireland; Wortmann *et al*., 1998: South Africa; Gesti *et al*., 2005: Spain; Charpentier *et al*., 2005: France).

The South African National Water Act (Act 36 of 1998) requires sufficient water be delivered to all South African estuaries. This ensures that estuaries are maintained in a recommended ecological state and with a related ecological flow requirement. The ecological state is determined as one that is able to be managed as close as possible to the estuary's original state and faunal and floral dependence to the set flow is maintained.
For example, in the Great Brak Estuary in South Africa water is released from the Wolwedans Dam to keep the mouth open in spring/summer in order to maximize fish recruitment and stimulate salt marsh germination and growth (Adams *et al*., 1999).

The results of this study assist in determining flow requirements and water level scenarios in the East Kleinemonde Estuary by means of the following objectives, a) to determine the relationship between macrophyte cover, water level and environmental variables in a temporarily open/closed estuary and b) to quantify the temporal changes of macrophytes in the East Kleinemonde Estuary. Furthermore, the consequences of any alterations to freshwater input and thus water level in this estuary, either through water abstraction or artificial mouth manipulation, can easily be predicted through the results of this study.

## *2.2 Materials and Methods*

### *2.2.1 Description of Study site*

The East Kleinemonde Estuary lies on the south eastern coast of South Africa approximately 15 km north east of Port Alfred (33°32'S and 27°03' E) (Figure 2.1). The catchment of the estuary is estimated to be 43.5 km$^2$ with a surface area of 35.7 ha (Badenhorst, 1988). .

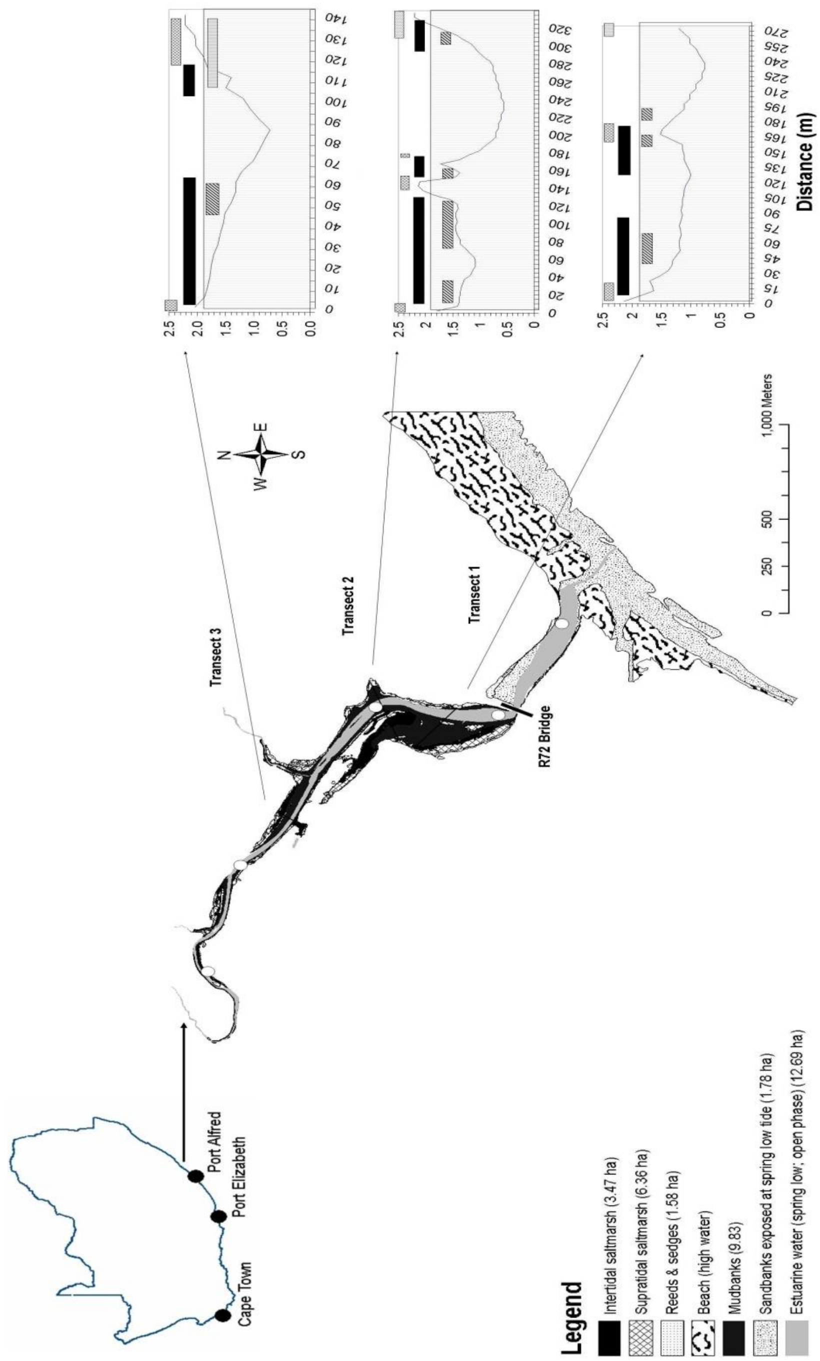


Figure 2.1. Study site showing the three transects, their elevation profiles, water sampling sites (○) and extent of macrophyte habitats. Habitat map taken from Riddin and Adams (2007).

The estuary is generally shallow with depths seldom exceeding 2 m; the deepest is 6 m that occurs below the R72 bridge. The estuary is approximately 4 km long and 250 m at its widest point. Large shallow mudflat area occurs upstream of the bridge. Farming and residential development is present in parts of the catchment and surrounding areas respectively.

### *2.2.2 Abiotic Variables*

Water column salinity (ppt), water column temperature (°C) and turbidity (NTU) were measured monthly *in situ* at five stations along the length of the estuary (Figure 2.1) using a YSI 650 MDS Multiprobe. Average values at the time of sampling were used in the calculations. Average daily water levels were obtained from a water level recorder installed beneath the R72 Bridge where water level is recorded every ten minutes. Elevation profiles for the three transects were measured using a Wild Heerbrugg Dumpy Level and referenced against MSL (Figure 2.1).

### *2.2.3 Macrophyte dynamics*

The change in macrophyte cover was assessed along three permanent transects on a monthly basis from March 2006 to March 2007 (Figure 2.1). Data from a quarterly sampling period in 2005 are also included. However, this quarterly sampling failed to capture the rapid changes observed under the monthly sampling regime. Species cover abundance (%) was measured within duplicate 1 $m^2$ quadrats that were placed every 5 m along the length of each transect. Only the quadrats in which each species occurred at the time of sampling, or in previous sampling trips, were used to calculate average species cover along each transect. This was felt to be a better method than averaging species cover over the whole transect as some habitats never form in many parts of a transect. Therefore only the maximum and minimum elevation extent was used for habitat data averaging. For example, the intertidal salt marsh habitat in the East Kleinemonde Estuary represented by *Sarcocornia tegetaria*, *Salicornia meyeriana* and *Sporobolus virginicus*, only occurred at elevations of between 0.71 to 1.8 m amsl during the sampling period. Cover data for quadrats occurring within this range were only used for averaging. The average cover abundance for the macrophytes in the three transects was used in correlation analysis against environmental variables. Expansion rates, expressed as average percentage change per month (within 1 $m^2$ quadrats), was calculated for each habitat, as well as for individual species.

### *2.2.4 Data analysis*

Nonparametric Spearman Rank Order Correlation was used to determine the relationship between average macrophyte cover and water level, salinity, temperature and turbidity for each sampling period. Because the health of the macrophytes at the time of sampling is a reflection of preceding water level and is not related to depth at time of sampling (Steinman *et al*., 2002), a one to three month water level time lag was also included for all water level analyses. These latter data were obtained from the water level recorder and values for one, two and three months prior to the sampling date were used.

## *2.3 Results*

### *2.3.1 Abiotic Variables*

During the sampling period the mouth was closed for a maximum period of only six months, i.e. between December 2005 and June 2006 (Figure 2.2). A flood event in August 2006 (195 mm) resulted in a series of rapid breaching and mouth closure events, which occurred over the following four months. This long and unusual period of open mouth condition significantly increased tidal amplitude from 0.4 m to 1.2 m ($P < 0.01$). The mouth closed in December 2006 and remained closed until 18 March 2007, at which time the estuary breached at a level of 1.9 m amsl after a monthly rainfall of 189 mm. Mouth closure generally occurred at water levels varying between 0.5 and 1 m amsl. During this study the maximum water level recorded was 2.3 m amsl. Studies have shown that mouth breaching events and number of days the mouth is open in the East Kleinemonde Estuary is directly related to river inflow (Van Niekerk *et al*., 2008).

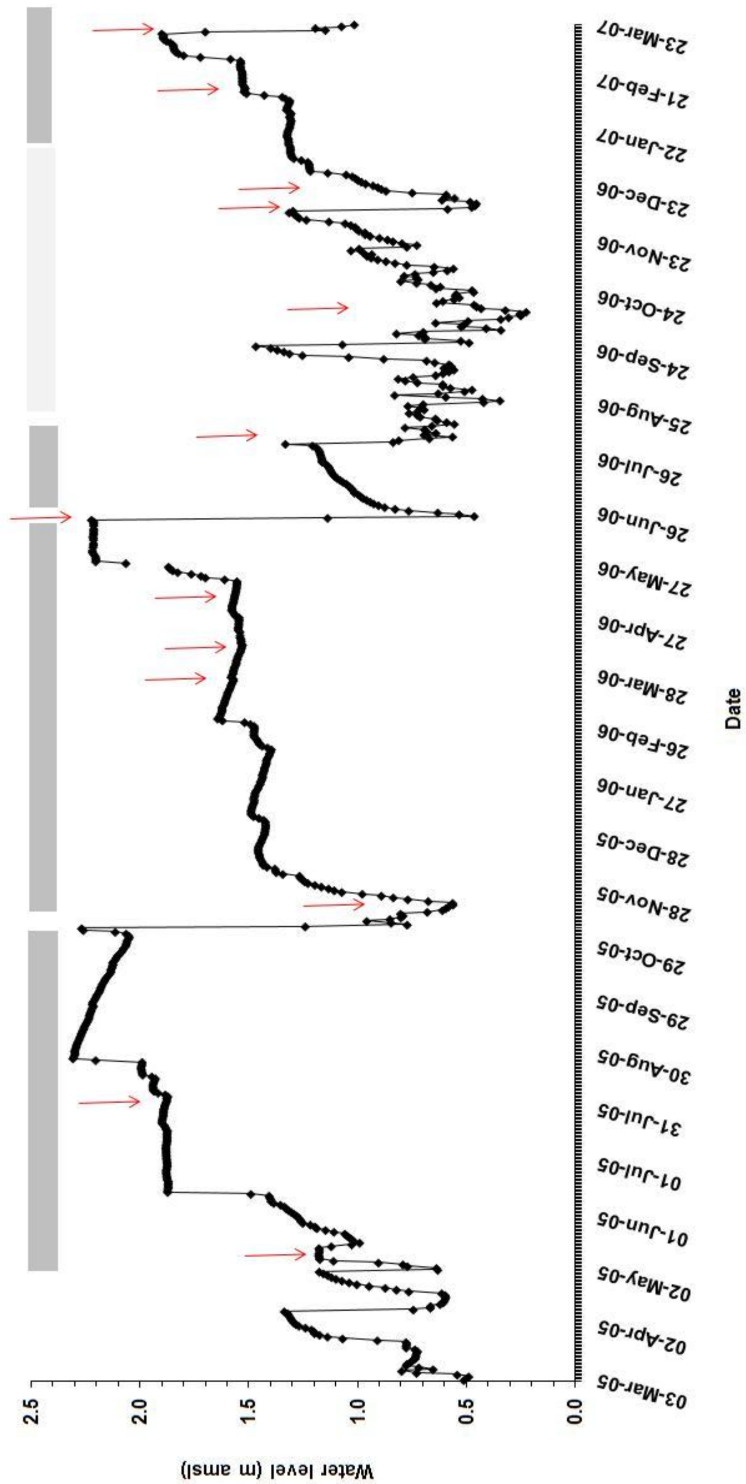


Figure 2.2. Water level and open/closed events over the sampling period. Arrows represent sampling dates. Dark grey block = closed, light grey block=series of rapid open/closed conditions.

Under open mouth conditions salinity ranged between 0.7 to 35.5 ppt (average = 18.5), compared to closed values of 15 to 23 ppt (Figure 2.3). Temperatures varied between 20 and 25 $^{0}$C in the summer months, compared to 13 to 20 $^{0}$C in the winter months. Turbidity differed greatly between open (up to 84 NTU) and closed open conditions (up to 26 NTU). This was probably due to the re-suspension of bottom sediments during tidal exchange since turbidity was negatively correlated to water level ($r^2 = -0.696$, $P < 0.001$).

### *2.3.2 Macrophyte dynamics*

Figure 2.4 shows the percentage cover (within a 1 $m^2$ quadrat) of the macrophytes in the East Kleinemonde Estuary during the sampling period. Intertidal salt marsh, represented by *Sarcocornia tegetaria*, the annual *Salicornia meyeriana* and the hygrophilous grass *Sporobolus virginicus,* occurred between elevations of 0.7 to 1.8 m amsl. Under open mouth conditions, this habitat expanded to a maximum cover of 51 %. Average intertidal salt marsh cover in the transects was 36 % ±10. Maximum cover occurred from August 2006 to January 2007 after a period of rapid open and closed conditions. The maximum monthly rate of expansion for all three transects was measured to be 25 % (Table 2.1), compared to an average monthly expansion rate, quadrat for quadrat, of only 8 % change in cover. Under inundated conditions, intertidal salt marsh became heavily epiphytised with the filamentous green alga *Ulva intestinalis* (L.) Link. Regrowth after habitat exposure, due to the estuary mouth breaching, was mainly from regrowth of existing plant material, as well as from seedling growth. This new growth was visible within one week of habitat exposure. Correlation analysis of percentage cover versus environmental variables showed that intertidal salt marsh cover was significantly affected by water level three months preceding the time of sampling ($r^2 = -0.732$, $P < 0.005$). Neither salinity, temperature nor turbidity significantly influenced a change in intertidal cover abundance.

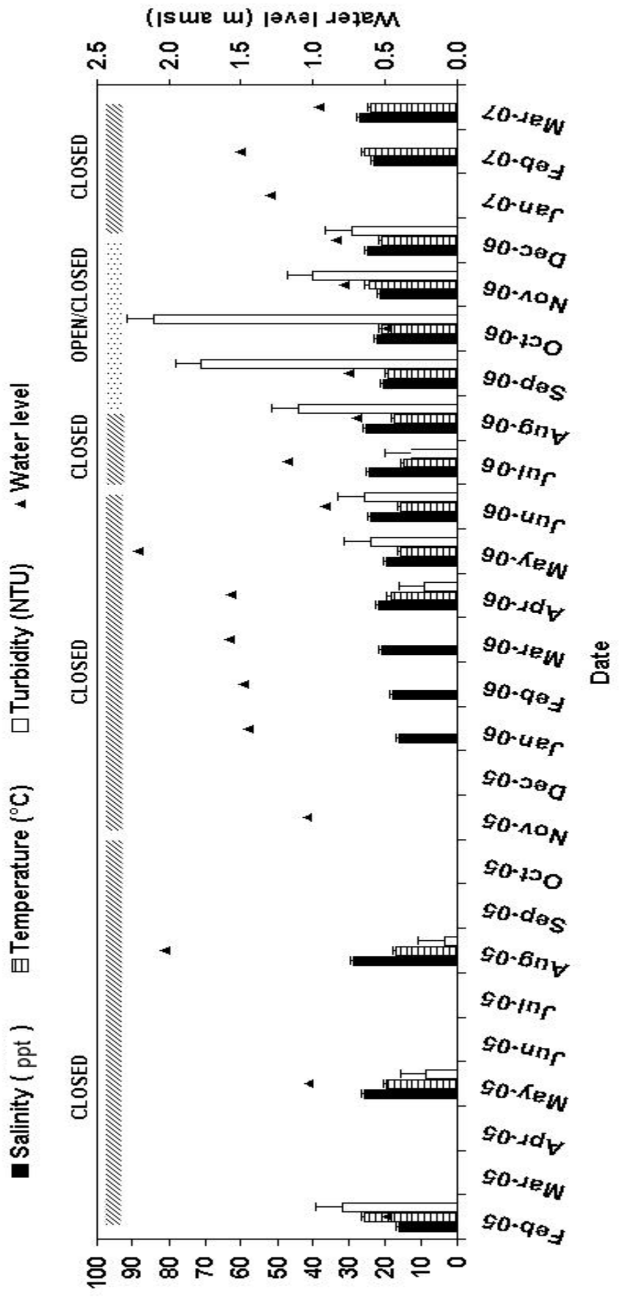


Figure 2.3. Environmental variables in relation to average monthly water level at the time of sampling in the East Kleinemonde Estuary.

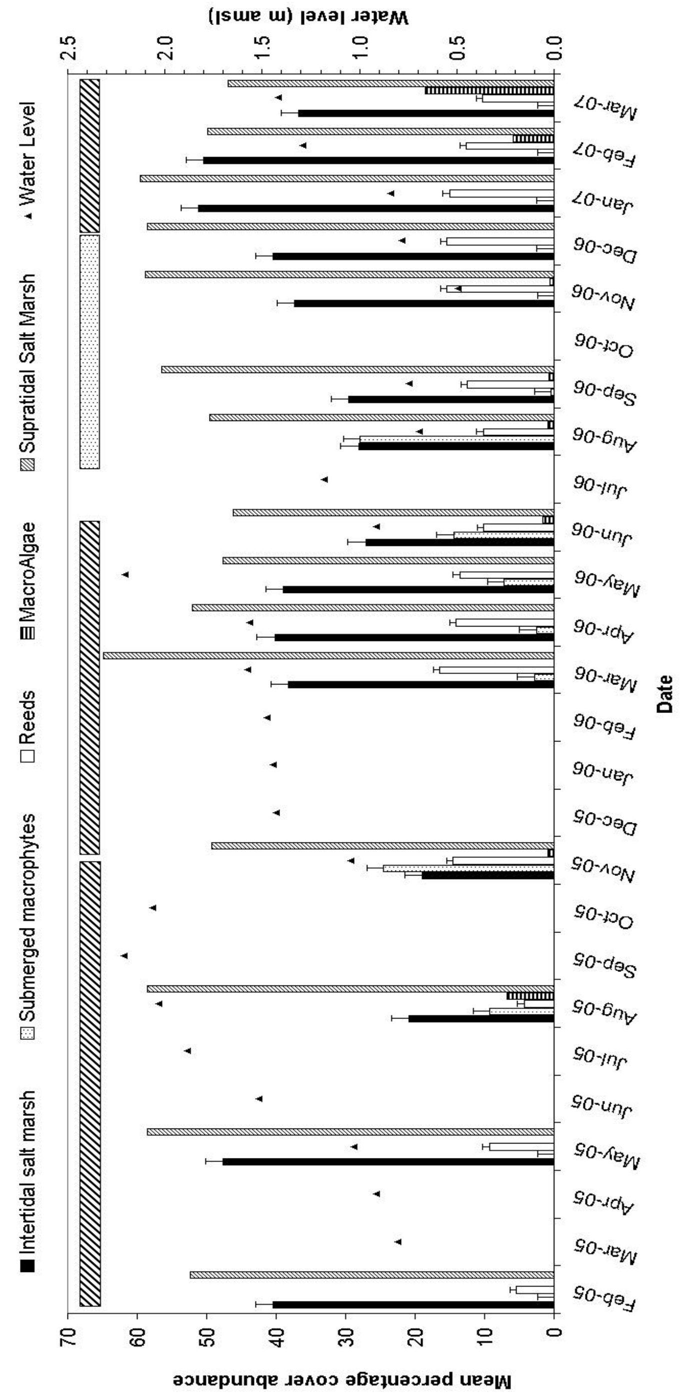


Figure 2.4. Cover abundance (per $m^2$) of the main habitats in relation to average water level at the time of sampling. Hashed block = closed, dotted block=series of rapid open/closed conditions.

Table 2.1. Monthly expansion rates (change in percent cover per month within a 1 $m^2$ quadrat. n = number of samples.

| Habitat | Monthly expansion rates (change in % cover per $m^2$) | | |
|---|---|---|---|
| | Average | Minimum | Maximum |
| Intertidal Salt Marsh: | | | |
| *Sarcocornia tegetaria* | 10 (n=38) | 1 | 32 |
| *Salicornia meyeriana* | 3 (n=22) | 1 | 9 |
| *Sporobolus virginicus* | 12 (n=39) | 1 | 36 |
| Supratidal Salt Marsh | | | |
| *Sarcocornia decumbens* | 11 (n=29) | 1 | 42 |
| *Stenotaphrum secundatum* | 13 (n=25) | 1 | 46 |
| *Juncus kraussii* | 4 (n=30) | 1 | 11 |
| Submerged macrophytes: | | | |
| *Chara* spp. | 9 (n=9) | 2 | 25 |
| *Ruppia* spp. | 5 (n=14) | 1 | 20 |
| Reeds | | | |
| *Bolboschoenus maritimus* | 10 (n=16) | 1 | 28 |
| *Phragmites australis* | 7 (n=19) | 1 | 32 |
| Macroalgae | 15 (n=14) | 2 | 62 |

Supratidal salt marsh, comprising *Sarcocornia decumbens, Stenotaphrum secundatum, Juncus kraussii* and *Limonium scabrum,* occurred mostly at elevations above 1.8 m amsl. Between August 2005 and November 2005, there was a period of two months during which time the water level rose above 2 m and supratidal salt marsh decreased in cover by 9 %. Average monthly expansion rates showed a 13 % change in cover. Maximum expansion rates showed a 33 % change in cover. Correlation analysis of change in cover was significantly related to water depth one month ($r^2 = -0.527$, $P < 0.005$) and two months ($r^2 = -0.557$, $P < 0.005$) preceding sampling.

From March 2006 when the water level in the estuary reached 1.6 amsl, the submerged macrophytes *Chara vulgaris* Thuill. and *Ruppia cirrhosa* (Petagna) Grande began to appear (Figure 2.4). Although *Chara* is a macroalgae, it was included here because it occupies the same habitat areas as *Ruppia cirrhosa* and has the same habitat function. The pioneer species *Halophila ovalis* (R. Brown) J.D. Hooker only occurred in monospecific stands in the main channel at Transect 1 when the mouth was closed. The cover value of 26 %, measured in November 2005 a day after a breach event, represented the maximum submerged macrophyte cover for the previous 9 months of closed conditions. After breaching, this habitat was completely lost due to exposure and desiccation. Recovery once the mouth closed resulted in a maximum cover of only 28 % in August 2006, after 6 months of habitat inundation. This was the maximum cover abundance recorded. Average monthly expansion rates for submerged macrophytes were 7 % change in cover, with a maximum rate of 23 %. An analysis of submerged macrophyte cover and environmental variables showed the strongest correlation to be with temperature ($r^2 = -0.846$, $P < 0.005$) and water level two months preceding the time of sampling ($r^2 = 0.866$, $P < 0.005$).

The reed and sedge habitat remained fairly stable throughout the sampling period, with a maximum increase in cover occurring in November 2005. This was probably in response to increasing water level. The reeds increased at an average rate of 9 % per month, with a maximum change in cover of 30 % per month measured. This habitat was dominated at times by 100 % *Phragmites australis*, with *Bolboschoenus maritimus* and other sedges, all occurring at elevations above 0.9 m amsl.

Macroalgae, represented by *Ulva intestinalis* and *Cladophora* sp. appeared once intertidal salt marsh became inundated by rising water level and were correlated to water depth at the time of sampling ($r^2 = -0.589$, $P < 0.05$). An average expansion rate of 15 % per month was measured, with a maximum rate of 62 %.

## *2.4 Discussion*

Mouth status in small temporarily open/closed closed estuaries is determined by the balance between scouring forces (catchment run-off and tidal prism) and blocking forces (onshore and long shore deposition of sediments) (Whitfield and Bate, 2007). In the East Kleinemonde Estuary the open mouth condition is primarily driven by floods and freshettes, and to a lesser extent by base flow (Van Niekerk *et al*., 2008). Because of the large fluctuation in the physical environment in these small estuaries, estuarine habitats are largely influenced by abiotic rather than biotic variables. By monitoring of fixed transects over regular intervals, this study has shown that water level and habitat inundation influenced macrophyte cover in the East Kleinemonde. Of the other abiotic variables measured, neither salinity nor turbidity was shown to influence macrophyte cover. Temperature negatively influenced submerged macrophyte cover. Salinity was well within the tolerance range of all species as most halophytes can tolerate wide salinity ranges between 0 to 35 ppt (Chapman, 1960; O'Callaghan, 1992; Day, 1981; Adams and Bate, 1994a; Bornman *et al*., 2002). Submerged macrophytes in temporarily open/closed estuaries appear to occur at salinities between 10 and 20 ppt, with 'freshwater' species favouring the lower end of the range, for example *Stukenia pectinata* and 'marine' species the upper end, for example *Ruppia cirrhosa*. *Stukenia pectinata* has been known to occur in the East Kleinemonde Estuary under extended periods of mouth closure following a gradual freshening of the system. Likewise, turbidity did not affect macrophyte growth in this estuary, as has been found in other small systems (Congdon and McComb, 1979; Orth and Moore, 1983; Carruthers *et al*,. 1999; Bernard *et al*., 2007).

After a three month inundation period, the cover of intertidal salt marsh was significantly decreased. This supports the finding of Tölken (1967) who reported that *Sarcocornia natalensis* (Steud.) Dur and Schinz were only able to withstand submergence for up to three months after which time it died. Similarly, in the Great Brak Estuary in South Africa, inundation of *Sarcocornia natalensis* for more than two months resulted in their die-back (Adams *et al*., 1999). In this study however, intertidal salt marsh did not die back completely. When the water level dropped after a breach in June 2006, regrowth occurred from existing material as well as from seed resources in the sediment at an average expansion rate of 8 % per month. Using the maximum monthly expansion rate of 25 % change in cover, a minimum period of four months would be required for intertidal salt marsh to develop 100 % cover under ideal conditions if there was a complete loss in cover. This would be either an open and tidal mouth state or a closed and low water state when intertidal salt marsh is not inundated. Using the average rates of expansion a development period of 12 months was calculated. However since calculations include growth of intertidal salt marsh under slightly flooded conditions, i.e. not completely submerged, growth during this period does not represent ideal conditions. Maximum rates of growth would rather occur when the mouth is open and tidal. Although the form of growth, i.e. linear, exponential, etc. is not known, it is the opinion of the authors that field observations confirm habitat development periods somewhere between values using average and maximum monthly expansion rates.

Supratidal salt marsh in the East Kleinemonde Estuary was negatively affected by inundation at a water level of 1.8 m amsl and higher for one to two months. *Juncus kraussii* generally occurs at higher elevations than supratidal salt marsh implying that it requires lower salinity and less frequent inundation (Naidoo and Kift, 2006). These authors found that flooding with 0.05 m water did not affect biomass of *Juncus kraussii*. In this study there was a significant decrease in percentage cover when the plants were covered with 0.2 m for 2 months, i.e. water level of 1.8 m amsl. None of the other environmental variables affected cover, probably because recent studies have indicated groundwater salinity and depth to groundwater as the primary determinants of growth of this habitat (Bornman *et al*., 2002). Average monthly expansion rates of supratidal salt marsh were 13 %, compared to a maximum change in cover per month of 33 %. At this rate it would take a minimum of three months without high water level for supratidal salt marsh to establish after complete loss.

Submerged macrophyte cover was not correlated to water level at the time of sampling but rather to water level two months preceding. This supports the findings of Carruthers *et al*. (1999) who found that the change in abundance of *Ruppia megacarpa* in Wilson Inlet, SW Australia was strongly related to a depth time lag of two months. This estuary remains closed to the sea for seven months of the year during which time *Ruppia* forms large beds. Exposure of submerged macrophytes after mouth breaching can result in partial or complete loss of biomass within hours due to desiccation (Verhoeven, 1979; Tyler-Walters, 2001; Adams and Bate, 1994 a and b). They can however show rapid growth once an optimum water level returns, completing their life-cycles within 8 to 12 weeks (Calado and Duarte, 2000; Gesti *et al*., 2005). In the East Kleinemonde Estuary with a closed mouth and stable water level for longer than two months, submerged macrophytes developed in areas inundated with as little as 0.02 m increasing at an average rate of 7 % per month. At the maximum monthly rate of change of 23 %, it would require stable water level for four to five months to achieve 100 % cover. Submerged macrophytes occurred mainly in previously exposed intertidal areas. The presence of flooded intertidal macrophytes appears to promote conditions for the development of submerged macrophytes by reducing water movement and creating clear substrate conditions. Response times in this temporarily open/closed estuary, i.e. emergence within one week, were similar to that recorded in lake systems. Submerged macrophytes emerged after five weeks inundation in New Zealand lakes (De Winton *et al*., 2000) and in a temporary Australian Lake. Casanova and Brock (1990) found that Charophytes also emerged after five weeks inundation. If water level in the East Kleinemonde Estuary had remained stable for a longer period, submerged macrophytes would have expanded rapidly. This is because once they establish they stabilise the sediment and improve water clarity, in turn improving conditions for growth (Van den Berg, 1999; Scheffer, 1998; Hemminga and Duarte, 2000; Steinman *et al*., 2002).

Submerged macrophyte cover was negatively correlated to temperature ($r^2 = -0.846$, $P<0.05$) as high temperatures occurred during the open mouth phase when water level was low. As water level and submerged macrophyte cover increased from March 2006 to May 2006 temperature decreased due to the lower seasonal winter temperatures. Summer temperatures ranged between 20 to 25 $^0$C, compared to winter temperatures of 13 to 20 $^0$C. According to Verhoeven (1979), *Ruppia* spp. survives water temperatures of 0 to 38°C, but grows rapidly between 10 to 30°C (Kantrud, 1991). Temperatures in the East Kleinemonde estuary were well within the germination and tolerance ranges of *Ruppia*, and although there was a negative correlation between submerged macrophyte growth and temperature, it is the opinion of the authors that the response of submerged macrophytes to temperature in this study was not a real response but that water level was the main driver. No correlation between water level and temperature was found, only between water level and turbidity.

The reed and sedge habitat of the East Kleinemonde Estuary was not correlated to any of the environmental factors measured and changes in cover are probably linked to seasonal responses (Benfield, 1984). This habitat could potentially be inundated by 1.4 m of water, which may not be a problem because reeds and sedges have been shown to tolerate water depths of up to 4 m in Uganda (Haslam, 1971). The aerial parts of the reeds die off in winter and inundation up to 1.5 m had little impact on cover during closed mouth conditions in the Swartvlei Estuary, South Africa (Howard-Williams and Liptrot, 1980). In this study inundation of greater than 1 m only occurred for one month in winter and therefore did not affect cover. It is likely that even under open and tidal conditions with salinity changing to 34 ppt, freshwater seepage from the surrounding areas would maintain this habitat (Roman *et al*., 1984; Squires and Van der Valk, 1992; Chambers 1997; Rice *et al*., 2000; Soetaert *et al*., 2004; Alvarez-Rogel *et al*., 2007; Adams and Bate, 1999). Average monthly expansion rates of 9 % were calculated, with a maximum monthly rate of 30 %. A cover of 100 % would occur after just over three months.

Macroalgae found in temporarily open/closed estuaries are considered opportunistic, being able to tolerate fluctuating salinity. They proliferate during the closed mouth phase and are washed out to sea during the open phase (Adams *et al*., 1999). In the East Kleinemonde Estuary macroalgal cover was positively correlated to water level, was greatest immediately after mouth closure in the summer months (December 2006) and was not observed during open mouth conditions. Other studies have shown macroalgal growth to be positively related to nutrient input in small temporarily open/closed estuaries (Valiela *et al*., 1997; Menedez, 2005; Collado-Vides *et al*., 2007; Sousa *et al*., 2007), often to the detriment of other submerged species (Giusti and Marsili-Libeli, 2006; Hauxwell *et al*., 2006). The East Kleinemonde is an oligotrophic estuary and it is likely that the initial expansion of macroalgae in this estuary could have been due to the release of nutrients during the decomposition of inundated intertidal salt marsh material. During these pulses macroalgae have been considered important in the uptake of nutrients in shallow estuaries (Valiela *et al*., 1997). Studies on macroalgal biomass and their occurrence have shown water depth (Martins *et al*., 2007) and the presence of rooted macrophytes (Martins *et al*., 2002, cited in Martins *et al*., 2007) to be important drivers. In this study macroalgal cover also showed a positive correlation to water depth ($r^2$ = -0.589). Under these conditions, 100 % cover would be achieved after one and a half months. Macroalgae occurred in this study mainly in areas of inundated intertidal salt marsh, which would provide points of attachment, as supported by Nedwell *et al*. (2002).

In conclusion, this study has shown that macrophytes in a temporarily open/closed estuary are influenced by mouth status, water level, duration of inundation and temperature. The macrophytes responded quickly to these changes, which indicated that regular monthly monitoring is needed to understand macrophyte responses in these small, often highly variable systems. The lagged time response of macrophytes to water level in the months preceding sampling also highlights the importance of findings based on regular sampling efforts and not simply single day sampling. The data from this study can be used to predict macrophyte responses to any future changes in breaching patterns and water level scenarios induced either by natural (droughts or floods) or anthropogenic (abstraction or mouth manipulation) factors. Data on rates of habitat development are important for studies on fauna associated with these habitats. The once breeding population of the endangered river pipefish was lost in 2003 in the East Kleinemonde Estuary due to the loss of submerged macrophyte habitat following desiccation and salinity changes after a flood event altered the frequency and duration of mouth opening (Cowley and Whitfield, 2001).

## *2.5 Acknowledgements*

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## *2.6 References*

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# CHAPTER 3

# THE EFFECT OF A STORM EVENT ON MACROPHYTES IN A TEMPORARILY OPEN/CLOSED ESTUARY

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## *Abstract*

Temporarily open/closed estuaries typically open to the sea due to freshwater inflow coupled with storm surge events. In September 2008, in the absence of freshwater inflow, the mouth of the East Kleinemonde Estuary breached in response to a storm surge. The mouth of the estuary closed the following day at a high level. Marine overwash events following the breach introduced large volumes of saline water into the estuary and raised the water level by 0.07 to 0.33 m. Salinity was significantly higher in the 15 month closed phase after the breach (31 ± 0.9 ppt) compared to 21.9 ± 0.9 ppt in the closed polyhaline phase before the breach. The historical average salinity for the estuary during a closed period is 23 to 25 ppt. The increase in salinity has reduced submerged macrophytes *Ruppia cirrhosa* and *Chara vulgaris* cover by 38.1 %. Macroalgal cover of species such as *Dictyota dichotoma*, *Caulacanthus ustulatus, Codium tenue* and *Ulva* spp. have increased by 7.9 %. The euhaline high water level also significantly reduced supratidal salt marsh cover by 15.2 %, and reed and sedge cover by 19.7 %. Loss of these habitats may result in bank destabilisation and erosion. This is the first record of an extended euhaline period in the 15 years the estuary has been monitored. Sea level rise in association with climate change, together with localised freshwater inflow reduction is likely to result in an increase in marine overwash events. The frequency and duration of closed euhaline periods is likely to increase in this type of estuary. A loss of submerged macrophytes may have significant impacts on faunal composition and abundance and on the subsequent functioning of

temporarily open/closed estuaries. This has serious ecological implications since these estuaries represent 70% of the different types of estuaries found in South Africa.

Keywords: macrophyte; macroalgae; temporarily open/closed estuary; salinity; South Africa

## *3.1 Introduction*

Warming of sea temperature with climatic change will increase wind speeds, wave heights and storm surge events (Scavia *et al*., 2002; Lozano *et al*., 2004; Slott *et al*., 2006; Karim and Mimura, 2008; Wang *et al*., 2008). On the South African coastline, a 10 % change in wind intensity has been shown to result in a 26 % increase in wave height along the South African coastline (Theron and Roussow, 2009). In temporarily open/closed estuaries (TOCEs) an increase in wave height can either result in a build-up of the sand bar across the mouth of the estuary leading to longer periods of closure, more marine overwash events or breaching of the mouth during storm surges. Marine overwash events can be short (< 3 h) or long (> 3 h) (Whitfield *et al*., 2008). The introduction of marine water during an overwash event can lead to periods of increased salinity, especially during hot summer months when evaporation and reduced rainfall occur. However, after prolonged mouth closure, water level usually rises in the estuary due to freshwater input and salinity is gradually reduced over time. Analysis of a modelled 80 year data set showed that overwash events in the East Kleinemonde TOCE occur for 26 % of the time in one year when the mouth is closed (Whitfield *et al*., 2008). Storm surges can introduce large volumes of marine water into a closed estuary. TOCEs are known to fluctuate between periods of macroalgal dominance under euhaline conditions, to submerged macrophytes such as *Ruppia* and *Stukenia* spp. (Healy, 1997; Largier *et al*., 1997; Bachelet *et al*., 2000; Lamptey and Armah, 2008) under mesolhaline and polyhaline conditions. Submerged macrophytes are important in TOCEs because they provide refugia, a food source and reproductive sites for invertebrates (Henninger *et al*., 2009) and fish (Cowley, 1998) and their loss could have serious ecological consequences.

In the East Kleinemonde Estuary, a storm surge event breached the mouth in September 2008. Breaching of the mouth is historically mainly in response to freshwater discharge following rainfall events, usually greater than 100 mm (Cowley and Whitfield, 2001; Van Niekerk *et al*., 2008). The mouth closed within days and salinity gradually increased in the following 15 month closed period, despite periods of high freshwater inflow. This was due to several subsequent marine overwash events linked to storm surge events. This paper documents the changes in cover abundance of the estuarine habitats in response to environmental variables between the closed polyhaline period before the breach and the closed euhaline period afterwards.

## 3.2 Materials and Methods

### 3.2.1 Study site

The East Kleinemonde Estuary (33° 32' S, 27° 03' E) lies approximately 143 km north of Port Elizabeth. It is one of 175 temporarily open/closed estuaries in South Africa. It is a small estuary with a catchment of 43.5 $km^2$ and a length of 3.7 km (Badenhorst, 1988). Under closed mouth conditions and high water level the estuary has a surface area of 35.7 ha (Whitfield *et al*., 2008). Mouth closure normally occurs for approximately 90 % of the year. The catchment of the estuary is used for cattle and pineapple farming and the lower reaches near the mouth has residential development. The East Kleinemonde Estuary is oligotrophic with localised nutrient inputs from septic tanks occurring near the mouth.

### 3.2.2 Vegetation analysis

Macrophyte cover was assessed monthly along three permanent transects in the East Kleinemonde Estuary from March 2006 until December 2009. Each transect spanned the width of the estuary and ranged from 140 m to 335 m in length. Duplicate quadrats were placed at 5 m intervals along the length of each transect and macrophyte cover abundance (%) was assessed within each quadrat. Cover data was averaged along the elevation range of each habitat, as done in Chapter 2.2.3.

### 3.2.3 Water column analysis

Water column data were collected at nine stations along the length of the estuary with readings taken at the surface, middle and bottom of the water column using a YSI 556 MPS multiprobe. Water column salinity (ppt), conductivity (EC), temperature (Celsius) and turbidity (measured as Secchi depth) were recorded monthly for 2006, 2008 and 2009. In 2007 data were collected every second month. A water level recorder (P4H002) was installed on the R72 bridge in 2004 by the Department of Water Affairs, South Africa. Water level data are logged every 10 minutes and these data are corrected for mean sea level. Average daily water levels (m above mean sea level) were used for the study period. Daily rainfall data and air temperatures were obtained from the South African Weather Bureau for the Port Alfred area.

### *3.2.4 Data analysis*

A student t-test was used to compare data for the periods before and after the September 2008 breach. Cover data for submerged macrophytes and macroalgae were assessed for the low salinity closed period (August 2007 to August 2008) compared to the euhaline closed period (September 2008 to December 2009). These were the periods when the mouth was closed and habitat was available for submerged macrophyte and macroalgal growth. Pearsons Product Moment Correlation was used to determine the strength of association between vegetation cover abundance and environmental variables. Water column data were averaged for the entire estuary per sampling period because there were few vertical or horizontal differences. Habitat cover was averaged over each transect and these data were used for statistical analysis in relation to water column variables. The full sampling period, namely March 2006 to December 2009 was used for intertidal salt marsh, supratidal salt marsh and reeds and sedges. Statistical analysis was done using Statistica Version 7 (Statsoft Inc., 2004).

## *3.3 Results*

A storm surge event on the 1 September 2008 breached the East Kleinemonde Estuary after a closed period of 15 months. Tidal heights in Port Elizabeth were 0.68 m above predicted values due to a low pressure system and strong onshore winds. The mouth of the East Kleinemonde Estuary breached at 2.15 m amsl and closed the following day at a high water level of 1.18 m amsl. Subsequent overwash events raised the water level in the estuary substantially (Figure 3.1a). In October 2008 water level increased by 0.21 m, in December 2008 by 0.33 m and in May 2009 by 0.13 m. In the closed period prior to the breach average salinity was 21.9 ± 0.9 ppt, significantly lower than in the closed euhaline phase after mouth closure (31 ± 0.9 ppt, $P < 0.001$, $n = 17$). Two rainfall events of 42.2 mm and 34.6 mm, increased water level by 0.26 m and 0.66 m respectively. Although salinity decreased to 23 following these freshwater inputs, it increased due to subsequent overwash events and summer evaporation.

The East Kleinemonde Estuary has five dominant macrophyte habitats, namely supratidal salt marsh, intertidal salt marsh, submerged macrophytes, macroalgae and reed and sedges (Riddin and Adams, 2008). Submerged macrophytes were represented primarily by *Ruppia cirrhosa* and *Chara vulgaris* with isolated patches of *Halophila ovalis* (R. Brown) Hooker *f.* and *Zannichelia palustris* L. occurring sporadically. The macroalgae, *Chara vulgaris* has been included with submerged macrophytes due to its similarity in habitat provision. Cover abundance of submerged macrophytes significantly decreased from 53 ± 6.6 % to 14.9 ± 3.1 % after the September 2008 breach ($P < 0.001$) (Figure 3.1b). Cover decreased progressively until only 1.7 % was recorded in March 2009. By this time *Chara vulgaris* had almost completely disappeared. Submerged macrophyte cover increased to 48 % in December 2009. Macroalgal cover was significantly higher in the closed euhaline period after the breach compared to the closed polyhaline period before (10.3 ± 2.1 compared to 2.4 ± 1.1 %, $P < 0.05$) (Figure 3.1b). There was a significant decrease in average supratidal salt marsh cover from 94.5 ± 2.5 % to 79.3 ± 3.2 % after the breach ($P < 0.05$) (Figure 3.1c). Dominant species affected were *Juncus acutus, Sarcocornia decumbens* and *Stenotaphrum secundatum*. Average cover for the dominant reed *Phragmites australis* also decreased significantly from 36.1 ± 3 % to 16.4 ± 2.5 % (Figure 3.1c).

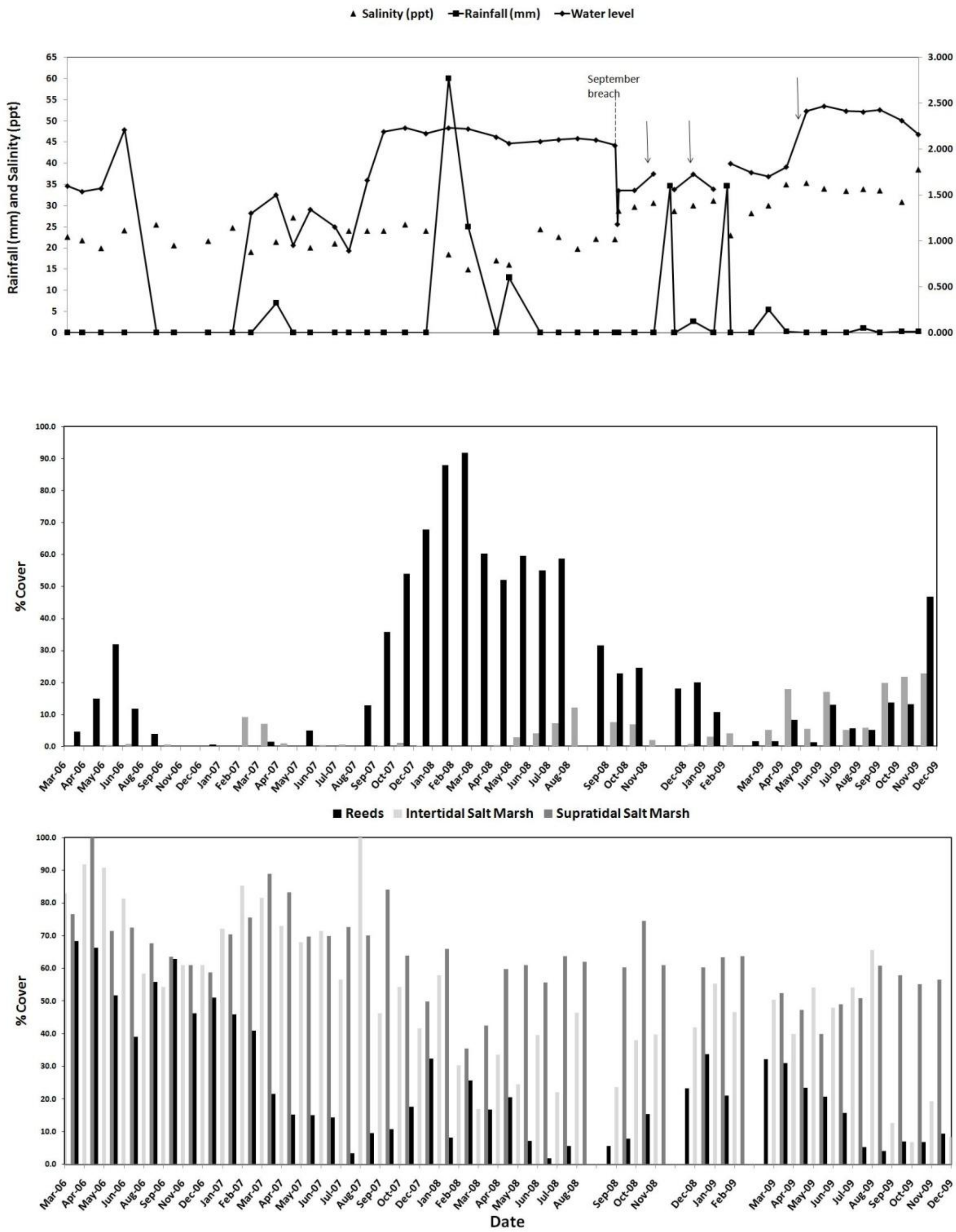


Figure 3.1. Estuary water level, salinity and rainfall (a) in the East Kleinemonde Estuary between March 2006 and December 2009, with the associated change in submerged and macroalgal cover (%) (b) and supratidal, intertidal salt marsh and reed habitat (%) (c). Arrows indicate the overwash events subsequent to the September 2008 storm induced breach. Boxes represent the closed polyhaline period before the breach and closed euhaline period after the breach, as well as an open euahaline phase.

Correlation analysis showed that all habitats were influenced by salinity and water level except for supratidal salt marsh where cover was only correlated to salinity ($r^2$ = -0.51, n = 33, P < 0.05). For all habitat types an increase in salinity decreased cover except for macroalgal cover where there was a positive relationship with salinity ($r^2$ = 0.61, n = 33, P < 0.05) and water level ($r^2$ = 0.43, n = 33, P < 0.05). A reduction in cover for both intertidal salt marsh and reeds and sedges was related to an increase in salinity ($r^2$ = -0.49 and -0.48 respectively, n = 33, P < 0.05) and an increase in water level ($r^2$ = -0.39 and -0.58 respectively, n = 33, P < 0.05). Submerged macrophyte cover was significantly related to water level ($r^2$ = 0.42, n = 33, P < 0.05) and salinity ($r^2$ = -0.37, n = 33, P < 0.05). Air temperature was related to increased cover of intertidal salt marsh ($r^2$ = 0.45, n = 33, P < 0.05) and reeds and sedge ($r^2$ = 0.41, n = 33, P < 0.05) and decreased cover of submerged macrophytes ($r^2$ = -0.36, n = 33, P < 0.05). Rainfall was only correlated to submerged macrophytes ($r^2$ = 0.36, n = 33, P < 0.05).

## *3.4 Discussion*

This study reports on the effect of a storm induced breach on a temporarily open/closed estuary (TOCE) and the resultant change in estuarine vegetation due to increased saline conditions. At the time of the breach analysis of actual versus predicted tidal data for nearby Port Elizabeth showed that tidal heights were 0.68 m higher than predicted due to a low pressure system and southerly winds. A similar storm surge in March 2007 had devastating affects on the east coast of South Africa where tide heights were 0.55 m higher than predicted (Brundrit, 2008). Under the present climate change scenario extreme weather patterns and storm events are likely to increase, leading to increased tidal surges and marine overwash events in TOCEs.

Marine overwash is common in the East Kleinemonde Estuary, occurring 26 % of the time when the estuary is closed to the sea (Whitfield *et al*., 2008). Historically only short term salinity changes of up to 3.8 occur during these overwash events, this change usually only affects the mouth region of the estuary (Whitfield *et al*., 2008). After a few weeks salinity decreases due to dilution from direct freshwater inflow or seepage (Gama *et al*, 2005; Snow and Taljaard, 2007). However during this study period, salinity was significantly higher in the estuary in the closed euhaline period after the breach than before, a difference of 9.1 ppt occurred in the 15 month closed periods after the breach. The higher salinity resulted in a 15.2 % decrease in supratidal salt marsh cover and a 19.7 % decrease in reeds and sedge cover. Loss of these peripheral habitats have been shown to result in bank destabilisation such as in the St Lucia Estuary, South Africa under extreme hypersaline periods, resulting in increased bank erosion (Taylor *et al*., 2006).

In the closed euhaline period macroalgal cover increased by 7.9 % and this increase was significantly correlated to an increase in salinity and water level. Macroalgae and submerged macrophytes are dominant in the East Kleinemonde Estuary when water level in the estuary is greater than 1.3 m amsl (Riddin and Adams, 2008). *Codium tenue* Kűtze, *Dictyota dichotoma (*Hudson) Lamouroux, *Caulacanthus ustulatus* (Mertens ex Turner Kűtzing) and *Ulva* spp. replaced *Cladophora*, *Enteromorpha*, *Chaetomorpha*, *Polysiphonia* and *Fucus* species. The latter are common opportunistic species in estuaries worldwide (Adams *et al*., 1992; Fong *et al*., 1996), often forming blooms and dense mats, mainly in response to eutrophication (Hauxwell *et al*., 2001; Kamer *et al*., 2001; Menendez, 2005). Under these conditions they out-compete long lived macroalgal species (Wilkinson *et al*., 2007).

Submerged macrophyte cover was significantly reduced for *Ruppia cirrhosa* and almost entirely removed *Chara vulgaris*. This was correlated to the higher salinity. In the Wilson Inlet, Australia Carruthers *et al*. (1999) found that conductivity, turbidity and depth accounted for 40 % of the change in seasonal cover abundance of *Ruppia megacarpa*. Although *Ruppia cirrhosa* germinates and grows well over the 0 to 35 salinity range (Adams and Bate, 1994) the higher salinity values are likely to have slowed the recovery period. In the present study macrophyte beds are slowly recovering, with a maximum cover of 48 % recorded in December 2009. *Chara vulgaris* had a maximum cover of only 0.17 %. *Chara vulgaris* has a lower salinity tolerance than *Ruppia cirrhosa* with maximum germination occurring at 0 ppt. Germination is significantly reduced and delayed at 35 (Riddin and Adams, 2009). *Stukenia pectinata* has been known to occur in the East Kleinemonde Estuary but it is unlikely that this species will reappear under the current salinity regime. Furthermore local drought conditions have reduced freshwater inflow into the East Kleinemonde Estuary further increasing salinity. In the nearby Kasouga Estuary, there was an 80 % loss in submerged macrophytes (mainly *Ruppia cirrhosa*) due to hypersaline conditions (up to 40). Hypersalinity in this estuary was also due to below average rainfall, evaporation and marine overwash events (Froneman and Henninger, 2009). The loss of macrophyte beds resulted in a significant decrease in shrimp and isopod populations, as well as an increase in predation by invertebrate feeding fish species due to loss of refuge areas provided by macrophyte beds.

Long term hypersalinity in temporarily open/closed estuaries is well documented in many Australian and Mediterranean estuaries due to their intermittent link with the sea (Largier *et al*., 1997; Bachelet *et al*., 2000), UK estuaries (Healy 1997), and African estuaries (Lamptey and Armah, 2008). These long term changes are mostly due to annual or decadal climatic variability and have been recorded to occur every 2 to 4 years in some estuaries (Healy, 1997). In the East Kleinemonde Estuary this has been the longest period of mouth closure and euhaline conditions in the 15 years that the estuary has been monitored. In the early 2000s, mouth closure occurred for a maximum of 2 years but polyhaline to mesohaline conditions occurred due to gradual freshening from freshwater inflow. *Stukenia pectinata* was the dominant submerged macrophyte (Whitfield *et al*., 2008).

This paper suggests increased storm surge and marine overwash events associated with large scale climatic conditions, together with localised decrease in freshwater inflow, could result in increased frequency and duration of euhaline closed periods in TOCEs. This in turn could result in a loss of peripheral, bank stabilising vegetation, as well as submerged macrophytes and their associated fauna such as invertebrates, fish and bird populations (McGlathery, 2001; Mannino and Sara, 2006; Froneman and Henninger, 2009). This has serious ecological implications since these estuaries represent 70% of the different types of estuaries in South Africa.

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# CHAPTER 4

# THE SEED BANKS OF TWO TEMPORARILY OPEN/CLOSED ESTUARIES IN SOUTH AFRICA

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## *Abstract*

The seed banks of two temporarily open/closed estuaries in South Africa were quantified in this study. Charophyte öospores represented almost 72 % of the sexual propagules in the sediment with a mean öospore density of 31 306 ± 2 293 öospores m$^{-2}$. This was followed by the seeds of the intertidal salt marsh plant *Sarcocornia tegetaria* (18 %) (7 929 ± 688 seed m$^{-2}$) and the submerged angiosperm *Ruppia cirrhosa* (7 %) (2 852 ± 327 seeds m$^{-2}$). The remaining 3 % was made up of a mixture of species such as *Salicornia meyeriana*, *Sporobolus virginicus*, *Stukenia pectinata*, *Bolboschoenus maritimus* and terrestrial species. Although seed density did not differ significantly with depth, seeds still occurred at 20 cm depth providing a regeneration source in the event of sediment disturbance. Three salinity (0, 17 and 35 ppt) and moisture treatments (exposed, waterlogged and submerged) were applied to collected sediment to determine how fast species would germinate. *Sarcocornia tegetaria* germinated after 3 days to a maximum of 82 %. Submerged species began to germinate only after 18 days (*Chara vulgaris* and *Ruppia cirrhosa*) and had low germination percentages of between 11 and 15 % after 91 days. Results from this study indicate that in the event of unpredictable disturbance events such as water level fluctuations, large sediment seed reserves would ensure habitat persistence.

**Keywords**: seed bank, estuary, germination rate, Charophytes, *Ruppia*, *Sarcocornia*

## *4.1. Introduction*

Temporarily open/closed estuaries can show large variations in environmental conditions, mainly salinity and water level, driven by freshwater inflow (Van Niekerk *et al*., 2008). These variations can be so extreme that whole macrophyte communities may be lost after each disturbance event. Salt marsh for example, is permanently lost after 3 months submergence (Riddin and Adams, 2008), whereas submerged macrophytes are lost within days due to exposure and desiccation when the mouth of the estuary breaches (Adams and Bate, 1994). These changes take place rapidly and at unpredictable frequencies. Persistence of macrophyte habitats depends on the rate and extent of recovery. Recovery can take place through either vegetative/asexual growth, for example rhizomes and tubers that exist in the sediment, or through viable seed/sexual reserves in the sediment. Studies have shown that as habitat disturbance increases, growth from seed reserves becomes more important (Casanova and Brock, 1996; Combroux and Bornette, 2004). Although most seed occurs within the top 5 cm of the sediment, seed reserves with depth can be a source of propagules in areas where disturbance occurs (De Winton *et al*., 2000; Dugdale *et al*., 2001).

Seed banks have been widely quantified in freshwater marshes and wetlands (Harwell and Havens, 2003; Shilli *et al*., 2007) and in coastal salt marshes (Grillas *et al*., 1993; Wolters and Bakker, 2002). However limited quantification of seed reserves in temporarily open/closed estuaries has been reported internationally (Gesti *et al*., 2005; Shaw *et al*., 2008). No studies of this nature have been done on temporarily open/closed estuaries in South Africa. A few studies have focussed on the germination requirements of estuarine macrophytes (Naidoo and Naicker, 1992; Naidoo and Kift, 2006; Shaw *et al*., 2008). Germination rates also give an indication of how quickly habitats respond when optimum conditions occur. The main objective of this study was therefore to quantify and describe the seed banks of two temporarily open/closed estuaries. Germination rates of key estuarine species under three salinity and three water level treatments were also assessed. It was hypothesised that seed numbers will be comparable to other unpredictable habitats that show large environmental variations. This study forms part of a larger multidisciplinary study on the functioning of temporarily open/closed estuaries in South Africa. Data will provide an understanding of the dynamics of macrophytes in these highly variable and unpredictable ecosystems.

## *4.2. Materials and methods*

### *4.2.1 Site description*

The East and West Kleinemonde estuaries are situated 15 km north-east of Port Alfred in the Eastern Cape of South Africa (33° 32´ S, 27° 03´ E) and are about 400 m apart. At times in the past the two estuaries shared the same mouth. The East Kleinemonde Estuary has a smaller catchment area (43.5 $km^2$) than the West Kleinemonde (93.7 $km^2$) (Badenhorst, 1988) and the mouth opens and closes rapidly in response to freshwater inflow. The surface area of the West Kleinemonde is approximately 80 ha when full, in comparison to 35.7 ha in the East Kleinemonde Estuary. Both estuaries can remain closed for extended periods of time, *circa* 2 years, due to the formation of a sand bar at the mouth. During the study period the mouth of the East Kleinemonde Estuary opened more frequently than that of the West Kleinemonde. The mouth in both estuaries may remain open for a few days to a few weeks. Intertidal salt marsh in both estuaries is characterised by *Sarcocornia tegetaria*, *Sporobolus virginicus* and *Salicornia meyeriana*. When the mouths of both estuaries are closed, inundated salt marsh is replaced by submerged macrophytes after two months (Riddin and Adams, 2008). Submerged species are characterised by Charophytes *Chara vulgaris* and *Lamprothamnium papulosum* (Wallr.) J. Gr. and angiosperms *Ruppia cirrhosa* and *Stukenia pectinata* depending on the salinity at the time. Water level fluctuations of up to 1 m occur at the sample sites for both estuaries. In a previous study vegetation change in response to water level fluctuations was investigated along three permanent transects at three different sites in the East Kleinemonde Estuary (Riddin and Adams, 2008), Percentage cover abundance of extant vegetation was measured every 5 m within duplicate 1 $m^2$ quadrats over the length of each transect. Data from this study, as well as data based on visual observations from the West Kleinemonde were used to relate vegetation cover to sediment seed density in this study.

### *4.2.2. Seed density*

Three sites were selected in both estuaries in areas where the vegetation changes in response to water level fluctuations. Sites in both estuaries were sampled in March, May 2006 and in February 2007. In addition to this the West Kleinemonde Estuary was sampled in August 2006 when the mouth of the estuary opened and seed sampling sites were exposed and in the East Kleinemonde Estuary in November 2006 when water level was low. At each site, 45 sediment cores were randomly collected within a 10 x 10 m plot (4 cm diameter and 5 cm deep). To account for the spatial heterogeneity of seed distribution, samples for each site were aggregated into separate buckets and then sub sampled. The surface area of the aggregate sample for each site represented 0.0565 $m^2$. The buckets were closed and stored at 4 $^0$C until analysis.

The seed banks were quantified using the direct counting method. This method is less commonly used than the indirect or seedling emergence method. However since little information exists on the effects of storage conditions and germination requirements of estuarine macrophytes in this study, the direct counting method was considered to give a more accurate estimate of viable seed density. From the collected aggregate sample for each site, three sub samples of 100 mL each were analysed for seed numbers. Each sub sample was wet sieved with tap water through a 250 µm sieve. This sieve size represents the smallest propagule size, namely the öospore of Charophytes (hereafter included in the terms "seed" and "seed banks"). The concentrated seed sample was then vacuum-filtered onto filter paper through a Buchner funnel to remove excess water. The filter paper containing the seed material was left to air dry at room temperature for a minimum of 3 to 4 days. Once dried, the concentrated sample was analysed under a dissecting microscope. Seed numbers were extrapolated to express seed density per $m^2$ for the combined sites. The percentage of seeds represented by the dominant estuarine species was calculated. Viable seeds and öospores were identified as those having an intact seed coat, turgid condition (by applying a light pressure to the propagules) and healthy starch reserves when squeezed (Casanova and Brock, 1990).

The distribution of seeds with depth was measured from sediment cores taken in May 2006 at 0 - 5, 5 - 10 and 10 - 20 cm depths in the East Kleinemonde Estuary. This was a preliminary investigation on depth distribution and therefore sampling only took place at the one estuary. At three sites in the estuary 45 core samples were collected for each of the three depths and aggregated. At Site 3 (EK3) sediment at 10 to 20 cm could not be collected due to hard bedrock. From each aggregate sample 3 sub samples were analysed. The procedure outlined in the previous paragraph was used to determine seed density and percentage representation of the dominant estuarine species.

### *4.2.3. Germination rates*

The seedling emergence method was used to measure germination rates for the dominant estuarine species using three salinity treatments (0, 17 and 35 ppt) and three water level treatments (E=exposed, W=waterlogged and S=submerged). Salinity and hydrological conditions are considered the two main environmental variables influencing germination in estuarine macrophytes (Khan and Gul, 2006). Samples from March 2006 were used to assess germination rates for seeds from the East and West Kleinemonde estuaries. Sediment from three different sites in each estuary was exposed to the different treatments. From the aggregated sample per site, 50 mL of sediment was spread over a 4 cm layer mixture of potting soil and river sand in 12 cm diameter pots. Three replicate samples were used from each site. The data were pooled for the sediment from the two estuaries as percentage germination was expressed per species. This mixture was used since pure potting soil can result in gas formations and algal blooms in submerged samples (Boedeltjie *et al.,* 2002). Seawater was diluted with fresh tap water to obtain 17 ppt. For the submerged treatments, pots were placed in plastic drums and a water level of 5 cm was maintained over the sediment surface by topping up as and when needed. Waterlogged treatment pots were placed in plastic drums and stood in water so that the surface of the sediment always remained moist. Exposed samples were watered when dry with fresh tap water. A control pot was used to determine the presence of seed in the potting soil and seed dispersal in the greenhouse. The experiment ran for a period of 91 days, adequate time for both submerged and intertidal salt marsh species to germinate. Glasshouse temperatures ranged from 2 to 36 $^{\circ}$C and photoperiod ranged from 12:12 (Light:Dark) to 10:14 (Light:Dark) at the end of the study (South African Weather Bureau). Emergence of seedlings was assessed initially every 3 days for the first two weeks and thereafter at weekly intervals until the end of the experiment. Seedlings were counted and removed after identification so as to remove any harmful allelopathic or competitive interactions. Emergence was defined as the development of a germinated seedling to a stage where it could be detected by eye (De Winton *et al*., 2000). After completion of the experiment, sediment from the pots was sieved through a 250 µm sieve to extract any remaining seed so that percentage germination could be calculated. Germination rates were expressed as the accumulative number of seedlings (germination %) germinating over the trial period.

### *4.2.4. Data analysis*

A one-way ANOVA was used to determine the significant difference among the mean seed density in both East and West Kleinemonde estuaries, as well as with depth. When significant differences were found ($P < 0.05$), a post-hoc comparison of means was run using Tukey's Honest Significant difference test. Regression analysis was done to determine the relationship between sediment seed density and cover abundance of extant vegetation. All statistical analyses were run using Statistica (Version 7.1) (StatSoft Inc. 2006).

## ***4.3. Results***

### *4.3.1. Seed density*

The highest proportion of seeds in the top 5 cm of the sediment for both estuaries was represented by Charophyte öospores, showing some degree of calcification (71.8 %) (Table 4.1). Although the extant vegetation in each estuary was at times represented by both *Lamprothamnium papulosum* and *Chara vulgaris* it was not possible to separate the species using öospore characteristics and therefore they were grouped together as "Charophytes". A maximum density of 100 000 öospores $m^{-2}$ was recorded at one site in the West Kleinemonde Estuary. There was no significant difference in the mean density of

Charophyte öospores between the two estuaries ($P > 0.05$, $n = 72$). No relationship was found between sediment seed density and cover abundance of extant vegetation ($P > 0.05$, $n = 85$). Percent cover abundance for extant vegetation is given as a range in Table 4.1 since variance was high due to large water level fluctuations and the resultant vegetation changes (0 to 100 % cover in some cases). The second highest seed density was represented by the intertidal halophyte *Sarcocornia tegetaria* (18.2 %), followed by the submerged angiosperm *Ruppia cirrhosa* (6.6 %). Other estuarine species, namely *Salicornia meyeriana*, *Sporobolus virginicus* and *Stukenia pectinata*, made up less than 2 % of the sediment seed bank. *Sarcocornia tegetaria* seed numbers were significantly higher in the East Kleinemonde Estuary (10 521 seeds $m^{-2}$; $P < 0.05$; $n = 12$) whereas *Ruppia cirrhosa* seeds were significantly higher in the West Kleinemonde Estuary (4 990 seeds $m^{-2}$; $P < 0.001$; $n = 12$).

Table 4.1. Mean seed density of the dominant estuarine macrophytes (± SE, n = 72), their percentage composition of the seed bank collected in the East and West Kleinemonde estuaries (2006 and 2007) and extant cover per $m^2$ at the time of seed sampling. Both seed densities and cover values were combined for the all sampling sites in both estuaries. n = number of samples.

| Species | Mean Seed Density (No. $m^{-2}$) | Mean Proportion in seed bank (%) | Extant Vegetation cover range (% $m^{-2}$) |
|---|---|---|---|
| Charophytes | 31 306 ± 2 293 | 71.8 | 0 to 32.5 % |
| *Sarcocornia tegetaria* | 7 929 ± 688 | 18.2 | 0 to 97.5 % |
| *Ruppia cirrhosa* | 2 852 ± 327 | 6.6 | 0 to 100 % |
| *Salicornia meyeriana* | 306 ± 58 | 0.7 | 0 to 25 % |
| *Sporobolus virginicus* | 163 ± 37 | 0.4 | 0 to 100 % |
| *Stukenia pectinata* | 77 ± 22 | 0.2 | 0 |
| *Bolboschoenus maritimus* | 31 ± 12 | 0.1 | 0 to 10 % |
| Other | 639 ± 21 | 2 | 0 to 25 % |

Seed density did not show any significant variation with sediment depth for the East Kleinemonde Estuary (Figure 4.1). Seeds of all species showed a high degree of variation with sediment depth.

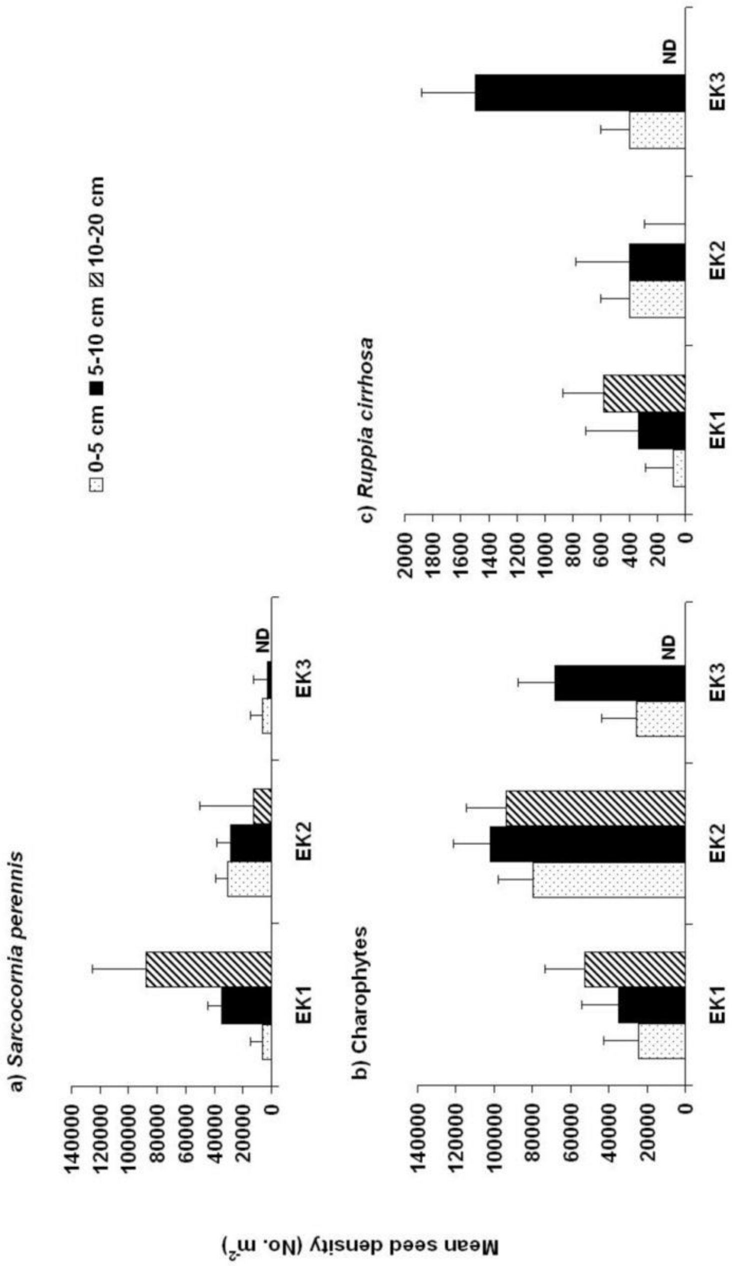


Figure 4.1. Mean sediment seed density with depth for East Kleinemonde Estuary in May 2006 (bars ± SE, n = 3). EK1 to 3 refers to the sampling sites in the East Kleinemonde Estuary. ND = No Data

### *4.3.2 Germination rates*

The intertidal salt marsh species *Sarcocornia tegetaria* germinated after three days (Figure 4.2). A maximum germination of 82 % was achieved after the 91 day trial period and during this time germination took place continuously. Germination was highest in freshwater. After 24 days, 50 % of the *Sarcocornia* seeds had germinated at 0 ppt. At 17 ppt, 50 % germination took place after 60 days and at 35 ppt 50 % germination only took place after 77 days. Maximum germination occurred at 0 ppt (82 %) and differed significantly from germination at 35 ppt (57 %) ($P < 0.05$, $n = 18$). There was no significant difference in germination between exposed or waterlogged treatments. However germination was significantly reduced by submergence with 5 cm water ($P < 0.05$, $n = 18$). Under waterlogged conditions maximum germination was 39 % compared to only 6 % for the submerged treatment.

Submerged macrophytes only began to germinate after 18 days for both *Chara vulgaris* and *Ruppia cirrhosa*. Maximum germination for *Chara vulgaris* was 15 % (at 0 ppt) and 11 % for *Ruppia cirrhosa* (at 35 ppt). Salinity had no significant effect on the germination of *Ruppia*. However germination of *Chara vulgaris* was significantly higher at 17 compared with 35 ppt ($P < 0.05$, $n = 18$).

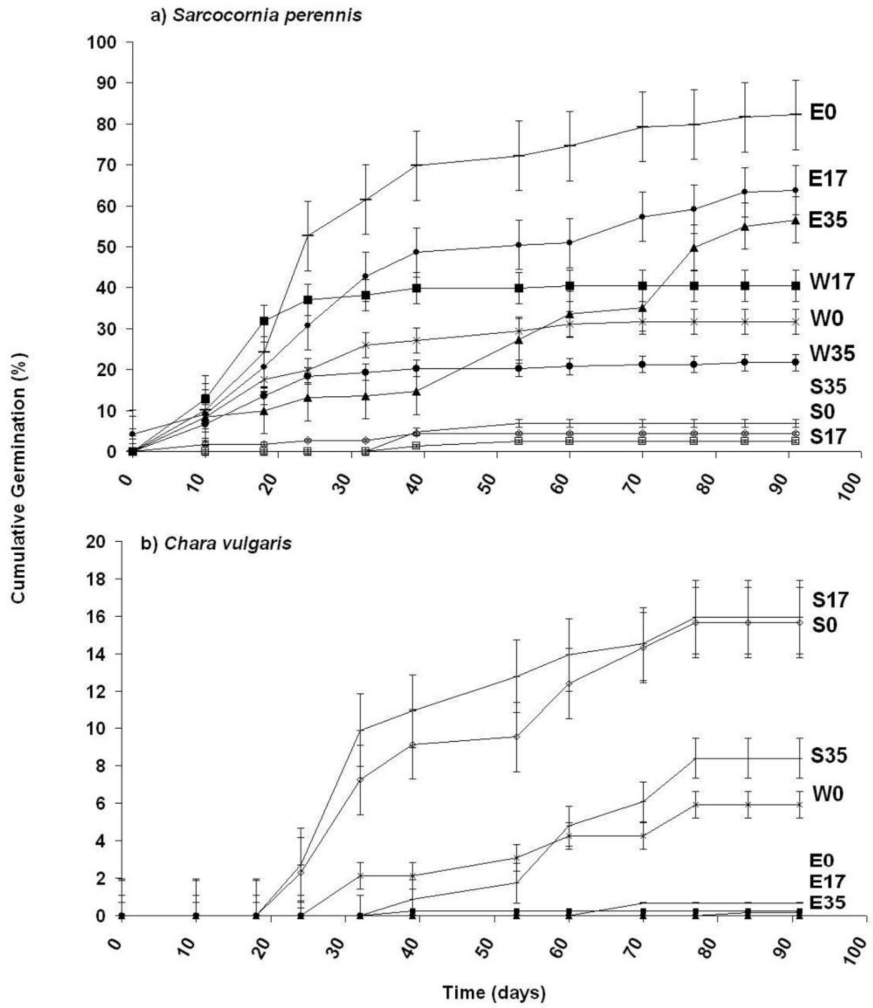


Figure 4.2. Cumulative germination (%) of *Sarcocornia tegetaria* and *Chara vulgaris* from the East and West Kleinemonde estuaries under three salinity (0, 15 and 30 ppt) and three moisture (E = Exposed, W = Waterlogged and S = Submerged) treatments (bars ± SE, n = 18).

## *4.4. Discussion*

This study has shown that the seed banks of two temporarily open/closed estuaries are characterised by low species diversity but high seed density. Seed numbers were high for three of the dominant estuarine macrophytes which made up 98 % of the seed bank. Charophyte öospore densities in this study (31 306 öospores m$^{-2}$) were similar to the densities found in temporary marshes (29 000 to 417 700 seeds $m^{-2}$) (Bonis *et al*., 1995). The second most represented species was *Sarcocornia tegetaria* with a mean seed density of 7 929 seeds $m^{-2}$. Seed density data does not exist for this species, but a similar supratidal species *Sarcocornia pillansii* was found to have 3 616 seeds $m^{-2}$ (Shaw *et al*., 2008), whereas an annual intertidal salt marsh species *Salicornia europaea* had much higher seed densities (38 944 to 128 000 seeds $m^{-2}$) (Philipupillai and Ungar, 1984). *Ruppia cirrhosa* seed density was much higher in this study (2 852 seeds $m^{-2}$) than in a Mediterranean coastal lagoon (593 seeds $m^{-2}$) (Gesti *et al*., 2005) but similar to that found in a marsh environment (2 920 to 5030 seeds $m^{-2}$) (Bonis *et al*., 1995). High numbers in all these species are probably due to persistent seed bank reserves brought about by adverse environmental conditions, such as water level fluctuations which are typical of temporarily open/closed estuaries. Water level decreases have been shown to increase öospore production (Kautsky, 1990; Casanova and Brock, 1996; Asaeda *et al*., 2007). The six sites sampled in both estuaries can be exposed for 2 months at a time or can be flooded by 1 m water depth within a one month period due to the variability in mouth condition. These regular changes in water level may stimulate öospore production resulting in the high densities found in this study. Sediment seed density was not correlated to the extant vegetation at the time of sampling. In highly variable environments such as temporarily open/closed estuaries, seeds may remain dormant until their specific germination requirements are met. For example *Stukenia pectinata* never occurred in the extant vegetation in either estuary during this study period, but has been observed in the past. The lack of stable water level and lower salinity (< 15 ppt) is the likely cause. Long periods of mouth closure in the West Kleinemonde (11 months prior to the first sampling period in March 2006) resulted in large beds of *Ruppia cirrhosa* forming (up to 100 % cover) and replenishment of the seed reserves. By contrast the East Kleinemonde Estuary opens on average 2.6 times per year (Van Niekerk *et al*., 2008) and during this study remained closed for only six months. As a result submerged macrophytes only formed about 8 % cover. *Sarcocornia tegetaria* reached up to 98 % cover in the East Kleinemonde due to more available habitat, resulting in seed bank replenishment

In contrast with other studies (Grillas *et al*., 1993; Bonis and Lepart, 1994; De Winton *et al*., 2000) seed density did not differ significantly with sediment depth. High seed densities at lower sediment depths will still provide a regeneration source in the event of sediment disturbance. For all seed counts numbers were highly variable and although this could be attributed to sample size the dimensions used in this study were similar to that used in other studies (Grillas *et al*., 1993; Brock *et al*., 1994; De Winton *et al*., 2000). Despite this it is recommended that future seed bank analysis in temporarily open/closed estuaries use larger sub sample sizes for assessment since extrapolation of seed numbers to $m^{-2}$ of smaller sub samples introduces variation. In addition larger sub sample sizes may be more appropriate for larger seeds such as *Ruppia cirrhosa* and *Stukenia pectinata*.

The rapid emergence of *Sarcocornia tegetaria* (three days) is similar to rates (5 to 10 days) reported for other intertidal halophytes (Tölken, 1967; Naidoo and Naicker, 1992; Rubio-Casal *et al*., 2002; Redondo *et al*., 2004; Naidoo and Kift, 2006; Shaw *et al*., 2008). Rapid germination provides a competitive advantage for space in erratic and unstable habitats (Casanova and Brock, 1996; Brock and Rogers, 1998). Although germination was significantly reduced at 35 ppt, 57 % germination still occurred. *Chara vulgaris* had maximum germination at 17 ppt. In both estuaries, during open conditions salinity ranged between 0.7 and 35 ppt, compared to 15 to 23 ppt during closed conditions (Van Niekerk *et al*., 2008). Although *Stukenia pectinata* did not germinate during the trial period, it has occurred in these estuaries together with *Ruppia cirrhosa* when salinity was 15 ppt and below. The slower emergence of submerged species (18 d) and the lack of germination of other species present in the seed bank may be an indication that specific germination requirements or physiological dormancy exists. Charophytes in temporary water bodies require a period of desiccation as well as intermittent moistening as a prerequisite for germination (Harwell and Havens, 2003; Casanova and Brock, 1996). Desiccation also stimulates germination in *Ruppia* (Kantrud, 1991). Fresh öospores of *Chara vulgaris* require a 60 d “after ripening” period to increase germination percentages (Sederias and Colman, 2007). Further studies are needed to investigate the germination requirements of these species so that the seedling emergence method can be used to measure seed density. In conclusion the high seed numbers found for the dominant species indicates that despite the variable nature of these estuaries, adequate seed reserves are available to ensure habitat persistence under the present open/closed mouth conditions of the East and West Kleinemonde estuaries.

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# CHAPTER 5

# LONG TERM DYNAMICS AND PREDICTION OF HABITAT AVAILABILITY

## *Abstract*

Temporarily open/closed estuaries are known to shift between abiotic states, either due to stochastic events such as floods or due to longer decadal cycles such as those associated with rainfall. Analysis of monthly macrophyte cover data for a 5 year period indicated that macrophytes in the East Kleinemonde Estuary in South Africa showed significant changes in cover when water level exceeded 1.55 m amsl and salinity 30 ppt. Four abiotic states could be identified based on these thresholds namely State A, an open tidal state with intertidal and supratidal salt marsh, reeds and sedges and patches of submerged macrophytes and macroalgae. State B occurred when the mouth of the estuary closed and water level remained low, below 1.55 m amsl. Macrophyte cover was similar to that in State A. In State C water level exceeded 1.55 m amsl and inundated intertidal salt marsh which was slowly replaced by patchy growth of submerged macrophytes such as *Ruppia cirrhosa*. Supratidal salt marsh and reeds and sedges continued to grow providing the water level did not exceed 1.77 m amsl. In State D water level remained high but salinity was above 30 ppt resulting in a loss of submerged macrophytes, reeds and sedges and supratidal salt marsh; macroalgae were dominant. During the 5 year study period the East Kleinemonde Estuary remained in State A for 9 % of the time, State B for 8 % of the time, State C for 48 % of the time and in State D for 35 % of the time. This threshold method was integrated into the methodology for determining the freshwater requirements of South African estuaries. This was done by filtering simulated hydrological data for past/reference conditions, present and two future freshwater inflow scenarios for the 1.55 m water level. The simulated hydrological data covered a 72-year period. Water level was converted to water volume using a digital elevation model of the estuary. Results showed that the estuary has changed little from past to present conditions with an increase of 1 % in State C and D having occurred. Under the future inflow scenario 1, State C and D (closed, high water states) would increase, whereas under scenario 2 there would be an increase in State B (closed and low water level). Dominant macrophyte habitats were identified for each of these states. The benefit of using this threshold method is that it is rapid, requiring only information on the elevation range of the main habitats in an estuary. An additional approach was used to quantify available habitat for different water level conditions using a spatial model built in Model Builder (ArcGIS 9.3.1). The model intersects a habitat map with a bathymetric map and produced empirical equations of water level versus potential habitat development (ha). This method can be applied to any other estuary providing these two maps are available. Data analysis of the 5 year cover data for macrophytes provided information on the time required to achieve maximum habitat cover and it ranged from 0.3 to 0.6 months for emergent habitat to 0.3 to 11 months for submerged habitat. Both methods can be integrated in the current freshwater requirement methodology used for South Africa estuaries and can be used for all levels of assessment; from rapid to comprehensive studies. These methods can easily be applied to other estuaries and will assist managers in determining macrophyte response to abiotic states.

**Keywords**: spatial temporal habitat availability freshwater requirements model threshold macrophyte response

## *5.1 Introduction*

Threshold effects have been known to cause regime or state shifts in many ecosystems (Andersen *et al*., 2008). Shifts from turbid to clear water states have been well documented in lakes and coastal systems internationally (Healy, 1997; Scheffer *et al*., 2003; Folke *et al*., 2004; Alber *et al*., 2008; Jeppensen, *et al*., 2007; Zhang *et al*., 2010). Florida Bay switched from a clear-water seagrass-dominated state to a turbid, phytoplankton-dominated state. This shift was thought to be due to a combination of factors including increased hurricane frequency, reduced freshwater inflow, increased nutrient input, the removal of large grazers and the construction of hard structures (Folke *et al*., 2004). The loss of seagrasses through shading by phytoplankton, resulted in sediment destablisation and sediment resuspension by wind and wave action. Temporarily open/closed estuaries shift between different abiotic states due to variation in mouth status and freshwater inflow (Whitfield *et al*., 2008). These shifts may either be due to stochastic events such as floods and droughts which can cause a breach and a change from a closed to an open abiotic state or due to reduced freshwater inflow from increased abstraction (Healy, 1997; Largier *et al*., 1997; Bachelet, 2000; Lamptey and Armah, 2008). In Lake Ellesmere, a TOCE in New Zealand, losses of submerged macrophytes were linked to severe storm activity where large beds prior to a storm in 1968 never recovered. Submerged macrophytes only occurred in isolated patches and they were replaced due to an increase in turbidity and high phytoplankton biomass (Schallenberg *et al*., 2010). Decadal climatic cycles can also cause abiotic shifts such as in the St Lucia Estuary, South Africa. Weather in the St Lucia area is known to fluctuate in a 10 year rainfall cycle and this results in shifts of primary habitats (Taylor *et al*., 2006). This estuary is phytoplankton dominated during hypersaline conditions but changes to submerged macrophytes during higher rainfall periods when there is lower salinity. In the Seekoei Estuary in the Eastern Cape, Whitfield (1989) documented a fish kill due to extreme hypersaline conditions (98 ppt) following freshwater abstraction from the river during a drought. The hypersaline state thus represents the most extreme abiotic state in estuaries when there is a loss in biodiversity and biomass. The type of mouth breaching also determines the salinity and biotic responses. For example a deep mouth breaching following a large river flood tends to result in major tidal inputs of marine water prior to mouth closure and higher salinity (15 to 25 ppt) (Whitfield *et al*., 2008). The mouth of the East Kleinemonde Estuary can stay open for between 1 to 28 days with an initial salinity above 30 ppt. *Ruppia cirrhosa* is the dominant submerged macrophyte in the estuary following this type of breaching. By contrast, a shallow mouth breaching with reduced tidal exchange during the open phase often leads to a much lower salinity regime at the time of mouth closure (5 to 15 ppt). *Stukenia pectinata* is dominant in the estuary following this shallow breach as it grows best in these conditions.

In freshwater inflow studies specialists must provide input on the effects of reduced flows on the ecology, social and economic aspects of an estuary (Estevez, 2002; Turpie *et al*., 2008). Methods used to model and predict responses to altered freshwater inputs include inflow based methods (Alber and Flory, 2002; Flannery *et al*., 2002), resource based methods (Mattson, 2002; Montagna *et al*., 2002; Robins *et al*., 2002; Halliday *et al*., 2003), condition based methods (Kimmerer and Schubel, 1994; Jassby *et al*., 1995; Alber, 2002) and holistic based methods (Peirson *et al*., 2001; Taljaard *et al*., 2004; Richter *et al*., 2005;

Gippel *et al*., 2008). The South African method (Taljaard *et al.*, 2004) takes a holistic approach in that it relates all biota to different abiotic states. Abiotic states are described for an estuary based on simulated monthly inflow volumes for the present state and the reference state, as well as the prediction of future inflow volumes under different future scenarios. These abiotic states are linked to macrophyte responses through knowledge of species environmental tolerance limits and using this information, thresholds of potential concern, measurable end points related to specific abiotic or biotic indicators, are set. Thresholds of potential concern act as early warning signals and by setting these thresholds, various water abstraction activities can be altered to ensure the ecological functioning of the macrophytes in the estuary remains acceptable. In the Sundays Estuary for example, salinity thresholds were set for the upper reaches so that salinity would not be greater than 5 ppt. This is because *Stukenia pectinata* occurs in the upper reaches and an increase in salinity above 10 ppt for three months will significantly reduce the cover thereof.

Attempts have been made to model and predict abiotic and biotic responses to altered freshwater inflows into estuaries (Wortmann *et al*., 1998; Slinger, 2000; Mattson, 2002; Taylor *et al*., 2006; Wolanski *et al*., 2006; Wolanski, 2007; Kim and Montagna, 2009). However biotic responses can sometimes be difficult to predict because they are not only controlled by exogenous factors such as rainfall and climate, but also through endogenous controls such as feedback mechanisms and time lags. Loss of submerged macrophyte beds for example, result in an increase in sediment re-suspension and reduced light availability (Van Nes *et al*., 2003; Burkholder *et al*., 2007). Furthermore the slow response of habitats to changing abiotic conditions is well documented. *Ruppia cirrhosa* had a time lag of two months in terms of response to water depth and a time lag of four months before responding to conductivity changes (Carruthers *et al*., 1999; Calado and Duarte, 2000; Biber *et al*., 2004). Furthermore macrophytes have a wide tolerance range and the uses of abiotic states with ranges of environmental conditions are more appropriate for predicting macrophyte responses in estuaries.

The three abiotic states identified for the East Kleinemonde Estuary from the RDM study of Van Niekerk *et al*., 2008 were:

1. Intermittently open/closed driven by high flow events > $0.3 \times 10^6$ m$^3$

2. Intermittently open/closed driven by persistent low flow periods < 0.3 x $10^6$ $m^3$ and cumulative inflows > 0.3 x $10^6$ $m^3$
3. Closed mouth with flow volume < 0.3 x $10^6$ $m^3$ and cumulative inflows < 0.3 x $10^6$ $m^3$

Analysis of daily mouth conditions between 1993 and 2003 along with simulated monthly inflow volumes showed that the East Kleinemonde Estuary remained in State 3 for 78.4 % of the time. These states are not dissimilar to the five phases proposed for the East Kleinemonde Estuary by Whitfield *et al*. (2008) with the addition of an outflow phase and an overwash phase. Although phases are taken to represent short periods of time, probably less than one month, the closed phase identified by Whitfield *et al*. (2008) could last a few years.

Given these abiotic states of the East Kleinemonde Estuary and acknowledging time lags in the response of biota, this study assessed the abiotic drivers of macrophyte change and determined if there were thresholds, specifically in terms of water level and salinity. For example, at which water level is intertidal salt marsh in the East Kleinemonde Estuary most affected by inundation and likely to cause a shift to submerged macrophytes? The time taken for these changes to take place was also determined. Furthermore, a rapid method was proposed using these thresholds that could be integrated into the current South African freshwater requirement methodology for estuaries (Taljaard *et al*., 2004) to determine the frequency of abiotic states based on these thresholds and the macrophyte response. In this way the dominant macrophyte habitats can be determined for present, past and future freshwater inflow scenarios. A second method was proposed to quantify the spatial habitat available under different water level scenarios using a spatial model that was produced in Model Builder in ArcGIS 9.3.1. The model is modified from the approach used by Sharma *et al*. (2009). The model works by intersecting a macrophyte habitat map with water level to determine the amount of habitat flooded or exposed. Empirical equations of habitat availability to water level (m amsl) were produced for the East Kleinemonde Estuary but can be used in other TOCEs providing a macrophyte habitat map and a bathymetric map is available. Quantification of available habitat is useful when considering habitat connectivity to higher trophic levels, for example the use of flooded salt marsh by fish species (Able, 2005; Becker and Laurenson, 2007).

## *5.2 Materials and Methods*

### *5.2.1 Abiotic drivers*

Abiotic water column variables that were measured were salinity (ppt), temperature (°C) and Secchi depth (m). These were measured at five stations along the length of the estuary using a YSI 650 MDS Multiprobe and a Secchi disc. Data were averaged for analysis because there were no longitudinal or vertical gradients throughout the study. Average daily water level data were obtained from a water level recorder located beneath the R72 Bridge where water level was recorded every ten minutes (station number P4T002, www.dwaf.gov.za). Elevation profiles for the three transects were measured using a Wild Heerbrugg Dumpy Level and referenced against mean sea level. This was used to calculate the extent of habitat range and the influence of water level through inundation for each sampling period. Macrophyte response was assessed monthly along three permanent transects from March 2006 to January 2010. Species cover abundance (%) was measured within duplicate 1 $m^2$ quadrats that were placed every 5 m along the length of each transect and data were averaged for each transect (n = 144). Cover data was averaged along the elevation range of each habitat, as done in Chapter 2.2.3.

### *5.2.2 Identification of dominant macrophyte habitats for different abiotic states*

The range of elevation for each habitat was determined for the five year period, i.e. the minimum and maximum elevation at which macrophyte species occurred. The elevation where maximum cover occurred for macrophyte habitats (intertidal salt marsh, supratidal salt marsh, reeds and sedges, submerged macrophytes) over the study period was taken as the threshold water level since above this level significant inundation would occur. The average water level based on each habitat was used. This method ensures a rapid assessment of the threshold water level in other TOCEs as it simply requires a survey of a number of transects within an estuary and an assessment of the elevation range of habitats that are present.

In the determination of the freshwater requirements of the East Kleinemonde Estuary, monthly freshwater inflows (volume) and breaches for the period 1920 to 2002 were simulated, representing the present condition of the estuary (Appendix 1). The purple areas represent State 1 (high flow breaching), the light blue areas State 2 (low flow breaching) and the white areas State 3 (closed state). With the use of a water volume to water level equation produced by Theron and Bornman (2008), the threshold water level was converted to an inflow volume. The present state (Appendix 1), reference state (Appendix 2) and two future inflow scenarios (Appendix 3 and 4) were filtered for this volume and the frequency of time the estuary occurred above and below this volume. Dominant macrophyte habitats associated with the different water level conditions were identified.

The water volume to monthly water level equation was:

$$y = 3.198\,x^3 - 6.634x^2 + 6.288x + 0.056$$

y = water level (m amsl)
x = simulated monthly water volume (million $m^3$)

This equation is specific for the East Kleinemonde Estuary but can be recalculated for any other TOCEs using a digital elevation model produced from a bathymetric map. For the calculation of macroalgal habitat versus water level, a value of 25 % of water area was taken to represent available habitat. This is the average macroalgal cover for each of the transects monitored over the 5 year period.

### *5.2.3. Calculation of time required to achieve maximum macrophyte habitat cover*

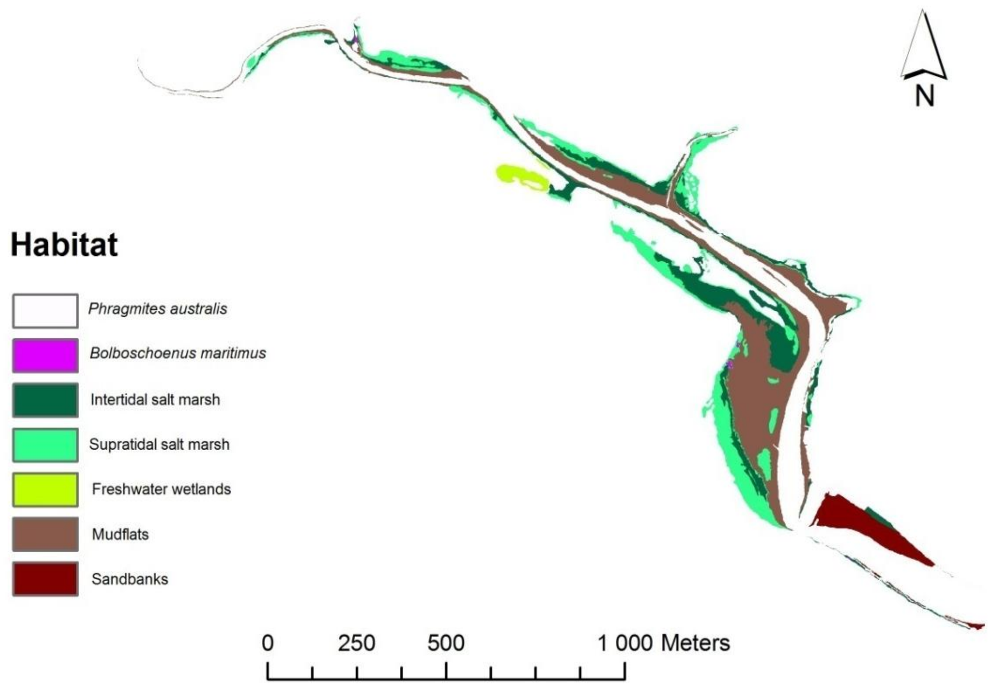
The rate of macrophyte habitat growth was calculated by averaging the rate of monthly cover change for each transect and then determining the numbers of months required to achieve 100 % cover. Results were based on the monthly quadrat cover change for the period March 2006 to January 2010. Average values for each transect were used instead of on a quadrat to quadrat basis as misplacement of a quadrat could result in a large sampling error and misinterpretation of the results (n = 48).

### *5.2.4 Spatial habitat availability model*

Macrophyte habitat availability (ha) for water levels ranging from 0.5 to 3 m amsl was determined using a spatial model written in Model Builder, ArcGis 9.3.1. For all macrophyte habitats the model works by intersecting a macrophyte habitat map with a digital elevation model based on a selected water level (Figure 5.1).

The habitat map was completed under open mouth conditions in 2006 and the survey included the assessment of photographic records, spatial data (GPS and ArcPad® version 7) and the collection of plant material for identification (Riddin and Adams, 2008 in Van Niekerk *et al*., 2008). Elevation profiles for 21 transects were measured using a Wild Heerbrugg Dumpy Level and the positions recorded using a GPS with ArcPad® software. Data were interpolated to raster images using Kriging in 3D Analyst in ESRI ArcMap™ (Version 9.3.1). The area of habitat covered by selected water level produced empirical equations of area of habitat exposed (ha) versus water level (m amsl). These are specific to the East Kleinemonde Estuary but the model can be used for other TOCEs, the only requirements would be a habitat distribution map and a bathymetric map because the analysis functions such as "extract" and "clip" are generic. The projection system of the macrophyte and bathymetric map was Transverse Mercator, Longitude 27, geographic co-ordinate system GCS_Clarke_1880 and datum WGS84. Each map was converted into a raster file for use in Model Builder. Model Builder is an application used to create, edit and manage models. Models are workflows that string together sequences of geoprocessing actions such as extracting a selected water level from a bathymetric map, with the output of one action becoming the input of another. It therefore creates a workflow of a sequence of tools or processes. For example in Figure 5.2, the model for submerged macrophytes, the habitat map (ek_all veg_RECT) was masked by a selected water level, represented by the "where clause", from the bathymetric map (ek_contours). The potential area for submerged macrophyte development was taken as flooded intertidal salt marsh and mudflats and these habitats were extracted by their attributes, i.e. their habitat code in the habitat map, and a new habitat map was produced showing the new water level and potential submerged macrophyte habitat (subm%n%).

a)




b)

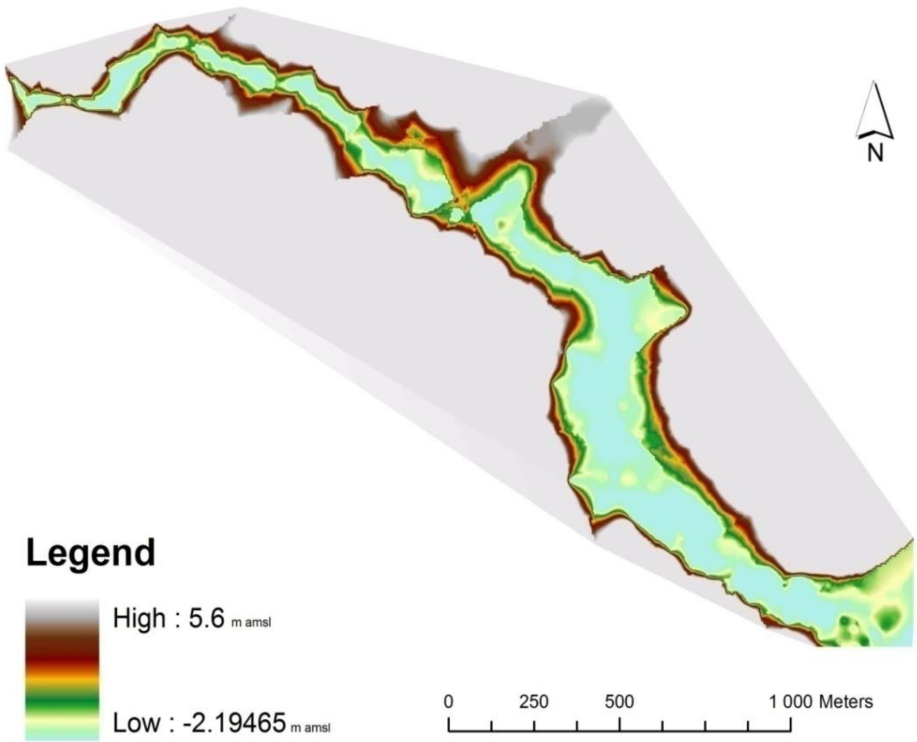



Figure 5.1. The data sets used to determine habitat availability, a) macrophyte distribution map (Riddin and Adams, 2008) and (b) bathymetric map (Theron and Bornman, 2008 in Van Niekerk *et al*., 2008) of the East Kleinemonde Estuary.

Also included in the submerged macrophyte model is a surface volume calculation, which gives the volume of water to 1.5 m below the selected water level. This 1.5 m represents the lower depth limit of submerged macrophytes despite the average Secchi depth over the past 5 years of 1.28 m. *Ruppia cirrhosa* was observed in both the East and West Kleinemonde estuaries in water up to 1.5 m deep. Literature confirms a lower limit of 1.5 m (Verhoeven, 1979; Costa and Seeliger, 1989; Calado and Duarte, 2000; Da Silva and Asmus, 2001; Obrador and Pretus, 2010), although Carruthers and Walker (1999) have shown that these macrophytes can occur at depths of up to 2.9 m in the Wilson Inlet. Generally however submerged macrophytes have a minimum light requirement of 5 to 29 % of surface light (Duarte, 1991). The depth limit of *Ruppia cirrhosa* however depends on the depth of the estuary and the average height of the canopy of the submerged macrophyte stands (Calado and Duarte, 2000). This is because self-shading increases as the stand develops and reduces the light in near-bottom zones. Water volume available for submerged macrophyte habitat was included because this represents their potential vertical distribution through the water column. The maximum macroalgal habitat was calculated as 25 % of the total water surface area from the digital elevation model calculated for each water level.

The empirical equations of habitat versus water level produced from this spatial model was used to calculate the maximum available macrophyte habitat under two freshwater inflow scenarios that were identified in the East Kleinemonde RDM study (Van Niekerk *et al*., 2008):

1. Scenario 1 involved the construction of a new dam on the main river resulting in a reduction in mean annual runoff (MAR) from 2.856 M $m^3$ $annum^{-1}$ in the reference state to 2.409 M $m^3$ $annum^{-1}$, a 16 % reduction. This would lead to low flows of < 0.12 $m^3$ $s^{-1}$ that would occur for 88.6 % of the year and there would be a 3.5 % reduction in floods. Open mouth conditions were estimated to change from 2.6 months of the year to 2 months, or a 13 % reduction in the open mouth state.
2. Scenario 2 involved the construction of an off-channel reservoir with intermittent pumping from the main river, leading to a 12 % reduction in MAR. This would result in low flows of < 0.12 $m^3$ $s^{-1}$ occurring for less than 87.5 % of the year and a 5.3 % reduction in floods. The open mouth state was estimated to change from 2.6 months of the year to 2.3 months, a reduction of 11.5 %.

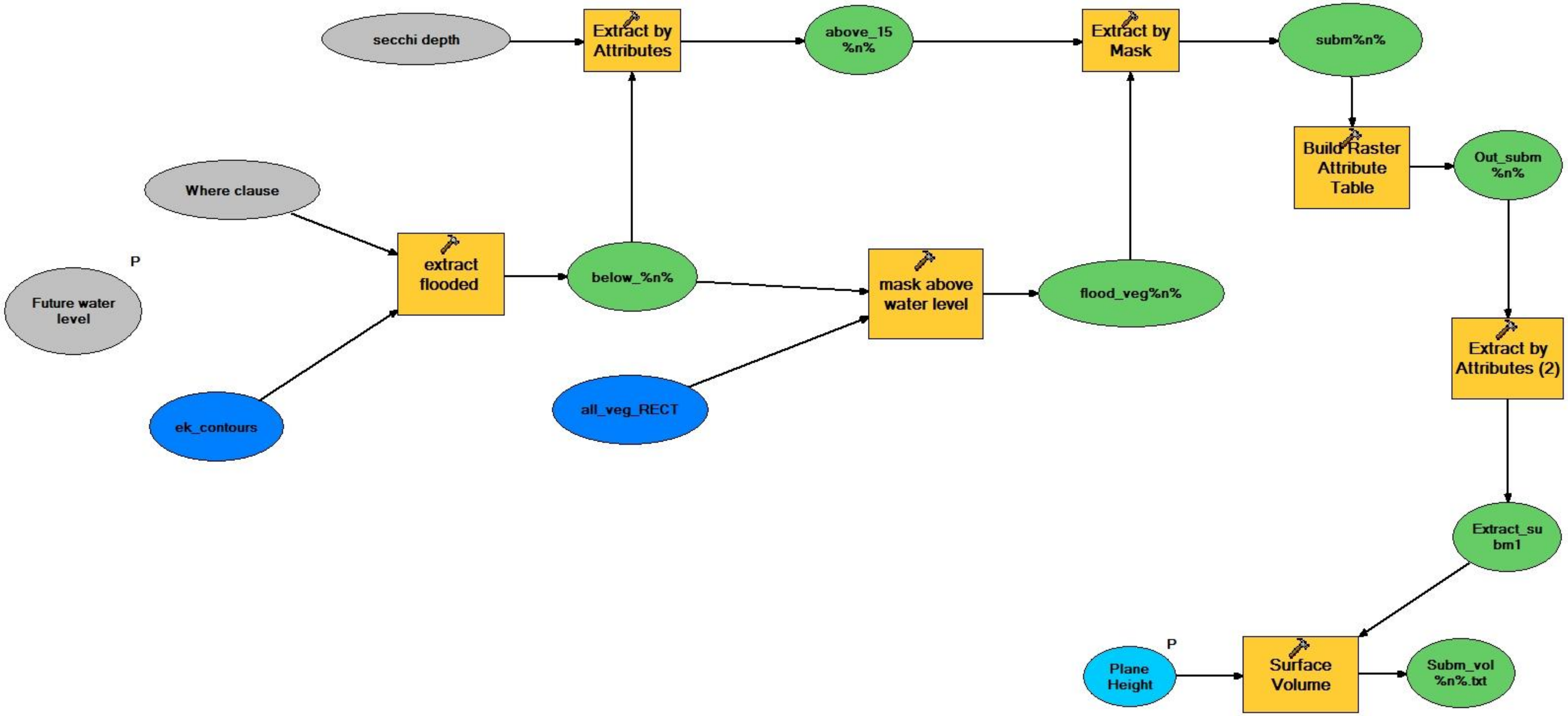



Figure 5.2. Submerged macrophyte spatial habitat model developed in Model Builder, ArcGis 9.3.1. Grey ovals represent parameter inputs (e.g. water level), dark blue ovals represent input layers (macrophyte and bathymetric map), green ovals represent intermediate and final outputs and yellow rectangles represent processes (e.g. mask habitat with water level, extract habitat area).

Using the water volume to water level equation of Theron and Bornman (2008) the predicted monthly inflow volumes (Table 5.1) were converted to monthly water level. These water level data were then used as input parameters in the spatial model to calculate available habitat.

Table 5.1. Simulated monthly volumes (million $m^3$) in the East Kleinemonde Estuary for future Scenario 1 and 2.

| | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Oct** | **Nov** | **Dec** | **Total breaches** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Scenario 1 | 0.272 | 0.275 | 0.284 | 0.287 | 0.269 | 0.262 | 0.269 | 0.264 | 0.246 | 0.254 | 0.259 | 0.275 | 2 |
| Scenario 2 | 0.256 | 0.259 | 0.256 | 0.266 | 0.262 | 0.267 | 0.263 | 0.272 | 0.249 | 0.242 | 0.241 | 0.252 | 2 |

### *5.2.5 Data analysis*

Multivariate analysis, Detrended Canonical Correspondence Analysis (DCCA), was used to determine the drivers of macrophyte cover change for the 5 year data period. Prior analysis of the length of the gradients, which measures beta diversity, indicated the suitability of DCCA (Jongman *et al*., 1995). If this length was above 4, then DCA, CA and CCA could be used (Lepš and Smilauer, 2003). Monte Carlo permutation tests (499) were performed to assess the significance of the canonical axis showing the relationship between species and the environmental variables. Because of the possible presence of environmental covariates, detrending by polynomials was performed rather than by segments (Lepš and Smilauer, 2003). The statistical results were displayed in a table below each DCCA ordination diagram. CANOCO for Windows (Version 4.5, Ter Braak and Smilauer, 2002) was used for DCCA analysis and CANODRAW (Version 4, Ter Braak and Smilauer, 2002) was used to plot the DCCA results. The environmental variables were plotted as arrows originating from the centre of the graph. The origin represents the mean value of each separate variable and the direction of the arrow line represents an increase in the value of that particular variable. The length of the environmental arrow indicates the importance of the variable and is equal to the multiple correlation of the variable with the displayed ordination axes. The influence of the abiotic drivers on the change in monthly cover was also determined using non parametric Spearman rank order correlation in Statistica Version 7.

## 5.3 Results

### 5.3.1 Abiotic drivers

Abiotic conditions showed greater variability over the 5 year period as opposed to the 1 year study (Chapter 2) with a greater fluctuation in salinity occurring over 5 years. Salinity ranged between 14.8 and 38.5 ppt (average = 25.5 ± 0.8 ppt, Figure 5.3) and differed significantly from the 19.9 to 27.2 ppt (average = 23.2 ± 0.6 ppt) measured in the one year study ($P < 0.05$). The average water level over the 5 year period was higher (1.8 ± 0.1 m compared to 1.0 ± 0.2 m amsl) and there was also a higher maximum water level of 2.57 m amsl compared to 2.2 m amsl in the 1 year period. Neither water temperature, rainfall nor Secchi depth varied significantly between the 1 and 5 year data. Secchi depth was on average 1.28 m.

Change in macrophyte cover was discussed in Chapter 3 and Figure 5.3 only includes an extra month's sampling data, January 2010. The numerical results of the Detrended Canonical Correspondence Analysis (DCCA) for the cover data are shown in Table 5.2 below the ordination diagram (Figure 5.4). The first canonical axis (horizontal) described 54 % of the variation of the species – environment relationship. This axis was negatively correlated with water level and positively correlated with air temperature. The strongest negative correlation (-0.82) was between the first canonical axis and water level for four months prior to sampling. Water level at the time of sampling (-0.64), one month (-0.69), two months (-0.73) and three months (-0.78) prior to sampling negatively affected cover of the emergent species. Cover of the submerged macrophyte *Ruppia cirrhosa* was positively correlated with high water level.

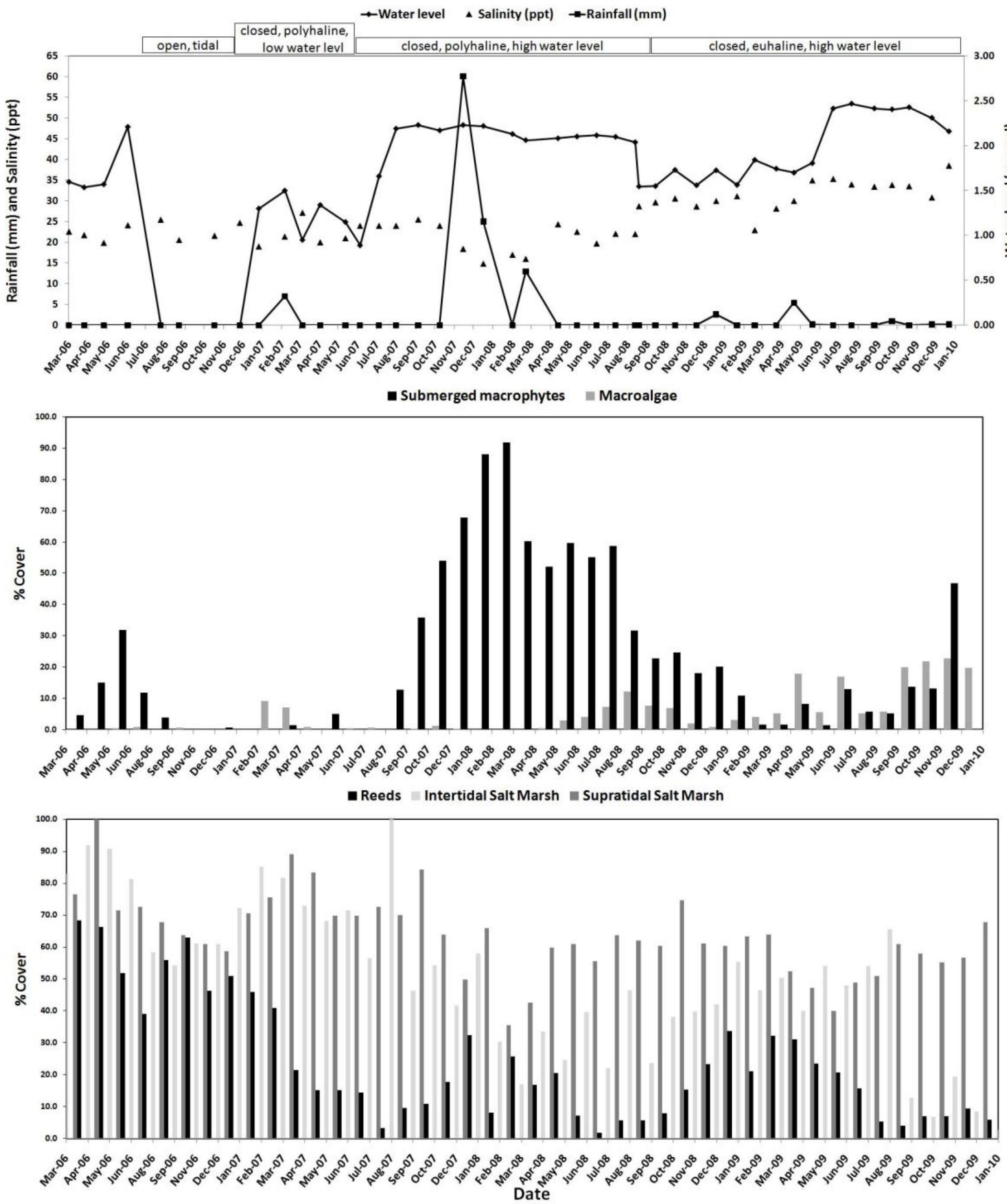



Figure 5.3. Long term change in abiotic (a) and and macrophyte cover data (b and c) for the period March 2006 to January 2010 in the East Kleinemonde Estuary. Graph a shows the abiotic states identified in Section 5.3.2.

The second canonical axis (vertical) described 70 % of the variation of the species – environment relationship. This axis was positively correlated with salinity, including salinity 4 months prior to sampling and was negatively correlated to rainfall. The strongest positive correlation (0.48) was between the second canonical axis and salinity at the time of sampling. Both salinity and air temperature were positively correlated with the cover of macroalgae (Figure 5.4). As salinity increased *Chara vulgaris* was replaced by *Ruppia cirrhosa* and finally by macroalgae (Figure 5.4). *Halophila ovalis* was not recorded other than in 2006 for the short term study and in isolated patches in the months of April 2007, February 2008 and December 2009 for the long term study. *Zannichellia palustris* L. was only observed in December 2007, January and February 2008.

Spearman correlation also confirmed the negative correlation of water level with intertidal macrophytes (*Sarcocornia tegetaria*, *Sporobolus virginicus, Salicornia meyeriana*), reeds and sedges (*Phragmites australis*, *Bolboschoenus maritimus*) and submerged macrophytes (*Ruppia cirrhosa* and *Chara* spp.) (Table 5.3). Salinity was negatively related to cover of supratidal species but positively correlated to an increase in macroalgal cover. Whereas in Chapter 2, intertidal salt marsh cover was negatively affected by inundation for 3 months, the 5 year data showed that inundation for 4 months significantly affected cover ($r^2$ = -0.78, $P < 0.05$). This indicated that the habitat was more tolerant of longer inundation than previously determined. However the depth of inundation was also important. Inundation for 1 month or longer negatively influenced the cover of the reeds and sedges. Submerged macrophyte habitat was positively influenced by inundation for 1 to 2 months supporting the findings of Chapter 2 and 4.

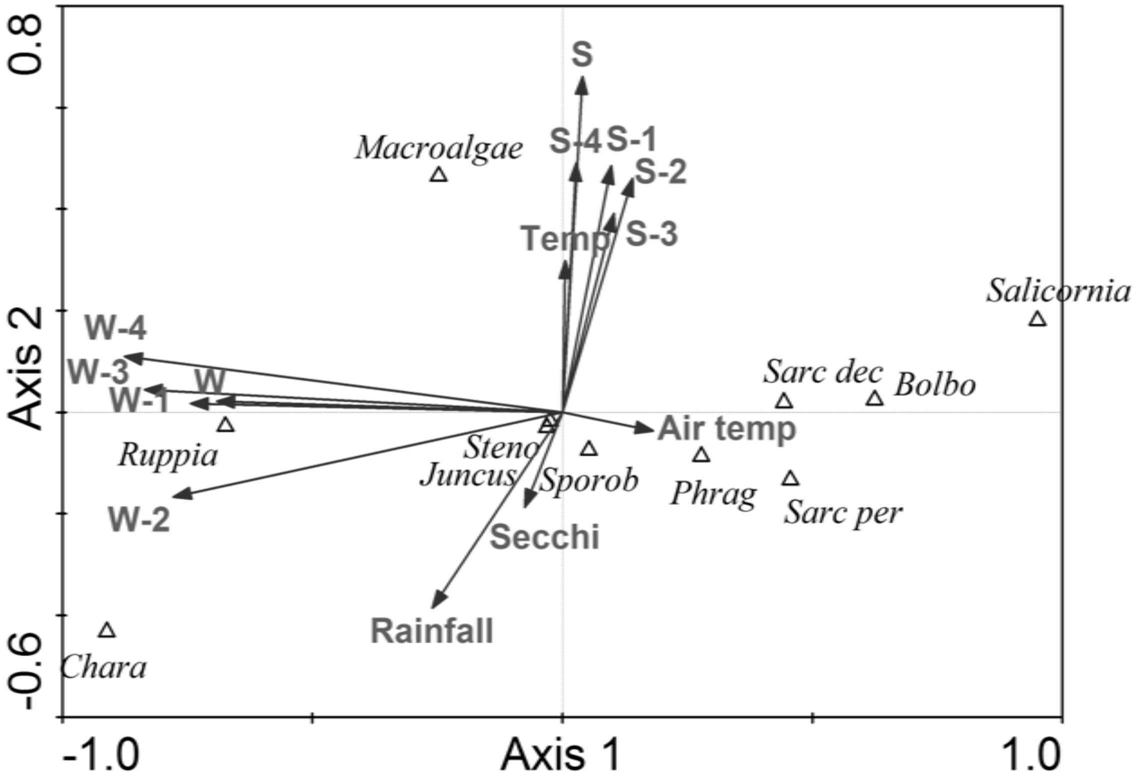



Figure 5.4. DCCA ordination plot of macrophyte species cover and environmental data for the 5 year study period in the East Kleinemonde Estuary. The arrows represent each environmental variable pointing in the direction of its maximum change. Abbreviations: Chara = *Chara vulgaris*, Ruppia = *Ruppia cirrhosa*, Phrag = *Phragmites australis*, Bolbo = *Bolboschoenus maritimus*, Sarc per = *Sarcocornia tegetaria*, Salicornia = *Salicornia meyeriana*, Steno = *Stenotaphrum secundatum*, Sporob = *Sporobolus virginicus*, Sarc dec = *Sarcocornia decumbens*, Juncus = *Juncus kraussii*, Temp = water temperature, W = water level, W-1 = water level for one month preceding sampling, W-2 = two months preceding sampling, W-3 = three months preceding sampling and W-4 = water level four months preceding sampling. S = salinity, S-1 = salinity one month preceding sampling, S-2 = salinity two months preceding sampling, S-3 = salinity three months preceding sampling and s_4 = salinity four months preceding sampling.

Table 5.2. Summary of the DCCA results for the East Kleinemonde macrophyte species and environmental data for the 5 year study period (P = 0.002).

| | Axis 1 | Axis 2 | Axis 3 | Axis 4 | Total inertia |
|---|---|---|---|---|---|
| Eigen values | 0.186 | 0.055 | 0.019 | 0.005 | 0.497 |
| Species-environment correlations | 0.940 | 0.729 | 0.804 | 0.508 | |
| Cumulative percentage variance | | | | | |
| Of species data | 37.3 | 48.5 | 52.2 | 53.2 | |
| Of species environment relation | 53.9 | 70 | 75.4 | 76.9 | |
| Sum of all eigenvalues | | | | | 0.497 |
| Sum of all canonical eigenvalues | | | | | 0.344 |

Table 5.3. Spearman correlation analysis ($r^2$) showing the significant abiotic parameters influencing macrophyte cover change in red ($P < 0.05$).

| | *Sarcocornia tegetaria* | *Salicornia meyeriana* | *Sporobolus virginicus* | *Juncus krausii* | *Stenotaphrum secundatum* | *Sarcocornia decumbens* | *Phragmites australis* | *Bolboschoenus maritimus* | *Chara* spp. | *Ruppia cirrhosa* | Macroalgae |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Salinity | -0.27 | -0.18 | -0.36 | -0.65 | -0.6 | -0.06 | -0.37 | -0.22 | -0.54 | -0.21 | 0.62 |
| Salinity-1 | -0.19 | -0.11 | -0.37 | -0.67 | -0.55 | -0.09 | -0.34 | -0.18 | -0.53 | -0.26 | 0.53 |
| Salinity-2 | -0.16 | -0.02 | -0.38 | -0.65 | -0.63 | 0.06 | -0.22 | -0.12 | -0.48 | -0.28 | 0.52 |
| Salinity-3 | -0.15 | -0.03 | -0.41 | -0.53 | -0.66 | -0.07 | -0.14 | -0.09 | -0.37 | -0.29 | 0.42 |
| Salinity-4 | -0.19 | 0.02 | -0.45 | -0.51 | -0.56 | -0.2 | -0.17 | -0.12 | -0.25 | -0.29 | 0.38 |
| Water level | -0.41 | -0.36 | -0.25 | -0.35 | -0.27 | -0.55 | -0.47 | -0.64 | 0.39 | 0.42 | 0.44 |
| Water level -1 | -0.47 | -0.45 | -0.29 | -0.21 | -0.29 | -0.53 | -0.51 | -0.66 | 0.43 | 0.44 | 0.42 |
| Water level -2 | -0.55 | -0.61 | -0.27 | -0.21 | -0.17 | -0.59 | -0.47 | -0.79 | 0.43 | 0.44 | 0.35 |
| Water level -3 | -0.64 | -0.63 | -0.28 | -0.12 | 0.01 | -0.5 | -0.37 | -0.74 | 0.42 | 0.57 | 0.45 |
| Water level -4 | -0.78 | -0.6 | -0.19 | -0.1 | 0.01 | -0.66 | -0.43 | -0.74 | 0.4 | 0.66 | 0.4 |
| Temperature | -0.09 | 0.39 | -0.32 | 0.17 | -0.33 | -0.08 | 0.12 | 0.19 | 0.13 | -0.06 | -0.06 |
| Secchi depth | 0.1 | -0.32 | 0.19 | -0.23 | 0.14 | -0.06 | -0.13 | -0.16 | 0.02 | 0.1 | -0.03 |
| Air Temperature | 0.18 | 0.44 | -0.13 | 0.25 | -0.28 | -0.01 | 0.29 | 0.43 | 0.12 | -0.15 | -0.22 |
| Rainfall | -0.18 | -0.05 | 0.05 | 0.36 | 0.12 | -0.22 | 0.03 | -0.12 | 0.42 | 0.15 | -0.19 |

### *5.3.2 Identification of dominant macrophyte habitats for different abiotic states*

The average elevation range of each macrophyte habitat is given in Table 5.4. This elevation represents the threshold water level at which 50 % of the transect was inundated and therefore the water level at which a shift in habitat was likely to occur. The average threshold water level at which significant changes for all macrophyte cover occurred was found to be 1.55 m amsl (Table 5.4).

Table 5.4. The average elevation (m amsl) at which species and habitat occurred, based on three transects over 48 sampling periods. The equivalent water volume of the estuary is also shown, calculated using the equation of Theron and Bornman (2008).

| | Transect 1 | Transect 2 | Transect 3 | Average elevation for species | Average elevation for habitat | Volume (million $m^3$) |
|---|---|---|---|---|---|---|
| <u>Intertidal salt marsh</u> | | | | | 1.50 | 0.314 |
| *Sarcocornia tegetaria* | 1.38 | 1.46 | 1.62 | 1.49 | | |
| *Sporobolus virginicus* | 1.36 | 1.50 | 1.66 | 1.51 | | |
| <u>Supratidal salt marsh</u> | | | | | 1.77 | 0.402 |
| *Sarcocornia decumbens* | | 1.53 | 1.94 | 1.74 | | |
| *Stenotaphrum secundatum* | | 1.50 | 2.07 | 1.79 | | |
| *Juncus kraussii* | 1.50 | 1.77 | 2.07 | 1.78 | | |
| <u>Reeds and sedges</u> | | | | | 1.52 | 0.32 |
| *Phragmites australis* | 1.05 | 1.73 | | 1.39 | | |
| *Bolboschoenus maritimus* | | 1.66 | 1.64 | 1.65 | | |
| <u>Submerged macrophytes</u> | | | | | 1.39 | 0.282 |
| *Ruppia cirrhosa* | 1.25 | 1.33 | 1.57 | 1.38 | | |
| *Chara* spp. | 1.23 | 1.40 | 1.56 | 1.40 | | |
| | | | | 1.57 | 1.55 | 0.331 |

The average elevation at which intertidal salt marsh occurred was 1.5 m amsl whereas supratidal salt marsh occurred at 1.77 m amsl. Reeds and sedges occurred predominantly at an elevation of 1.52 m amsl and submerged macrophytes at 1.39 m amsl. The average elevation for the different macrophyte habitats was 1.55 m amsl and using this as a threshold of the water level at which most habitats would be inundated, four abiotic states with the associated dominant macrophytes could be identified. These states occurred sequentially from 2006 to 2010 and were (Figure 5.3):

A. Open and tidal, dominated by intertidal salt marsh, supratidal salt marsh, reeds and sedge and patchy *Ruppia* and macroalgae,
B. Closed and low water level (< 1.55 m amsl), mesohaline, dominated by intertidal salt marsh, supratidal salt marsh, reed and sedge and patchy *Ruppia* and macroalgae.
C. Closed and high water level (> 1.55 m amsl), polyhaline, with *Ruppia* common and with patches of supratidal salt marsh and reeds and sedges
D. Closed and high water level (> 1.55 m amsl), euhaline and macroalgae common

State C was present for 48 % of the time over the 5 year period while State D accounted for 35 % of the total time (Table 5.5). It is possible that a fifth state of euhaline conditions could take place under low water level but this state was not observed during the course of this study. It is also possible that a sixth state, a closed and high water level (> 1.55 m amsl) and mesohaline state could occur as has previously been observed (Whitfield *et al*., 2008). Figure 5.3 shows the sequential occurrence of the four abiotic states identified in this study. Polyhaline is taken to represent 18 to 30 ppt and mesohaline 5 to 18 ppt. Only water level and salinity as described by the canonical axis of the DCCA multivariate analysis were taken as the two main drivers.

Table 5.5. Frequency of states (based on 1.55 m amsl) in which the East Kleinemonde Estuary occurred between March 2006 and January 2010 and the associated abiotic conditions.

| State | Total period (%) | Salinity (ppt) |
|---|---|---|
| A: Open and tidal | 9 | 19 – 27.2 |
| B: Closed and low | 8 | 19.9 – 22.6 |
| C: Closed high, polyhaline | 48 | 19.7 – 22.5 |
| D: Closed and high, euhaline | 35 | 30.8 – 33.9 |

### *5.3.3. Time required to achieve maximum macrophyte habitat cover*

The numbers of months required to achieve 100 % cover for the dominant macrophytes ranged from 0.3 to 11 months (Figure 5.5). Submerged macrophytes established 100 % cover in 0.3 months in State C. Their growth was however patchy as they did not form continuous beds along the edge of the estuary as in the adjacent West Kleinemonde Estuary. The slow growth of submerged macrophytes was also probably due to the large fluctuations in salinity and water level that occurred over the five year period. State A, where the mouth was open and water levels fluctuated was also not conducive for submerged macrophyte or macroalgal development due to continual flooding and exposure from tidal action. These as well as floating macroalgae are washed out of the estuary under open conditions. Both *Juncus kraussii* and *Stenotaphrum secundatum* expanded the fastest under State C (Figure 5.5). This is probably a stress response to inundation. *Sarcocornia decumbens* responded quickest during open mouth conditions (State A; one month). Reeds and sedges expanded the fastest under State A (one month) and macroalgae responded quickest under closed, high water level, when salinity was both polyhaline and euhaline (0.3 to 0.6 months).

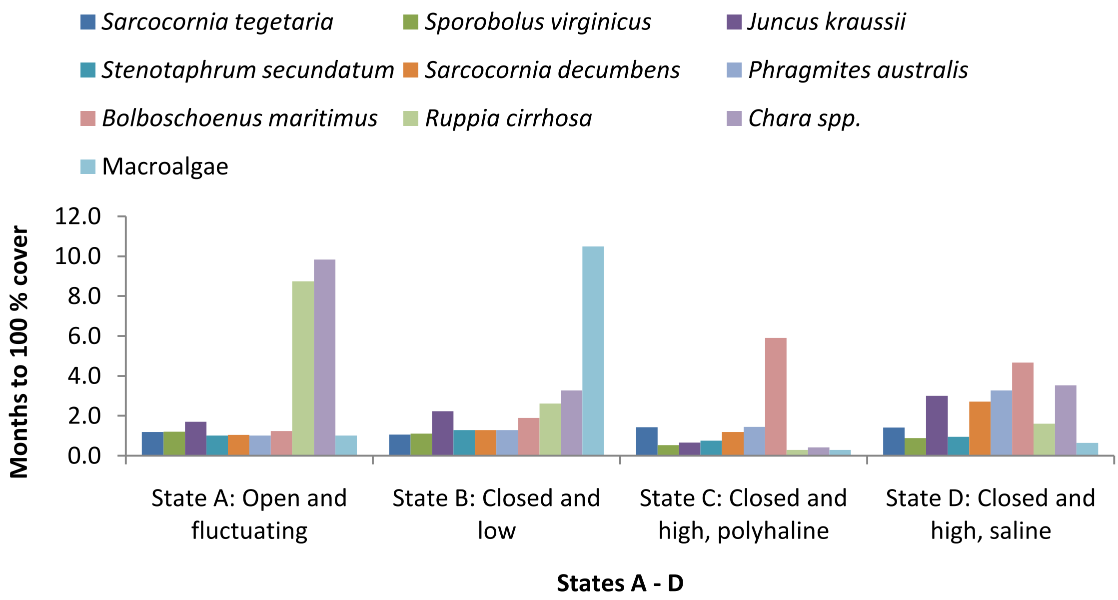



Figure 5.5. The number of months to achieve 100 % cover for the dominant macrophytes of the East Kleinemonde Estuary under each of the four states, using 1.55 m amsl as the water level at which the most significant changes in cover occurred.

There were large spatial and temporal changes in macrophyte cover along the transects sampled as shown in the comparative images (Plates 1, 2, 3a to 3d).

Plate 1. November 2006 when the mouth of the East Kleinemonde Estuary was open and the estuary was tidal (State A).

Plate 2. East Kleinemonde Estuary in September 2009 when the mouth was closed, water level was high and the estuary was euhaline (State D).

a) Transect 1, November 2005 (State B).

b) Transect 1, May 2006 (State C).

c) Transect 1, June 2006 just prior to the mouth breaching (State C).

d) Transect 1, June 2006 immediately after the breach (State A).

Plate 3. Macrophytes at Transect 1 showing high water level and submerged macrophytes changing to intertidal salt marsh in response to the drop in water level after a breach event.

Species such as *Sarcocornia tegetaria*, *S. decumbens*, *Sporobolus virginicus* and *Phragmites australis*, were able to reshoot from what appeared to be dead material (Plate 4 and 5).

Plate 4. *Sarcocornia decumbens* at the end of Transect 3 which was covered for two months but still continued to grow, despite being heavily covered with epiphytes.

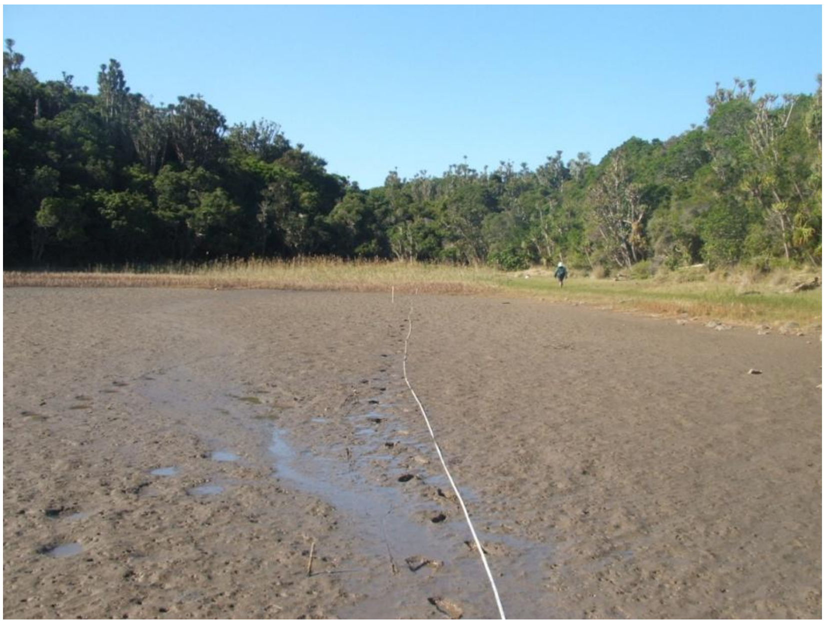


Plate 5a. Transect 2, east bank, in August 2006 shortly after the breach. The dominant macrophyte is *Phragmites australis* at the end of the transect (State A).

Plate 5b. Transect 2, east bank, showing vegetative regrowth of *Phragmites australis* and *Sporobolus virginicus* from existing plant material (November 2006) (State B).

The monthly inflow volumes for the present (Appendix 1), reference state (Appendix 2) and Scenario 1 and 2 calculated in the RDM freshwater requirement study for the East Kleinemonde Estuary (Appendix 3 and 4) were assessed for frequency at which water volumes below or above 0.332 million $m^3$ occurred (equivalent to a water level of 1.55 m water level). This determined the frequency of abiotic states identified in this study (Table 5.6). State D could not be predicted since simulations in the RDM study only took into account water level/volume and not salinity therefore for the purpose of this exercise the frequency of State C and D have been combined. Under the present condition closed high water conditions (State C and D) have increased (27 %) compared to the reference condition (26 %) indicating that the freshwater inflow into the estuary has remained largely unchanged. Using this threshold method, data indicated that the East Kleinemonde Estuary would have predominantly been in State B, closed and low water levels, represented by intertidal salt marsh, supratidal salt marsh, reeds and sedges and isolated patches of submerged macrophytes and macroalgae. Under Scenario 1 there would be an increase in the closed high water states (State C and D) (30 %), whereas Scenario 2 showed a slight decrease in the closed States C and D and an increase in the closed low water State B (56 %).

Table 5.6. Frequency of occurrence (%) of abiotic states identified in this study for past, present, future scenarios, together with water level data and salinity from this study (2006 - 2010).

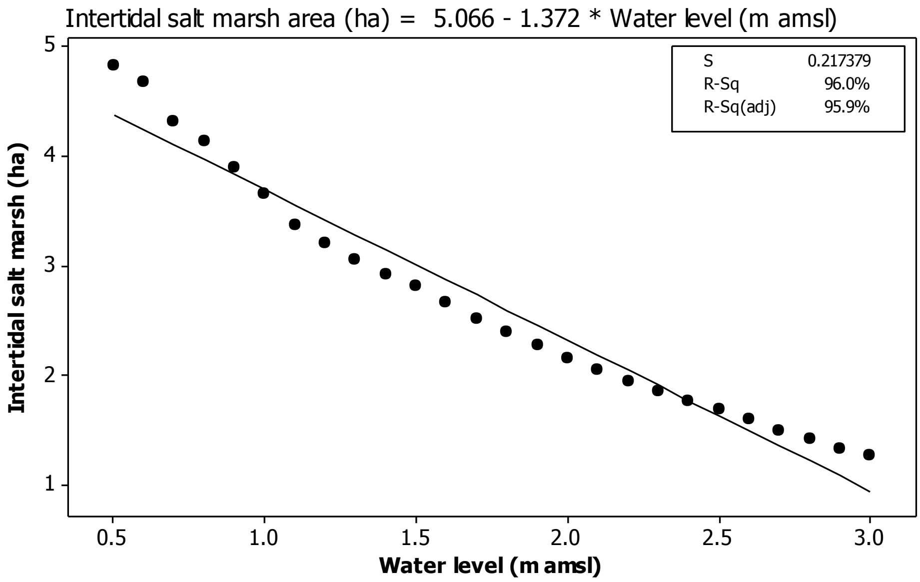
| | Frequency of occurrence (%) | | |
|---|---|---|---|
| | State A<br>Open and tidal | State B<br>Closed, low water level | State C and D<br>Closed, high water level, mesohaline to euhaline |
| Reference state from RDM study | 22 | 52 | 26 |
| Present state from RDM study | 21 | 53 | 27 |
| This study's data (2005 – 2010) | 9 | 8 | 83 (48 + 35) |
| Scenario 1 from RDM study | 16 | 53 | 30 |
| Scenario 2 from RDM study | 19 | 56 | 25 |

Analysis of the present 5 year water level/water volume data showed a much higher occurrence of States C and D (83 %). This represents drought conditions at the time of the 5 year study. The RDM study considered simulated water level over a 72 year period which includes larger fluctuations.

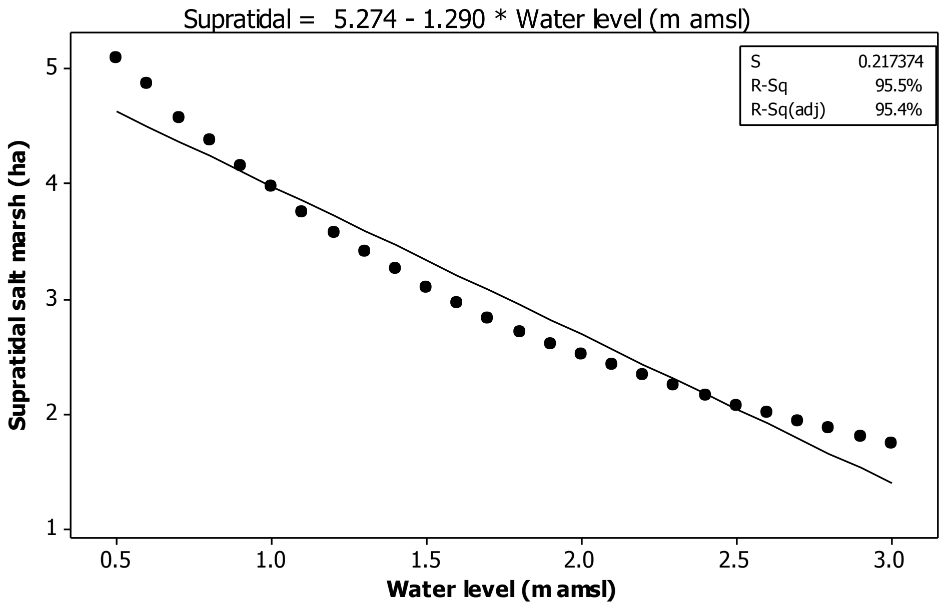
### *5.3.3 Spatial habitat availability model*

Emergent macrophyte habitat availability (ha) decreased linearly as water level rose amsl (Figures 5.6a to 5.6d). The empirical equations of available habitat area for each water level are shown on the associated graph. These equations are specific to the East Kleinemonde Estuary as they were based on this estuary's digital elevation model. Linear model were used as opposed to polynomial models as they produced the highest coefficient of determination.

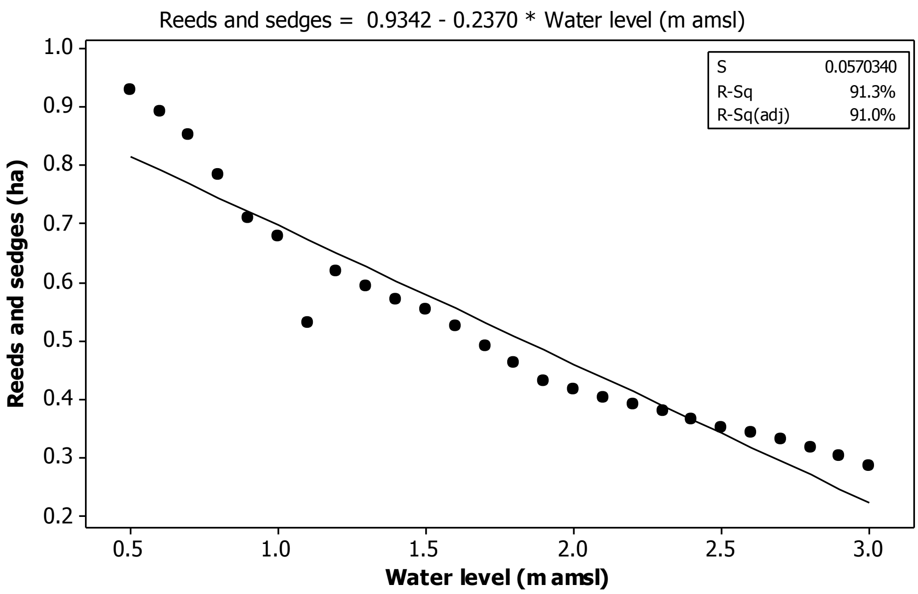
a)


Intertidal salt marsh area (ha) = 5.066 - 1.372 * Water level (m amsl)
S 0.217379
R-Sq 96.0%
R-Sq(adj) 95.9%
Intertidal salt marsh (ha)
Water level (m amsl)
0.5
1.0
1.5
2.0
2.5
3.0
1
2
3
4
5

b)


Supratidal = 5.274 - 1.290 * Water level (m amsl)
S 0.217374
R-Sq 95.5%
R-Sq(adj) 95.4%
Supratidal salt marsh (ha)
Water level (m amsl)
0.5
1.0
1.5
2.0
2.5
3.0
1
2
3
4
5

c)




d)

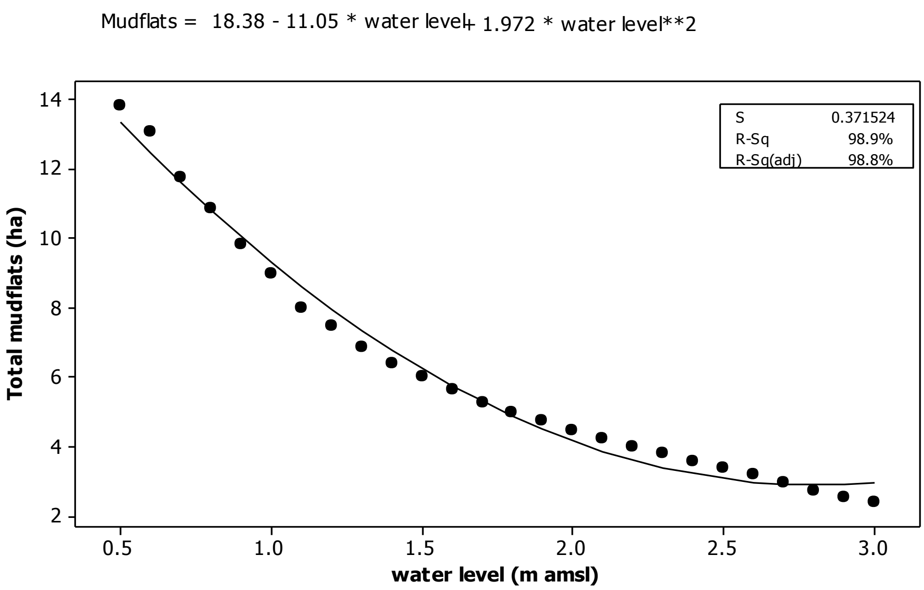



Figure 5.6. Available macrophyte habitat (ha) with water depth (m amsl) in the East Kleinemonde Estuary based on the spatial model calculations.

Submerged macrophyte habitat area increased to a maximum at a water level between 1.4 m to 1.8 m amsl (Figure 5.7). Vertical habitat availability (expressed as water volume) was also at a maximum between these two water levels (Figure 5.8).

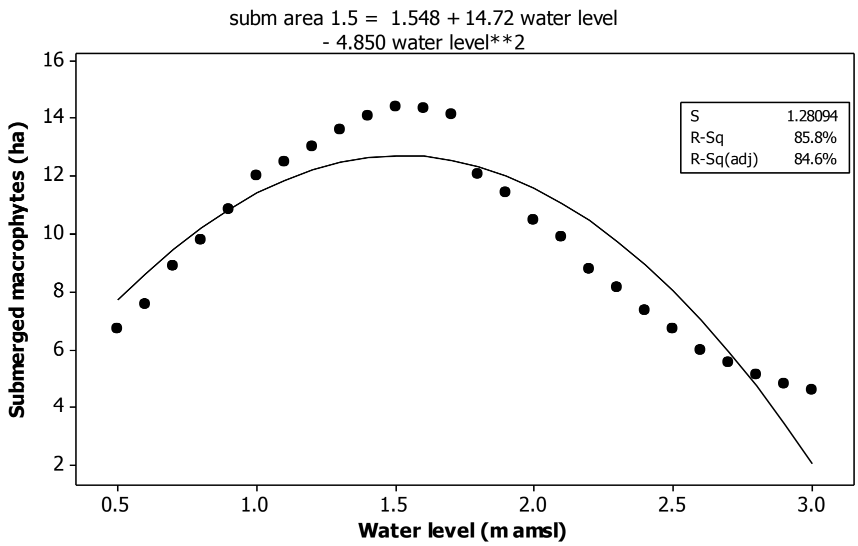



Figure 5.7. Available submerged macrophyte habitat area (ha) with water depth (m amsl) in the East Kleinemonde Estuary based on the spatial model calculations.

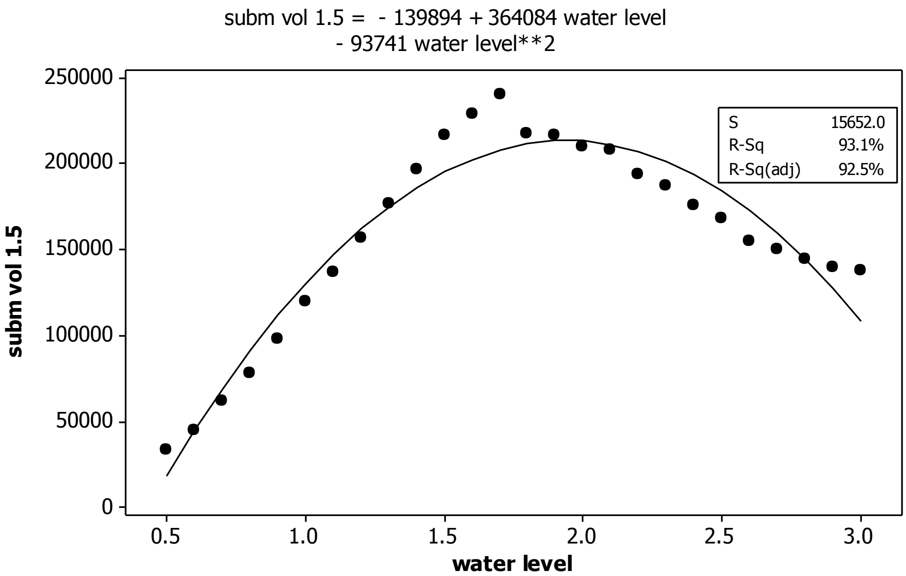



Figure 5.8. Available submerged macrophyte water volume for submerged macrophytes (vertical habitat) with water depth (m amsl) in the East Kleinemonde Estuary based on the spatial model calculations.

Macroalgal habitat (expressed as 25 % of the water area) increased as water level increased (Figure 5.9).

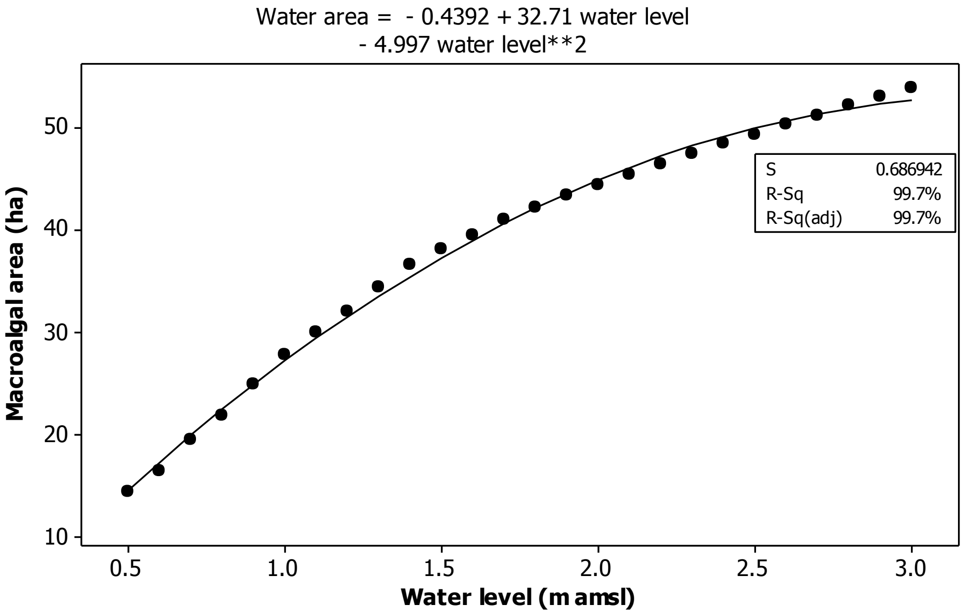



Figure 5.9. Available macroalgal habitat (ha) with water depth (m amsl) in the East Kleinemonde Estuary based on the spatial model calculations.

These spatial habitat equations were applied to the past, present and two future freshwater inflow scenarios to quantify the area covered by the dominant habitats in the East Kleinemonde Estuary. This was done by converting the monthly inflow volumes for the different scenarios to water level using the equation of Theron and Bornman (2008). Average monthly water level values for the 72 year period were used. Water level determined the dominant macrophyte habitats for the different scenarios. Scenario 1 had the highest submerged macrophyte area (12.56 ha versus 12.48 ha), whereas Scenario 2 produced the largest mudflat (7.34 ha) and intertidal salt marsh area (3.28 ha) (Figure 5.10). Submerged macrophytes were dominant during the closed state as this is when they grow rapidly. Mudflat and intertidal salt marsh were dominant during the tidal phase. During this study (2006-2010) there was a 26 % decrease in mudflat area and 16 % decrease in intertidal salt marsh habitat due to a greater frequency of States C and D (closed mouth, high water level).

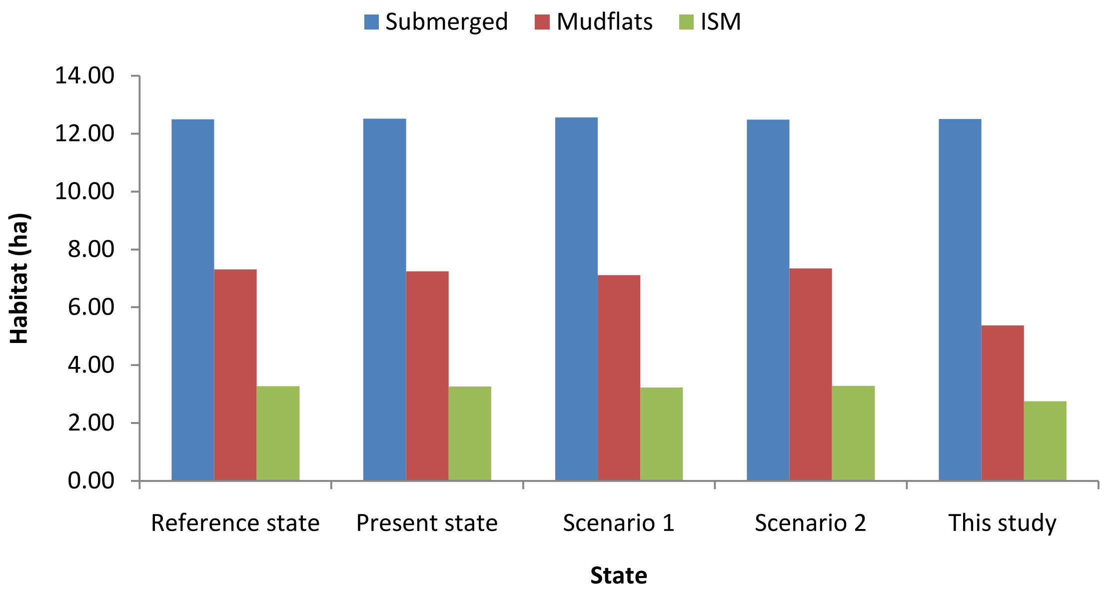



Figure 5.10. Available habitat (submerged macrophytes, mudflats and intertidal salt marsh ISM) for the reference, present, future scenarios and this study (2005-2010).

This method of determining available habitat is very rapid and broad as it works on the average water level, not taking into account monthly water level fluctuations. As has been shown in the present 5 year study, these fluctuations are important in affecting rate of habitat establishment, especially for submerged macrophytes.

## *5.4 Discussion*

The two main drivers of macrophyte cover change in the East Kleinemonde Estuary were water level and salinity. High water level with high salinity negatively influenced supratidal salt marsh, submerged macrophytes and reeds and sedges and positively influenced macroalgal cover. Due to the high variability in abiotic parameters that occurred during the 5 year study period, submerged macrophytes did not develop large stands as they require stable water level for at least four months to establish. The average water level at which habitats in the East Kleinemonde Estuary switched from intertidal salt marsh to submerged macrophytes was 1.55 m amsl. These data together with the long-term abiotic data were used to identify four different abiotic states (A-D) and the dominant macrophyte habitats associated with these states. State A was an open tidal state where intertidal salt marsh, supratidal salt marsh, reeds and sedges and patches of submerged macrophytes were common. State B occurred when the mouth closed and water level was below 1.55 m amsl. Macrophyte cover was similar to State A. States C and D were both closed high water states (> 1.55 m amsl) but whereas State C was polyhaline (18 to 30 ppt), for State D salinity was above 30 ppt and macroalgae replaced submerged macrophytes. Supratidal salt marsh, reeds and sedges died off as a result of inundation with euhaline water. Previous studies have also identified a closed state in the East Kleinemonde Estuary when there was low salinity and the submerged macrophyte *Ruppia cirrhosa* was replaced by *Stukenia pectinata* (Van Niekerk *et al*., 2008; Whitfield *et al*., 2008). Furthermore Whitfield *et al*. (2008) recognised the occurrence of small marine overwash events but showed that they would only result in a maximum salinity change of 3.8 ppt. In this study a major (episodic) marine overwash event following mouth closure in September 2008 increased salinity above 30 ppt from 21.9 ppt and more importantly, maintained these euhaline conditions for 16 months. Although previous studies have shown that the East Kleinemonde Estuary remains closed for 71.8 % of the time (Cowley and Whitfield, 2001: 14 year data set) and 78.4 % of the time (Van Niekerk *et al*., 2008: 72 year simulated data), this study (using a 5 year data set) has shown a total closed period of 83 % of the time, comprised of 48 % in a polyhaline state (State C) and 35 % in a euhaline state (State D). This highlights the variability of TOCEs and the possible error in drawing conclusions when using short term data sets.

Once habitat became available most macrophytes were quick to respond with recovery periods as short as 1 month. This was similar to what has been found for ephemeral systems such as pans and lakes which are characterised by fluctuating water level and highly variable abiotic conditions (Arendt, 1997). Similarly, rapid colonisation rates, high seed density and high total percentage germination of seed banks have been found to occur in frequently flooded sites (Capon and Brock, 2006). High seed numbers in species with persistent seed banks is due to continous recharge of the sediment over time, for example öospores of Charophytes. These can remain viable for up to 11 years (Brock and Britton, 1995). In this study they were found to be viable when stored for 5 years in the sediment of the East Kleinemonde Estuary as germination of *Chara* spp. were observed in January 2011 presumably from sediment reserves dating back to before the commencement of this study, as no large *Chara* beds formed during the study period. Fast growth rates in unfavourable environments means that individuals reach maturity and reproductive stage before abiotic conditions can change.

The identification of the euhaline state in TOCEs (State D) has important ecological consequences because submerged macrophytes such as *Ruppia cirrhosa* are replaced by macroalgae. Macroalgae in TOCEs are typically only seasonal in occurrence and are represented by species such as *Cladophora* and *Enteromorpha* (Fong *et al*., 1996; Kamer *et al*., 2001). They proliferate during the closed mouth phase and are washed out to sea during the open phase (Adams *et al.,* 1999). The large macroalgal cover established during this study probably also contributed to the loss of submerged macrophytes due to shading. Loss of submerged macrophytes and their associated fauna such as invertebrates, fish and bird populations have important ecological impacts (McGlathery, 2001; Mannino and Sara, 2006; Froneman and Henninger, 2009). The slow response and small area of submerged macrophyte habitat development that took place during this study period (2006 – 2010) was probably due the fluctuating water level and salinity. Submerged macrophytes require stable water level for adequate beds to establish. In the Painkalac and Angelsea estuaries in Australia, submerged macrophyte beds were correlated with stability of the water level and the absence of flood events, where a stable water level for a few months resulted in the establishment of submerged macrophyte beds (Pope, 2008). It has even been suggested that disturbance events, in particularly water level and salinity fluctuations, can cause a change in growth patterns (Pope, 2008). Growth forms of *Ruppia cirrhosa* with short life cycles (within 2 months) were found to occur in areas where water level fluctuations took place regularly. By contrast, stable water level led to longer life cycles (Calado and Duarte, 2000; Gesti *et al*., 2005). This was probably the reason why submerged macrophytes seldom lasted for more than three years in the St Lucia Estuary where large water level and salinity fluctuations occurred in response to rainfall cycles (Taylor *et al*., 1987). Although *Ruppia cirrhosa* occurs over a salinity range of 0 to 45 ppt (Adams and Bate, 1994), salinity fluctuations were found to be the limiting factor for *Ruppia maritima* recovery in the Gulf of Mexico (La Peyre and Rowe, 2003). This is probably why *Ruppia* cirrhosa also did not expand to form large beds in the East Kleinemonde Estuary due to continuously fluctuating water level and salinity.

Flood events also play an important role in submerged macrophyte development. In August 2006 there was a 1:30 year flood event which probably removed large quantities of sediment and viable propagules when the mouth breached. Prior to the flood, stable water level periods occurred between 1995 and 2000 and between 2000 and 2003 during which time submerged macrophyte beds developed (Sheppard, 2010). It was only in 2007 and 2008 during this study that beds of submerged macrophytes began to develop again. Flood and freshette events have been suggested as the drivers of abiotic changes in the East Kleinemonde Estuary with baseflow to a lesser extent. These flood events reset the estuary by lowering the base level and maintain longer open mouth periods. Despite these fluctuations in macrophytes cover, long term fish studies over 16 years have shown that fish populations in the East Kleinemonde Estuary have remained relatively stable with no species changes in composition; just changes in abundance (James *et al*., 2008).

The identification of a threshold water level for macrophyte change (1.55 m amsl for the East Kleinemonde Estuary) made it possible to integrate this into the RDM methodology by determining abiotic states and macrophyte responses for the past, present and future freshwater inflow scenarios. It also provides a further link between abiotic conditions and biota in the methodology. Simulated water inflow volume data can be quickly filtered for a threshold level to determine the presence of abiotic states with characteristic macrophyte habitats. Results showed that the East Kleinemonde Estuary has changed little in its macrophyte habitats under past and present simulated conditions, although data from this 5 year study showed a much higher occurrence of the closed States C and D. The high variability in abiotic variables found over the 5 year period highlights the variability characteristic of TOCEs; a factor which must be considered in the management of these estuaries.

The spatial habitat model is a further tool for linking biota to abiotic conditions, in this case water level. Under future inflow Scenario 1 the spatial model determined that there would be the highest potential submerged macrophyte area available, 12.56 ha versus 12.48 ha in Scenario 1 (State C in the threshold method), whereas Scenario 2 produced the largest potential mudflat area, 7.34 ha, and intertidal salt marsh area (3.28 ha) (State A and B in the threshold method). Although the model in this chapter was run on the average monthly water level data for the 72 year hydrological simulation data, it is possible to run it for each month of each year. This would be done by incorporating a looping ability so that monthly water level for the 72-year period could be used to generate maps as was done by Sharma *et al*. (2009). Each output map could then serve as the input for each new water level with the inclusion of a growth/expansion factor. The methods developed in this study can easily be applied to other TOCEs and will assist managers in determining macrophyte response to abiotic states; invaluable information when estuaries are considered not only in their ecological capacity, but also their social and economic benefits.

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# CHAPTER 6: CONCLUSIONS

The two main threats to temporarily open/closed estuaries (TOCEs) in the short and long term are development and climate change. Development not only leads to a direct loss of habitat, but also results in the modification of freshwater inflow into estuaries as a result of increased abstraction (Bornman *et al*., 2002; Peirson *et al*., 2002; Barton *et al*., 2008; Whitfield and Taylor, 2009; Cyrus *et al*., 2010). Reduced freshwater inflow will result in an increase in the duration of mouth closure and an increase in hypersaline periods particularly in dry climatic areas such as the Eastern Cape Province in South Africa. In this study the two main abiotic drivers of macrophyte change in the temporarily open/closed East Kleinemonde Estuary in the Eastern Cape were found to be salinity and water level (Chapters 2, 3 and 5). Results were based on monthly data of macrophyte cover change along three permanent transects monitored over a 5 year period. No similar monthly long term data set exists for either TOCEs or permanently open estuaries (POEs) in South Africa.

The study showed that increased periods of mouth closure will lead to increased submerged macrophyte abundance under mesohaline and polyhaline conditions and macroalgae under euhaline and hypersaline conditions (Chapters 2 and 5). Emergent habitat such as intertidal and supratidal salt marsh will also be lost due to longer periods of inundation. This study has shown that inundation for longer than two to three months resulted in a significant decrease in intertidal and supratidal salt marsh cover in the East Kleinemonde Estuary. However freshwater inflow into estuaries can also increase due to urban discharge and salt marsh species can be replaced with brackish species (Zedler *et al*., 1990; Adam, 2002). There is some evidence of this in the East Kleinemonde Estuary where reeds have established in localised areas above and below the R72 road bridge adjacent to residential development. These reeds were not there in 1995 (Riddin and Adams, 2008 in Van Niekerk *et al*., 2008).

Increased periods of hypersalinity (not only due to abstraction but also due to drought conditions) will result in a loss of submerged macrophytes and salt marsh habitat and an increase in macroalgae. In this study salinity above 30 ppt resulted in the replacement of submerged macrophytes by macroalgae (Chapter 3), as well as a significant decrease in supratidal salt marsh cover (Chapter 3). Increased salinity was also found to delay germination and the establishment of submerged macrophytes (Chapter 3, 4 and 5). A recent study by Becker *et al*. (2010) has shown the importance of littoral zones in the East Kleinemonde Estuary as nursery and foraging areas. Reed patches were regularly used by zooplanktivore fish. Loss of these littoral zones under altered freshwater inflows would have serious ecological implications. An increase in hypersaline periods is not only anticipated under altered freshwater inflows, but also over longer timescales due to climate change. As global warming increases it has been shown that wind speeds will increase as the oceans warm, wave height will also increase as will the frequency of storm surge events (Wang *et al*., 2008). This could have two effects; it can either lead to a build-up of the sand bar across the mouth of a TOCE resulting in longer periods of mouth closure, or it could lead to more marine overwash events and an increase in mouth breaches associated with storm surges. A storm surge in September 2008 in the East Kleinemonde Estuary resulted in a mouth breach for 1 day. The mouth closed when the water level in the estuary was high and was kept high due to a series of large marine overwash events during the next year. This resulted in an increase in mean salinity throughout the length of the estuary by 9.1 ppt, and salinity remained significantly higher for the next 16 months (Chapter 3). Macroalgae replaced submerged macrophytes and emergent habitat was also significantly reduced. The predicted increased activity in sea storms associated with global warming could therefore have significant affects on macrophyte habitats in TOCEs leading to a loss of submerged macrophytes. This is because submerged macrophytes in the East Kleinemonde Estuary were found to require stable water levels for two to four months for establishment (Chapter 2 and 5). Submerged macrophytes in the East Kleinemonde Estuary have historically supported populations of the critically endangered estuarine pipefish *Syngnathus watermeyeri* (Cowley and Whitfield, 2001). Their absence has been linked to the loss of submerged macrophytes as a result of reduced and fluctuating freshwater inflow (Whitfield, 1995). By contrast tropical fish species have increased in the East Kleinemonde Estuary due to an increase in the mean annual sea-surface temperature in the adjacent coast (James *et al*., 2008).

Even under fluctuating water level and salinity macrophyte habitats were found to show a high degree of resilience and persistence, i.e. they were able to return after a disturbance (Gunderson *et al*., 2010). This is because there is a high degree of abiotic variability in

TOCEs that the macrophytes are adapted to. Macrophytes were found to respond quickly to water level changes as they had fast growth rates. For example the intertidal species, *Sarcocornia tegetaria*, took only 0.3 months (10 days) to achieve 100 % cover (Chapter 2 and 5). This rapid change in cover was largely due to their rapid germination response from large sediment seed reserves. Germination studies in Chapter 4 showed that *Sarcocornia tegetaria* began to establish from seed after 3 days of exposure to conditions favourable for growth, in this case a reduction in water level leading to exposed habitat. Over a 91 day trial period, this species had a maximum germination of 82 % and germination took place continuously during this period. After 24 days, 50 % of the *Sarcocornia* seeds had germinated at 0 ppt. Germination was lower at the higher salinity treatment (50 % germination after 77 days at 35 ppt). Even under waterlogged conditions 39 % of the seeds germinated. By contrast submerged macrophytes required a stable water level for at least 18 days before germination began (Chapter 4). Germination for both *Ruppia cirrhosa* and *Chara vulgaris* was slow over a 91 day trial period (11 % and 15 %). Submerged macrophytes are therefore more likely to be influenced by water level fluctuations under increased mouth breach events.

As well as rapid growth responses, estuarine macrophytes were found to have large sediment seed reserves (Chapter 4). This study showed that the seed banks of two temporarily open/closed estuaries, the East and adjacent West Kleinemonde estuaries were characterised by low species diversity but high seed density. Seed numbers were high for three of the dominant estuarine macrophytes which made up 98 % of the seed bank. Charophyte öospore represented 71.8 % of the seeds within the top 5 cm of the sediment in both the East and West Kleinemonde estuaries. An average density of 31 306 ± 2 293 öospores $m^{-2}$ was recorded for both estuaries with no significant difference in the mean density between the two estuaries. This high density means that conditions must have been favourable prior to the sampling period that resulted in the establishment of Charophytes. Periods of desiccation associated with fluctuating water level probably increased öospore production. The second highest seed density was for the intertidal species *Sarcocornia tegetaria* (7 929 seeds $m^{-2}$), followed by the submerged species *Ruppia cirrhosa* (2 852 seeds $m^{-2}$). *Stukenia pectinata* was found to have only 77 seeds $m^{-2}$. High seed numbers for all these species, except *Stukenia pectinata*, were probably due to persistent seed bank reserves brought about by adverse abiotic conditions, such as water level fluctuations which are typical of temporarily open/closed estuaries. Water level decreases leading to exposed conditions have been shown to increase öospore production (Kautsky, 1990; Casanova and Brock, 1996; Asaeda *et al*., 2007). Charophytes in temporary water bodies require a period of desiccation as well as intermittent moistening as a prerequisite for germination (Harwell and Havens, 2003; Casanova and Brock, 1996). Desiccation also stimulates germination in *Ruppia* seed (Kantrud, 1991). Sediment seed reserves have until this study, not been quantified and the high seed density implies that habitats are capable of persisting over wide abiotic fluctuations. Viable seed was found at 20 cm depth indicating that seeds would probably remain after scouring by small flood events. The slow recovery of submerged macrophyte beds after the August 2006 1:30 year flood was probably because the flood removed viable seed during the scouring of the estuary when the mouth breached. However, enough seed probably remained to ensure regrowth, although not to the extent as had been documented during closed periods prior to 2006. High seed density also ensures that habitats are maintained together with replenishment from reproduction. Although Charophytes formed 100 % cover in places in this study, overall their distribution was patchy and scarce. Despite this high sediment reserves indicate their prior existence. Storage for 5 years also did not affect their viability. Other studies have shown that öospores can remain viable for up to 11 years (Brock and Britton, 1995).

This study has made an original contribution to the field of knowledge of macrophyte responses in a small TOCE as it showed that macrophyte habitats in the East Kleinemonde Estuary have a high natural variability in cover over time, they respond quickly after a disturbance event such as a mouth breach and there are large sediment seed reserves that can remain viable from 2 to more than 5 years. This ensures habitat persistence even under unfavourable conditions, such as prolonged periods of mouth closure with flooding and loss of salt marsh species. Given this natural variability and potential threats due to short term (freshwater inflow abstraction) and long term (global warming) changes, it is necessary to predict responses both spatially and temporally in order to manage and maintain ecological functioning in TOCEs. This study identified dominant macrophyte habitat for different abiotic states through the use of water level and salinity thresholds (Chapter 5). Threshold water level was based on the bathymetry and the elevation extent of the macrophyte habitats present in the East Kleinemonde Estuary and represented the water level at which macrophyte habitat changed from predominantly intertidal to submerged macrophytes. It was found to be on average 1.55 m amsl. A switch from intertidal to submerged macrophytes has ecological implications in terms of associated fauna, for example a change from waders to piscivore birds (Terörde and Turpie, 2008). The salinity at which the most significant change in macrophyte cover occurred was 30 ppt in the East Kleinemonde Estuary. Based on these thresholds this led to the identification of four abiotic states characterised by dominant macrophyte habitats that occurred during the study period. These were:

State A. Open and tidal

State B. Closed and low water level (< 1.55 m amsl), polyhaline (18 to 30 ppt)

State C. Closed and high water level (> 1.55 m amsl), polyhaline (18 to 30 ppt)

State D. Closed and high water level (> 1.55 m amsl), euhaline (> 30 ppt)

States A and B were characterised mainly by salt marsh and reeds and sedges. State C was characterised mainly by submerged macrophytes and State D mainly by macroalgae. States B to D may vary from days to months or even years depending on the rainfall patterns and other climatic conditions such as storm events (Perissinotto *et al*., 2010). Other states may well exist but were not identified during the 5 year study period (2006-2010). They are a closed and low water, euhaline state and a closed and low water, mesohaline state. The latter state has existed previously when large beds of *Stukenia pectinata* were observed in the East Kleinemonde Estuary (Whitfield *et al*., 2008) as this species grows best in water with salinity less than 10 ppt (Van Wijk *et al*., 1988). During this 5 year study the East Kleinemonde Estuary was predominantly in State C (48 % of the study time) and State D (35 % of the study time) (Chapter 5). State A occurred for 9 % of the time and State B for 8 % of the time between 2005 and 2010. Using the data from the RDM study it was estimated that the closed state would occur for 78.4 % of the time for the present estuary state. The longer closed mouth state during this 5 year study period was due to drought, as well as the fact that the RDM method covers a 72-year hydrological data record whereas this study only represented 5 years. The reference state and present state of the East Kleinemonde Estuary were very similar and indicated that the estuary is in a largely unmodified state (Van Niekerk *et al*., 2008). However persistent drought conditions, freshwater abstraction and increasing human impacts such as development and associated nutrient input will influence the future health of the estuary.

Although the threshold water level identified in this study is specific to the East Kleinemonde Estuary and is based on the elevation range of the habitats and species found in this estuary, threshold water level can also be determined for other TOCEs after preliminary surveys of the habitat range and elevation amsl. With the use of a threshold water level it is possible to take the assessment of the past, present and future freshwater inflow scenarios in the RDM freshwater inflow methodology a step further by quantitatively linking abiotic states to macrophyte habitat. In Chapter 5 two future freshwater inflow scenarios were tested using this method by converting the threshold water level of 1.55 m amsl to an equivalent water volume using the water level/water volume equation calculated in spatial analysis from a digital elevation model. The predicted monthly inflow scenarios for a simulated 72 year period was filtered for this water volume (0.332 million $m^3$) and were assessed for the frequency at which water volumes below or above that value occurred. Results showed that under both scenarios of reduced freshwater inflow mouth open events would decrease (State A) and periods of mouth closure would increase (States B, C and D). State D could not be predicted since hydrological simulations (Appendix 1 to 4) in the RDM study only take into account water level/volume and not monthly salinity changes. The method proposed here serves as a rapid and accurate method for determining habitat responses to abiotic states identified in the current South African inflow methodology. All that is required is elevation profiles of the lateral extent of habitats in estuaries, both present and under past conditions.

A second method was also proposed for additional use in the determination of freshwater inflow requirements of estuaries to quantify available habitat at selected water levels. A spatial model was produced in ArcGis Model Builder by combining a bathymetric and habitat map of the East Kleinemonde Estuary. Under selected water levels the corresponding spatial macrophyte habitat can be quantified thereby providing a link with other trophic levels. Although the spatial model presented in this study was based on average water level conditions in the East Kleinemonde Estuary, it could be applied to each monthly water level over the simulated 72-year period of the freshwater inflow scenarios. This would be done by incorporating a looping ability so that monthly water level for the 72-year period could be used to generate output maps of habitat availability as was done by Sharma *et al*. (2009). Each output map could then serve as the input for each new water level with the inclusion of a macrophyte growth/expansion factor. Maps generated by the spatial model produced in this study represent the final stable state for each water level. Water level may change before the maximum potential area of macrophyte habitat is achieved.

This study has improved the understanding of the dynamics of macrophyte habitat responses in TOCEs and has identified drivers of change, rates of change and quantified seed reserves. This information will be invaluable to the management of TOCEs. Questions such as how long can estuaries remain closed before habitat switches occur or whether habitats can be maintained over time despite frequent disturbance events such as water level fluctuations can now be answered. It is now possible to predict and quantify habitat responses to water level and salinity fluctuations for future inflow scenarios. However there have been limitations to this study as well as the formulation of new questions regarding macrophyte dynamics in TOCEs which are discussed in the next section.

Estuaries worldwide are represented by low species diversity and South Africa is no exception with similar species occurring in most estuaries. The dominant macrophytes are common in most Cape estuaries and therefore the results from this study are assumed to be representative of TOCEs throughout South Africa. However the frequency of disturbance events, i.e. how often water level or salinity fluctuates within TOCEs, may result in different macrophyte responses in terms of growth rate and rate of expansion. It could be hypothesised that sediment seed density would be higher in more variable TOCEs. Kautsky (1990) has shown that in permanently flooded habitats, Charophyte öospore seed banks are smaller than in temporary marshes where Charophytes occur together with angiosperms. Seed reserves may also be lower in POEs as opposed to TOCEs as environmental conditions are more "stable" and growth is primarily vegetative. Tidal influence may also disperse seeds along the length of the estuary or result in seed being washed out to sea. Comparative studies of sediment seed reserves between different types of TOCEs and between TOCEs and POEs are therefore required to determine if seed density measured in this study applies to all estuaries. Few studies also consider seed bank accumulation in the restoration and creation of salt marshes (Lindig-Cisneros and Zedler, 2002; Wolters *et al*., 2005; Diggory and Parker, 2010) and this information, together with effects of storage condition on seed viability will greatly assist in salt marsh restoration efforts. Whereas this study assessed the affects of water level and salinity on germination potential, other cues may be necessary to induce and increase germination. For example, Carr and Ross (1963) found that the germination of *Chara gymnopitys* öospores was stimulated by anaerobic conditions. Storage conditions and after-ripening periods may also be required to ensure increased seed viability, germination and restoration capacity of estuaries. Storage of seeds under high salt concentrations can reduce seed germination and initial seedling growth (Ungar, 1978; Shumway and Bertness, 1992; Kuhn and Zedler, 1997; Callaway and Zedler, 1998). If the mouth of an estuary remains closed for longer than 5 years the question is whether seed reserves other than Charophytes are still viable? It may be necessary to artificially breach the mouth of a closed estuary if seed viability has been found to decrease over time. This study showed that seeds remain viable from 2 to more than 5 years. Data presented in this study serves as a baseline for mouth management in other estuaries in South Africa. In 2010 the permanently open Uilkraals Estuary in the Western Cape, South Africa, closed for the first time. The mouth was artificially opened in November 2010 to ensure the survival of the salt marsh as rare and endemic species occur here, as well as to prevent flooding of adjacent development and possible sewerage leaks. It is hypothesised that the recovery of salt marsh habitat at this estuary will take place within two to three months.

The recovery time of associated fauna once habitats develop is also not known. There may well be a slower response in the dependant fauna and this highlights a further lack of understanding of the use of macrophytes by other trophic levels and their ecological role in the functioning in TOCEs. There are some published studies on the ecological importance of macrophyte habitats to invertebrates and fishes in South African estuaries for example from Swartvlei, an estuarine lake on the southern Cape coast (Davies 1982, Whitfield 1984, 1986), the Kromme system, a permanently open estuary on the Eastern Cape coast (Hanekom and Baird, 1984; Baird, 1999), the Kasouga Estuary (Henninger *et al*., 2009) and the East Kleinemonde Estuary (Cowley, 1998). In the Kasouga Estuary *Ruppia* beds supported large isopod populations which used epiphytic diatoms in these beds as a food source, as well as for refuge from predation.

Another research question is how long must the mouth of an estuary remain open, or water level remain low so as to ensure that salt marsh species are able to flower, set seed and replenish seed reserves in the sediment? These were identified as research questions in the study and are currently being addressed as an ongoing MSc study by D. Vromans, as part of the SeaChange Programme (Society, Ecosystems and Change programme). Outputs from the MSc study are the quantification of reproductive output (i.e. number of seeds produced $m^{-2}$ or gram of seeds $m^{-2}$) of target species in a TOCE versus a POE. It is hypothesised that macrophyte life-cycles in TOCEs are event driven compared to seasonal life cycles in POEs and that reproductive outputs are higher in TOCEs than in POEs. The quantification of the time from seed germination to the formation of viable seed is being assessed as this will assist with mouth management decisions in TOCEs. It is hypothesised that intertidal salt marsh requires at least two months to set viable seed whereas submerged macrophytes require stable water levels for at least three to four months for viable seeds to develop in TOCEs. Preliminary results indicate that *Bolboschoenus maritimus* requires an after-ripening period once seeds have formed and, once viable, it takes about 37 days before germination commences (Vromans, *pers comm*.). These data will assist with mouth management plans for estuaries in terms of how long habitats need to be exposed or inundated to allow flowering, seeding and after-ripening. For example the mouth management plan for the Great Brak Estuary ensures that the mouth is breached in spring to allow for salt marsh to complete their life cycle and replenish sediment seed reserves and for fish and invertebrate recruitment to take place (Slinger, 2000). Inundation of habitats at incorrect periods of the year may result in the loss of macrophyte habitats and loss of seed bank replenishment.

Biomass is the more frequently used parameter to monitor macrophyte change compared to cover and is used in studies which determine the association of higher trophic levels with macrophyte abundance (Sfriso and Ghetti, 1998; Obrador *et al*., 2007). However because this study used permanent transects it was not possible to determine monthly biomass changes. Such quantification over a 5 year period would be extremely destructive. An attempt was made to link salt marsh cover to biomass through empirical equations but this was not possible since water level in the East Kleinemonde Estuary remained high for long periods preventing the collection of biomass samples.

A greater understanding of the feedback mechanisms in macrophyte habitats is required, for example the effects of self shading within a submerged macrophyte stand or the effect of epiphytic algal colonisation on submerged macrophytes. This may well play a role in light limitation regardless of water column turbidity. Van Nes *et al*. (2003) found that despite adequate conditions for Charophytes to develop, they were outcompeted by *Stukenia pectinata* under low light conditions. This was found to be due to a competition for bicarbonate and not only a light limitation. Although submerged macrophytes beds did not persist for long periods of time in the East Kleinemonde Estuary, it is unknown whether stable water levels will in fact lead to extensive beds. It may well be that in the East Kleinemonde and other TOCEs where water level fluctuates regularly (in terms of months) life cycles are short and maximum biomass is reached within a few months in order to utilise optimum growing conditions.

Macroalgae have been widely used as indicators of eutrophication in estuaries (Bricker *et al*., 2003; Cohen and Fong, 2006; Scanes *et al*., 2007; Zaldivar *et al*., 2009). Nutrients were not considered in this study because the East Kleinemonde Estuary is oligotrophic with only localised input from septic tanks (Whitfield *et al*., 2008). However for the two future freshwater inflow scenarios nutrient levels were predicted to increase due to the increased frequency of the closed mouth state and lack of flushing (if closed for more than two months) (Van Niekerk *et al*., 2008). Human (2009) has shown that localised increases in *Phragmites australis* have occurred in the East Kleinemonde Estuary adjacent to developments with septic tanks where there is possible sewerage seepage. If nutrient input increases to an extent that macroalgal growth occurs, it could lead to the loss of submerged macrophytes through shading, as was found by McGlathery (2001), Menendez (2005) and Mannino and Sara (2006). *Ruppia* spp. were lost due to excessive macroalgal growth in response to eutrophication. Nutrient input is a major abiotic driver influencing macrophyte response in other South African TOCEs such as the Great Brak Estuary (DWAF, 2008). For these reasons the long term situation in the East Kleinemonde Estuary should be monitored.

The spatial model method proposed in this study is a simplification of the Stella® method proposed by Turpie *et al*. (2008). Similarly to Stella®, it provides information on the potential macrophyte area for habitat development. However Stella® was based on typical response curves of macrophyte cover to salinity, water level and nutrients. This study has found that macrophytes in the East Kleinemonde Estuary have large tolerance ranges to both salinity and water level and it was not possible to construct a typical species response curve, thus leading to the identification of abiotic states and macrophyte response in TOCEs. Despite this the spatial model using Geographical Information System (GIS) proposed here has more potential than the Stella® model because it can produce spatial maps of macrophyte habitat for future abiotic states. ArcGIS is also more readily used by scientists and estuarine managers. Additional layers of abiotic data, for example turbidity, sediment type, temperature and nutrients, showing the longitudinal distribution of these drivers along the length of an estuary can be added to the model. This study has only considered water level in the spatial model as the other abiotic drivers were homogenous throughout the estuary (Whitfield *et al*., 2008). Studies have however shown that sediment type can also influence the distribution of macrophytes (Haslam, 1971; Barko and Smart, 1986; Lehman, 1998; Harwell and Havens, 2003). In the East Kleinemonde Estuary the sediment has become slightly muddier over time due to catchment degradation (Van Niekerk *et al*., 2008) but this is not considered a cause of macrophyte change at present. Ailstock *et al*. (2010) found that the germination of *Ruppia maritima* increased on smaller sediment grain size whereas *Stukenia pectinata* had a wider tolerance to sediment grain size.

By including macrophyte habitat maps and maps of abiotic conditions in an estuary, a habitat suitability model could easily be produced for all estuaries in South Africa based on an existing knowledge of species responses to abiotic conditions. In a similar method Lehmann (1998) used GIS maps to determine the response of *Potamogeton* and *Chara* in Lake Geneva using General Additive Models (GAM). GIS together with GAM was a powerful tool to predict species distributions under changing abiotic variables. Using data obtained from the present study a similar modeling approach can be used for all estuaries in South Africa by simply expanding the current spatial model to include other abiotic maps, although the threshold method and the determination of abiotic states and macrophyte response is more applicable to TOCEs with no spatial gradients. However, in its present form the spatial model still provides a useful link between macrophytes and associated biota. It is possible for example to extract areas inundated by 30 cm and less of water which would then give an indication of habitat available to wading piscivorous birds as these birds feed in shallow water of this depth (Turpie *et al*., 2008). This is a particularly important predictive ability when linking habitat to freshwater inflow in the future management of South African estuaries under declining water resources. Together with knowledge of rates of habitat development it will serve an invaluable tool for future freshwater inflow requirement studies.

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# APPENDICES

Appendix 1. Simulated monthly volumes (million m³) in the East Kleinemonde Estuary for the Present State (Van Niekerk *et al*., 2008).

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breaches | High Flow Breaches |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1920 | 0.15 | 0.15 | 0.21 | 0.23 | 0.23 | 0.35 | 0.15 | 0.38 | 0.42 | 0.44 | 0.15 | 0.17 | 2 | 1 |
| 1921 | 0.18 | 0.38 | 0.15 | 0.17 | 0.17 | 0.17 | 0.27 | 0.34 | 0.15 | 0.15 | 0.15 | 0.36 | 4 | 3 |
| 1922 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.19 | 0.22 | 0.24 | 0.26 | 0.34 | 0.38 | 0.39 | 5 | 4 |
| 1923 | 0.41 | 0.15 | 0.19 | 0.21 | 0.21 | 0.22 | 0.22 | 0.25 | 0.26 | 0.26 | 0.26 | 0.27 | 1 | 0 |
| 1924 | 0.27 | 0.28 | 0.38 | 0.41 | 0.41 | 0.15 | 0.35 | 0.38 | 0.40 | 0.41 | 0.41 | 0.15 | 2 | 1 |
| 1925 | 0.21 | 0.21 | 0.23 | 0.24 | 0.29 | 0.33 | 0.34 | 0.34 | 0.35 | 0.35 | 0.35 | 0.36 | 0 | 0 |
| 1926 | 0.15 | 0.26 | 0.26 | 0.26 | 0.27 | 0.30 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 1 | 1 |
| 1927 | 0.31 | 0.31 | 0.31 | 0.31 | 0.32 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.23 | 0.15 | 4 | 2 |
| 1928 | 0.15 | 0.37 | 0.15 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.15 | 0.15 | 0.29 | 0.15 | 5 | 4 |
| 1929 | 0.15 | 0.30 | 0.31 | 0.31 | 0.32 | 0.15 | 0.22 | 0.22 | 0.25 | 0.26 | 0.30 | 0.37 | 2 | 1 |
| 1930 | 0.15 | 0.32 | 0.32 | 0.32 | 0.32 | 0.40 | 0.15 | 0.18 | 0.18 | 0.44 | 0.15 | 0.19 | 3 | 1 |
| 1931 | 0.15 | 0.15 | 0.15 | 0.44 | 0.15 | 0.16 | 0.16 | 0.16 | 0.16 | 0.18 | 0.18 | 0.15 | 5 | 4 |
| 1932 | 0.15 | 0.15 | 0.25 | 0.26 | 0.27 | 0.28 | 0.15 | 0.21 | 0.21 | 0.21 | 0.15 | 0.24 | 4 | 2 |
| 1933 | 0.24 | 0.28 | 0.30 | 0.31 | 0.32 | 0.15 | 0.31 | 0.31 | 0.31 | 0.15 | 0.15 | 0.20 | 3 | 3 |
| 1934 | 0.22 | 0.37 | 0.42 | 0.42 | 0.42 | 0.43 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.41 | 5 | 5 |
| 1935 | 0.15 | 0.31 | 0.40 | 0.15 | 0.21 | 0.27 | 0.30 | 0.35 | 0.38 | 0.40 | 0.41 | 0.41 | 2 | 0 |
| 1936 | 0.15 | 0.15 | 0.37 | 0.37 | 0.37 | 0.15 | 0.33 | 0.33 | 0.33 | 0.33 | 0.33 | 0.33 | 3 | 3 |
| 1937 | 0.34 | 0.35 | 0.15 | 0.15 | 0.16 | 0.26 | 0.38 | 0.41 | 0.41 | 0.41 | 0.41 | 0.41 | 2 | 2 |
| 1938 | 0.41 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 0.18 | 0.18 | 0.18 | 0.26 | 5 | 5 |
| 1939 | 0.43 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.38 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 5 | 4 |
| 1940 | 0.40 | 0.15 | 0.22 | 0.22 | 0.24 | 0.24 | 0.15 | 0.15 | 0.19 | 0.21 | 0.21 | 0.21 | 3 | 2 |
| 1941 | 0.21 | 0.26 | 0.15 | 0.33 | 0.36 | 0.37 | 0.38 | 0.15 | 0.18 | 0.18 | 0.19 | 0.19 | 2 | 1 |
| 1942 | 0.15 | 0.24 | 0.26 | 0.44 | 0.15 | 0.15 | 0.36 | 0.43 | 0.43 | 0.43 | 0.15 | 0.17 | 3 | 0 |
| 1943 | 0.17 | 0.15 | 0.32 | 0.33 | 0.35 | 0.15 | 0.25 | 0.33 | 0.36 | 0.36 | 0.36 | 0.15 | 3 | 2 |
| 1944 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 0.40 | 0.44 | 0.15 | 0.15 | 0.15 | 1 | 0 |
| 1945 | 0.39 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 2 | 0 |
| 1946 | 0.16 | 0.17 | 0.18 | 0.18 | 0.18 | 0.15 | 0.34 | 0.36 | 0.44 | 0.15 | 0.23 | 0.24 | 2 | 1 |
| 1947 | 0.24 | 0.27 | 0.28 | 0.28 | 0.28 | 0.28 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 2 | 2 |
| 1948 | 0.15 | 0.30 | 0.30 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.33 | 0.33 | 1 | 1 |
| 1949 | 0.34 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 0.18 | 0.19 | 4 | 4 |

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breaches | High Flow Breaches |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1950 | 0.15 | 0.15 | 0.15 | 0.15 | 0.35 | 0.36 | 0.36 | 0.36 | 0.36 | 0.37 | 0.37 | 0.15 | 5 | 5 |
| 1951 | 0.15 | 0.15 | 0.15 | 0.16 | 0.28 | 0.32 | 0.33 | 0.15 | 0.21 | 0.21 | 0.21 | 0.15 | 3 | 2 |
| 1952 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 0.17 | 0.44 | 0.15 | 2 | 2 |
| 1953 | 0.15 | 0.15 | 0.43 | 0.44 | 0.45 | 0.15 | 0.27 | 0.15 | 0.34 | 0.40 | 0.15 | 0.35 | 5 | 4 |
| 1954 | 0.15 | 0.39 | 0.15 | 0.22 | 0.32 | 0.36 | 0.37 | 0.38 | 0.39 | 0.39 | 0.39 | 0.15 | 3 | 0 |
| 1955 | 0.18 | 0.26 | 0.28 | 0.28 | 0.41 | 0.15 | 0.16 | 0.19 | 0.20 | 0.20 | 0.20 | 0.42 | 1 | 0 |
| 1956 | 0.15 | 0.21 | 0.35 | 0.39 | 0.15 | 0.24 | 0.25 | 0.25 | 0.26 | 0.27 | 0.27 | 0.15 | 3 | 1 |
| 1957 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 0.31 | 0.44 | 0.15 | 0.15 | 0.18 | 0.20 | 0.20 | 2 | 2 |
| 1958 | 0.20 | 0.20 | 0.40 | 0.15 | 0.16 | 0.23 | 0.33 | 0.38 | 0.38 | 0.38 | 0.38 | 0.39 | 1 | 0 |
| 1959 | 0.40 | 0.40 | 0.40 | 0.41 | 0.42 | 0.42 | 0.42 | 0.15 | 0.22 | 0.22 | 0.22 | 0.28 | 1 | 0 |
| 1960 | 0.15 | 0.22 | 0.22 | 0.23 | 0.28 | 0.30 | 0.30 | 0.40 | 0.43 | 0.43 | 0.44 | 0.44 | 1 | 0 |
| 1961 | 0.44 | 0.15 | 0.17 | 0.17 | 0.18 | 0.15 | 0.29 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 2 | 1 |
| 1962 | 0.15 | 0.35 | 0.35 | 0.15 | 0.20 | 0.15 | 0.15 | 0.15 | 0.17 | 0.26 | 0.32 | 0.35 | 5 | 4 |
| 1963 | 0.35 | 0.35 | 0.36 | 0.36 | 0.15 | 0.42 | 0.42 | 0.42 | 0.15 | 0.21 | 0.21 | 0.15 | 3 | 2 |
| 1964 | 0.41 | 0.15 | 0.16 | 0.16 | 0.17 | 0.17 | 0.17 | 0.32 | 0.15 | 0.22 | 0.25 | 0.27 | 2 | 0 |
| 1965 | 0.38 | 0.15 | 0.37 | 0.37 | 0.38 | 0.38 | 0.39 | 0.15 | 0.18 | 0.18 | 0.20 | 0.25 | 2 | 1 |
| 1966 | 0.26 | 0.28 | 0.28 | 0.28 | 0.33 | 0.35 | 0.40 | 0.15 | 0.15 | 0.40 | 0.15 | 0.23 | 3 | 2 |
| 1967 | 0.28 | 0.31 | 0.33 | 0.34 | 0.35 | 0.37 | 0.15 | 0.17 | 0.15 | 0.15 | 0.21 | 0.29 | 3 | 2 |
| 1968 | 0.34 | 0.36 | 0.36 | 0.36 | 0.40 | 0.15 | 0.23 | 0.23 | 0.23 | 0.24 | 0.24 | 0.24 | 1 | 0 |
| 1969 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.25 | 0.15 | 0.15 | 2 | 2 |
| 1970 | 0.15 | 0.27 | 0.15 | 0.15 | 0.24 | 0.27 | 0.15 | 0.33 | 0.36 | 0.39 | 0.15 | 0.20 | 5 | 4 |
| 1971 | 0.26 | 0.28 | 0.34 | 0.36 | 0.15 | 0.23 | 0.23 | 0.23 | 0.23 | 0.25 | 0.26 | 0.27 | 1 | 0 |
| 1972 | 0.27 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.30 | 0.30 | 0.30 | 0.30 | 0.38 | 0.41 | 0 | 0 |
| 1973 | 0.42 | 0.15 | 0.20 | 0.30 | 0.44 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 8 | 7 |
| 1974 | 0.30 | 0.40 | 0.15 | 0.20 | 0.28 | 0.32 | 0.34 | 0.35 | 0.37 | 0.38 | 0.40 | 0.15 | 2 | 1 |
| 1975 | 0.15 | 0.17 | 0.19 | 0.19 | 0.20 | 0.34 | 0.39 | 0.39 | 0.39 | 0.15 | 0.15 | 0.21 | 3 | 3 |
| 1976 | 0.38 | 0.15 | 0.19 | 0.19 | 0.15 | 0.39 | 0.42 | 0.15 | 0.30 | 0.36 | 0.39 | 0.43 | 3 | 1 |
| 1977 | 0.45 | 0.15 | 0.15 | 0.25 | 0.25 | 0.33 | 0.15 | 0.15 | 0.15 | 0.32 | 0.43 | 0.15 | 6 | 4 |
| 1978 | 0.15 | 0.29 | 0.41 | 0.15 | 0.18 | 0.20 | 0.21 | 0.23 | 0.25 | 0.15 | 0.15 | 0.15 | 5 | 4 |
| 1979 | 0.29 | 0.38 | 0.44 | 0.15 | 0.19 | 0.22 | 0.24 | 0.26 | 0.30 | 0.31 | 0.31 | 0.33 | 1 | 0 |
| 1980 | 0.34 | 0.42 | 0.45 | 0.15 | 0.28 | 0.15 | 0.15 | 0.42 | 0.15 | 0.24 | 0.15 | 0.32 | 5 | 4 |
| 1981 | 0.44 | 0.15 | 0.19 | 0.21 | 0.21 | 0.22 | 0.40 | 0.15 | 0.18 | 0.20 | 0.21 | 0.21 | 2 | 0 |
| 1982 | 0.23 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.15 | 0.27 | 0.27 | 1 | 1 |
| 1983 | 0.30 | 0.32 | 0.32 | 0.32 | 0.32 | 0.32 | 0.32 | 0.32 | 0.15 | 0.39 | 0.43 | 0.45 | 1 | 1 |
| 1984 | 0.15 | 0.15 | 0.15 | 0.17 | 0.18 | 0.18 | 0.18 | 0.19 | 0.19 | 0.19 | 0.19 | 0.19 | 1 | 0 |
| 1985 | 0.42 | 0.15 | 0.15 | 0.21 | 0.23 | 0.30 | 0.33 | 0.33 | 0.33 | 0.33 | 0.15 | 0.21 | 3 | 2 |
| 1986 | 0.26 | 0.30 | 0.31 | 0.31 | 0.32 | 0.33 | 0.34 | 0.34 | 0.43 | 0.15 | 0.18 | 0.20 | 1 | 0 |
| 1987 | 0.21 | 0.21 | 0.21 | 0.21 | 0.34 | 0.41 | 0.42 | 0.44 | 0.44 | 0.45 | 0.45 | 0.15 | 1 | 0 |

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breaches | High Flow Breaches |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1988 | 0.17 | 0.19 | 0.19 | 0.19 | 0.19 | 0.19 | 0.32 | 0.36 | 0.36 | 0.37 | 0.37 | 0.37 | 0 | 0 |
| 1989 | 0.15 | 0.15 | 0.15 | 0.15 | 0.20 | 0.31 | 0.34 | 0.34 | 0.45 | 0.15 | 0.15 | 0.16 | 4 | 3 |
| 1990 | 0.20 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.33 | 0.36 | 0 | 0 |
| 1991 | 0.15 | 0.20 | 0.20 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.15 | 0.15 | 3 | 2 |
| 1992 | 0.15 | 0.38 | 0.39 | 0.39 | 0.40 | 0.40 | 0.41 | 0.41 | 0.15 | 0.17 | 0.20 | 0.39 | 2 | 1 |
| 1993 | 0.45 | 0.45 | 0.15 | 0.27 | 0.32 | 0.33 | 0.33 | 0.33 | 0.33 | 0.33 | 0.15 | 0.25 | 2 | 0 |
| 1994 | 0.25 | 0.25 | 0.15 | 0.15 | 0.17 | 0.19 | 0.31 | 0.36 | 0.36 | 0.36 | 0.36 | 0.36 | 2 | 2 |
| 1995 | 0.36 | 0.41 | 0.43 | 0.43 | 0.43 | 0.44 | 0.44 | 0.44 | 0.44 | 0.45 | 0.15 | 0.15 | 1 | 0 |
| 1996 | 0.15 | 0.15 | 0.15 | 0.16 | 0.16 | 0.16 | 0.42 | 0.15 | 0.15 | 0.26 | 0.26 | 0.26 | 4 | 3 |
| 1997 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.15 | 0.30 | 0.30 | 0.30 | 0.32 | 0.36 | 0.38 | 1 | 1 |
| 1998 | 0.39 | 0.40 | 0.40 | 0.41 | 0.41 | 0.42 | 0.42 | 0.42 | 0.42 | 0.15 | 0.19 | 0.22 | 1 | 0 |
| 1999 | 0.15 | 0.26 | 0.26 | 0.15 | 0.28 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 4 | 4 |
| 2000 | 0.25 | 0.15 | 0.22 | 0.27 | 0.28 | 0.31 | 0.35 | 0.36 | 0.36 | 0.15 | 0.23 | 0.38 | 2 | 0 |
| 2001 | 0.43 | 0.15 | 0.20 | 0.20 | 0.20 | 0.20 | 0.21 | 0.21 | 0.21 | 0.15 | 0.15 | 0.15 | 4 | 3 |
| 2002 | 0.15 | 0.21 | 0.23 | 0.24 | 0.24 | 0.24 | 0.27 | 0.15 | 0.15 | 0.20 | 0.24 | 0.25 | 3 | 3 |
| | | | | | | | | | | | | Ave | 2.63 | 1.72 |

High flow breaching

Low flow breaching

Appendix 2. Simulated monthly volumes (million m$^3$) in the East Kleinemonde Estuary for the reference state (Van Niekerk *et al*., 2008).

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breaches | High Flow Breaches |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1920 | 0.15 | 0.15 | 0.21 | 0.23 | 0.23 | 0.35 | 0.15 | 0.38 | 0.42 | 0.44 | 0.15 | 0.17 | 2 | 1 |
| 1921 | 0.18 | 0.38 | 0.15 | 0.17 | 0.17 | 0.17 | 0.27 | 0.34 | 0.15 | 0.15 | 0.15 | 0.36 | 4 | 3 |
| 1922 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.19 | 0.22 | 0.24 | 0.26 | 0.34 | 0.38 | 0.39 | 5 | 4 |
| 1923 | 0.41 | 0.15 | 0.19 | 0.21 | 0.21 | 0.22 | 0.22 | 0.25 | 0.26 | 0.26 | 0.26 | 0.27 | 1 | 0 |
| 1924 | 0.27 | 0.28 | 0.38 | 0.41 | 0.41 | 0.15 | 0.35 | 0.38 | 0.40 | 0.41 | 0.41 | 0.15 | 2 | 1 |
| 1925 | 0.21 | 0.21 | 0.23 | 0.24 | 0.29 | 0.33 | 0.34 | 0.34 | 0.35 | 0.35 | 0.35 | 0.36 | 0 | 0 |
| 1926 | 0.15 | 0.26 | 0.26 | 0.26 | 0.27 | 0.30 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 1 | 1 |
| 1927 | 0.31 | 0.31 | 0.31 | 0.31 | 0.32 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.23 | 0.15 | 4 | 2 |
| 1928 | 0.15 | 0.37 | 0.15 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.15 | 0.15 | 0.29 | 0.15 | 5 | 4 |
| 1929 | 0.15 | 0.30 | 0.31 | 0.31 | 0.32 | 0.15 | 0.22 | 0.22 | 0.25 | 0.26 | 0.30 | 0.37 | 2 | 1 |
| 1930 | 0.15 | 0.32 | 0.32 | 0.32 | 0.32 | 0.40 | 0.15 | 0.18 | 0.18 | 0.44 | 0.15 | 0.19 | 3 | 1 |
| 1931 | 0.15 | 0.15 | 0.15 | 0.44 | 0.15 | 0.16 | 0.16 | 0.16 | 0.16 | 0.18 | 0.18 | 0.15 | 5 | 4 |
| 1932 | 0.15 | 0.15 | 0.25 | 0.26 | 0.27 | 0.28 | 0.15 | 0.21 | 0.21 | 0.21 | 0.15 | 0.24 | 4 | 2 |
| 1933 | 0.24 | 0.28 | 0.30 | 0.31 | 0.32 | 0.15 | 0.31 | 0.31 | 0.31 | 0.15 | 0.15 | 0.20 | 3 | 3 |
| 1934 | 0.22 | 0.37 | 0.42 | 0.42 | 0.42 | 0.43 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.41 | 5 | 5 |
| 1935 | 0.15 | 0.31 | 0.40 | 0.15 | 0.21 | 0.27 | 0.30 | 0.35 | 0.38 | 0.40 | 0.41 | 0.41 | 2 | 0 |
| 1936 | 0.15 | 0.15 | 0.37 | 0.37 | 0.37 | 0.15 | 0.33 | 0.33 | 0.33 | 0.33 | 0.33 | 0.33 | 3 | 3 |
| 1937 | 0.34 | 0.35 | 0.15 | 0.15 | 0.16 | 0.26 | 0.38 | 0.41 | 0.41 | 0.41 | 0.41 | 0.41 | 2 | 2 |
| 1938 | 0.41 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 0.18 | 0.18 | 0.18 | 0.26 | 5 | 5 |
| 1939 | 0.43 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.38 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 5 | 4 |
| 1940 | 0.40 | 0.15 | 0.22 | 0.22 | 0.24 | 0.24 | 0.15 | 0.15 | 0.19 | 0.21 | 0.21 | 0.21 | 3 | 2 |
| 1941 | 0.21 | 0.26 | 0.15 | 0.33 | 0.36 | 0.37 | 0.38 | 0.15 | 0.18 | 0.18 | 0.19 | 0.19 | 2 | 1 |
| 1942 | 0.15 | 0.24 | 0.26 | 0.44 | 0.15 | 0.15 | 0.36 | 0.43 | 0.43 | 0.43 | 0.15 | 0.17 | 3 | 0 |
| 1943 | 0.17 | 0.15 | 0.32 | 0.33 | 0.35 | 0.15 | 0.25 | 0.33 | 0.36 | 0.36 | 0.36 | 0.15 | 3 | 2 |
| 1944 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 0.40 | 0.44 | 0.15 | 0.15 | 0.15 | 1 | 0 |
| 1945 | 0.39 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 2 | 0 |
| 1946 | 0.16 | 0.17 | 0.18 | 0.18 | 0.18 | 0.15 | 0.34 | 0.36 | 0.44 | 0.15 | 0.23 | 0.24 | 2 | 1 |
| 1947 | 0.24 | 0.27 | 0.28 | 0.28 | 0.28 | 0.28 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 2 | 2 |
| 1948 | 0.15 | 0.30 | 0.30 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.33 | 0.33 | 1 | 1 |
| 1949 | 0.34 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 0.18 | 0.19 | 4 | 4 |
| 1950 | 0.15 | 0.15 | 0.15 | 0.15 | 0.35 | 0.36 | 0.36 | 0.36 | 0.36 | 0.37 | 0.37 | 0.15 | 5 | 5 |
| 1951 | 0.15 | 0.15 | 0.15 | 0.16 | 0.28 | 0.32 | 0.33 | 0.15 | 0.21 | 0.21 | 0.21 | 0.15 | 3 | 2 |
| 1952 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 0.17 | 0.44 | 0.15 | 2 | 2 |
| 1953 | 0.15 | 0.15 | 0.43 | 0.44 | 0.45 | 0.15 | 0.27 | 0.15 | 0.34 | 0.40 | 0.15 | 0.35 | 5 | 4 |

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breaches | High Flow Breaches |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1954 | 0.15 | 0.39 | 0.15 | 0.22 | 0.32 | 0.36 | 0.37 | 0.38 | 0.39 | 0.39 | 0.39 | 0.15 | 3 | 0 |
| 1955 | 0.18 | 0.26 | 0.28 | 0.28 | 0.41 | 0.15 | 0.16 | 0.19 | 0.20 | 0.20 | 0.20 | 0.42 | 1 | 0 |
| 1956 | 0.15 | 0.21 | 0.35 | 0.39 | 0.15 | 0.24 | 0.25 | 0.25 | 0.26 | 0.27 | 0.27 | 0.15 | 3 | 1 |
| 1957 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 0.31 | 0.44 | 0.15 | 0.15 | 0.18 | 0.20 | 0.20 | 2 | 2 |
| 1958 | 0.20 | 0.20 | 0.40 | 0.15 | 0.16 | 0.23 | 0.33 | 0.38 | 0.38 | 0.38 | 0.38 | 0.39 | 1 | 0 |
| 1959 | 0.40 | 0.40 | 0.40 | 0.41 | 0.42 | 0.42 | 0.42 | 0.15 | 0.22 | 0.22 | 0.22 | 0.28 | 1 | 0 |
| 1960 | 0.15 | 0.22 | 0.22 | 0.23 | 0.28 | 0.30 | 0.30 | 0.40 | 0.43 | 0.43 | 0.44 | 0.44 | 1 | 0 |
| 1961 | 0.44 | 0.15 | 0.17 | 0.17 | 0.18 | 0.15 | 0.29 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 2 | 1 |
| 1962 | 0.15 | 0.35 | 0.35 | 0.15 | 0.20 | 0.15 | 0.15 | 0.15 | 0.17 | 0.26 | 0.32 | 0.35 | 5 | 4 |
| 1963 | 0.35 | 0.35 | 0.36 | 0.36 | 0.15 | 0.42 | 0.42 | 0.42 | 0.15 | 0.21 | 0.21 | 0.15 | 3 | 2 |
| 1964 | 0.41 | 0.15 | 0.16 | 0.16 | 0.17 | 0.17 | 0.17 | 0.32 | 0.15 | 0.22 | 0.25 | 0.27 | 2 | 0 |
| 1965 | 0.38 | 0.15 | 0.37 | 0.37 | 0.38 | 0.38 | 0.39 | 0.15 | 0.18 | 0.18 | 0.20 | 0.25 | 2 | 1 |
| 1966 | 0.26 | 0.28 | 0.28 | 0.28 | 0.33 | 0.35 | 0.40 | 0.15 | 0.15 | 0.40 | 0.15 | 0.23 | 3 | 2 |
| 1967 | 0.28 | 0.31 | 0.33 | 0.34 | 0.35 | 0.37 | 0.15 | 0.17 | 0.15 | 0.15 | 0.21 | 0.29 | 3 | 2 |
| 1968 | 0.34 | 0.36 | 0.36 | 0.36 | 0.40 | 0.15 | 0.23 | 0.23 | 0.23 | 0.24 | 0.24 | 0.24 | 1 | 0 |
| 1969 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.25 | 0.15 | 0.15 | 2 | 2 |
| 1970 | 0.15 | 0.27 | 0.15 | 0.15 | 0.24 | 0.27 | 0.15 | 0.33 | 0.36 | 0.39 | 0.15 | 0.20 | 5 | 4 |
| 1971 | 0.26 | 0.28 | 0.34 | 0.36 | 0.15 | 0.23 | 0.23 | 0.23 | 0.23 | 0.25 | 0.26 | 0.27 | 1 | 0 |
| 1972 | 0.27 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.30 | 0.30 | 0.30 | 0.30 | 0.38 | 0.41 | 0 | 0 |
| 1973 | 0.42 | 0.15 | 0.20 | 0.30 | 0.44 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 8 | 7 |
| 1974 | 0.30 | 0.40 | 0.15 | 0.20 | 0.28 | 0.32 | 0.34 | 0.35 | 0.37 | 0.38 | 0.40 | 0.15 | 2 | 1 |
| 1975 | 0.15 | 0.17 | 0.19 | 0.19 | 0.20 | 0.34 | 0.39 | 0.39 | 0.39 | 0.15 | 0.15 | 0.21 | 3 | 3 |
| 1976 | 0.38 | 0.15 | 0.19 | 0.19 | 0.15 | 0.39 | 0.42 | 0.15 | 0.30 | 0.36 | 0.39 | 0.43 | 3 | 1 |
| 1977 | 0.45 | 0.15 | 0.15 | 0.25 | 0.25 | 0.33 | 0.15 | 0.15 | 0.15 | 0.32 | 0.43 | 0.15 | 6 | 4 |
| 1978 | 0.15 | 0.29 | 0.41 | 0.15 | 0.18 | 0.20 | 0.21 | 0.23 | 0.25 | 0.15 | 0.15 | 0.15 | 5 | 4 |
| 1979 | 0.29 | 0.38 | 0.44 | 0.15 | 0.19 | 0.22 | 0.24 | 0.26 | 0.30 | 0.31 | 0.31 | 0.33 | 1 | 0 |
| 1980 | 0.34 | 0.42 | 0.45 | 0.15 | 0.28 | 0.15 | 0.15 | 0.42 | 0.15 | 0.24 | 0.15 | 0.32 | 5 | 4 |
| 1981 | 0.44 | 0.15 | 0.19 | 0.21 | 0.21 | 0.22 | 0.40 | 0.15 | 0.18 | 0.20 | 0.21 | 0.21 | 2 | 0 |
| 1982 | 0.23 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.15 | 0.27 | 0.27 | 1 | 1 |
| 1983 | 0.30 | 0.32 | 0.32 | 0.32 | 0.32 | 0.32 | 0.32 | 0.32 | 0.15 | 0.39 | 0.43 | 0.45 | 1 | 1 |
| 1984 | 0.15 | 0.15 | 0.15 | 0.17 | 0.18 | 0.18 | 0.18 | 0.19 | 0.19 | 0.19 | 0.19 | 0.19 | 1 | 0 |
| 1985 | 0.42 | 0.15 | 0.15 | 0.21 | 0.23 | 0.30 | 0.33 | 0.33 | 0.33 | 0.33 | 0.15 | 0.21 | 3 | 2 |
| 1986 | 0.26 | 0.30 | 0.31 | 0.31 | 0.32 | 0.33 | 0.34 | 0.34 | 0.43 | 0.15 | 0.18 | 0.20 | 1 | 0 |
| 1987 | 0.21 | 0.21 | 0.21 | 0.21 | 0.34 | 0.41 | 0.42 | 0.44 | 0.44 | 0.45 | 0.45 | 0.15 | 1 | 0 |
| 1988 | 0.17 | 0.19 | 0.19 | 0.19 | 0.19 | 0.19 | 0.32 | 0.36 | 0.36 | 0.37 | 0.37 | 0.37 | 0 | 0 |
| 1989 | 0.15 | 0.15 | 0.15 | 0.15 | 0.20 | 0.31 | 0.34 | 0.34 | 0.45 | 0.15 | 0.15 | 0.16 | 4 | 3 |
| 1990 | 0.20 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.33 | 0.36 | 0 | 0 |
| 1991 | 0.15 | 0.20 | 0.20 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.15 | 0.15 | 3 | 2 |

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breaches | High Flow Breaches |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1992 | 0.15 | 0.38 | 0.39 | 0.39 | 0.40 | 0.40 | 0.41 | 0.41 | 0.15 | 0.17 | 0.20 | 0.39 | 2 | 1 |
| 1993 | 0.45 | 0.45 | 0.15 | 0.27 | 0.32 | 0.33 | 0.33 | 0.33 | 0.33 | 0.33 | 0.15 | 0.25 | 2 | 0 |
| 1994 | 0.25 | 0.25 | 0.15 | 0.15 | 0.17 | 0.19 | 0.31 | 0.36 | 0.36 | 0.36 | 0.36 | 0.36 | 2 | 2 |
| 1995 | 0.36 | 0.41 | 0.43 | 0.43 | 0.43 | 0.44 | 0.44 | 0.44 | 0.44 | 0.45 | 0.15 | 0.15 | 1 | 0 |
| 1996 | 0.15 | 0.15 | 0.15 | 0.16 | 0.16 | 0.16 | 0.42 | 0.15 | 0.15 | 0.26 | 0.26 | 0.26 | 4 | 3 |
| 1997 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.15 | 0.30 | 0.30 | 0.30 | 0.32 | 0.36 | 0.38 | 1 | 1 |
| 1998 | 0.39 | 0.40 | 0.40 | 0.41 | 0.41 | 0.42 | 0.42 | 0.42 | 0.42 | 0.15 | 0.19 | 0.22 | 1 | 0 |
| 1999 | 0.15 | 0.26 | 0.26 | 0.15 | 0.28 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 4 | 4 |
| 2000 | 0.25 | 0.15 | 0.22 | 0.27 | 0.28 | 0.31 | 0.35 | 0.36 | 0.36 | 0.15 | 0.23 | 0.38 | 2 | 0 |
| 2001 | 0.43 | 0.15 | 0.20 | 0.20 | 0.20 | 0.20 | 0.21 | 0.21 | 0.21 | 0.15 | 0.15 | 0.15 | 4 | 3 |
| 2002 | 0.15 | 0.21 | 0.23 | 0.24 | 0.24 | 0.24 | 0.27 | 0.15 | 0.15 | 0.20 | 0.24 | 0.25 | 3 | 3 |
| | | | | | | | | | | | | Ave | 2.63 | 1.72 |

High flow breaching

Low flow breaching

Appendix 3. Simulated monthly volumes (million m$^3$) in the East Kleinemonde Estuary for Future Scenario 1 (Van Niekerk *et al*., 2008).

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total breach | High Flow Breach |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1920 | 0.15 | 0.15 | 0.17 | 0.17 | 0.18 | 0.21 | 0.42 | 0.15 | 0.16 | 0.17 | 0.18 | 0.18 | 2 | 0 |
| 1921 | 0.18 | 0.26 | 0.30 | 0.30 | 0.30 | 0.31 | 0.34 | 0.36 | 0.15 | 0.15 | 0.15 | 0.34 | 3 | 3 |
| 1922 | 0.44 | 0.15 | 0.15 | 0.15 | 0.43 | 0.44 | 0.15 | 0.16 | 0.16 | 0.19 | 0.20 | 0.21 | 4 | 3 |
| 1923 | 0.21 | 0.23 | 0.24 | 0.25 | 0.25 | 0.25 | 0.25 | 0.26 | 0.26 | 0.26 | 0.26 | 0.27 | 0 | 0 |
| 1924 | 0.27 | 0.27 | 0.30 | 0.31 | 0.31 | 0.42 | 0.15 | 0.16 | 0.17 | 0.17 | 0.17 | 0.23 | 1 | 0 |
| 1925 | 0.24 | 0.24 | 0.25 | 0.25 | 0.27 | 0.28 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 0 | 0 |
| 1926 | 0.41 | 0.44 | 0.44 | 0.44 | 0.44 | 0.15 | 0.15 | 0.16 | 0.16 | 0.16 | 0.16 | 0.16 | 1 | 0 |
| 1927 | 0.16 | 0.16 | 0.16 | 0.16 | 0.16 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.18 | 0.38 | 3 | 2 |
| 1928 | 0.15 | 0.35 | 0.40 | 0.42 | 0.42 | 0.42 | 0.42 | 0.42 | 0.15 | 0.15 | 0.28 | 0.15 | 4 | 3 |
| 1929 | 0.15 | 0.27 | 0.28 | 0.28 | 0.28 | 0.35 | 0.37 | 0.37 | 0.38 | 0.38 | 0.39 | 0.42 | 1 | 1 |
| 1930 | 0.15 | 0.27 | 0.27 | 0.27 | 0.27 | 0.29 | 0.33 | 0.34 | 0.34 | 0.43 | 0.15 | 0.16 | 2 | 1 |
| 1931 | 0.15 | 0.15 | 0.15 | 0.41 | 0.43 | 0.43 | 0.43 | 0.43 | 0.43 | 0.43 | 0.44 | 0.15 | 4 | 4 |
| 1932 | 0.15 | 0.15 | 0.22 | 0.22 | 0.23 | 0.23 | 0.29 | 0.31 | 0.31 | 0.31 | 0.42 | 0.15 | 3 | 2 |
| 1933 | 0.15 | 0.16 | 0.17 | 0.17 | 0.17 | 0.36 | 0.43 | 0.44 | 0.44 | 0.15 | 0.15 | 0.17 | 3 | 2 |
| 1934 | 0.18 | 0.27 | 0.30 | 0.30 | 0.30 | 0.30 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.40 | 5 | 5 |
| 1935 | 0.15 | 0.27 | 0.31 | 0.33 | 0.35 | 0.37 | 0.38 | 0.40 | 0.41 | 0.41 | 0.42 | 0.42 | 1 | 0 |
| 1936 | 0.15 | 0.15 | 0.35 | 0.35 | 0.35 | 0.15 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 3 | 3 |
| 1937 | 0.27 | 0.28 | 0.15 | 0.15 | 0.15 | 0.19 | 0.23 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 2 | 2 |
| 1938 | 0.24 | 0.15 | 0.43 | 0.43 | 0.15 | 0.15 | 0.45 | 0.15 | 0.15 | 0.15 | 0.15 | 0.18 | 4 | 3 |
| 1939 | 0.28 | 0.31 | 0.15 | 0.15 | 0.15 | 0.15 | 0.37 | 0.38 | 0.38 | 0.38 | 0.38 | 0.38 | 4 | 4 |
| 1940 | 0.38 | 0.15 | 0.17 | 0.17 | 0.18 | 0.18 | 0.15 | 0.15 | 0.17 | 0.17 | 0.17 | 0.17 | 3 | 2 |
| 1941 | 0.17 | 0.19 | 0.34 | 0.43 | 0.45 | 0.45 | 0.45 | 0.15 | 0.16 | 0.16 | 0.16 | 0.16 | 1 | 0 |
| 1942 | 0.26 | 0.29 | 0.30 | 0.36 | 0.38 | 0.38 | 0.15 | 0.18 | 0.18 | 0.18 | 0.20 | 0.21 | 1 | 0 |
| 1943 | 0.21 | 0.15 | 0.27 | 0.28 | 0.28 | 0.41 | 0.15 | 0.19 | 0.21 | 0.21 | 0.21 | 0.15 | 3 | 2 |
| 1944 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.39 | 0.39 | 0.39 | 0.39 | 0 | 0 |
| 1945 | 0.15 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 1 | 0 |
| 1946 | 0.18 | 0.18 | 0.19 | 0.19 | 0.19 | 0.37 | 0.43 | 0.44 | 0.15 | 0.22 | 0.25 | 0.25 | 1 | 0 |
| 1947 | 0.25 | 0.26 | 0.27 | 0.27 | 0.27 | 0.27 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 2 | 2 |
| 1948 | 0.15 | 0.26 | 0.26 | 0.26 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 1 | 1 |
| 1949 | 0.27 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 0.16 | 4 | 4 |
| 1950 | 0.15 | 0.15 | 0.15 | 0.15 | 0.34 | 0.34 | 0.34 | 0.34 | 0.34 | 0.34 | 0.35 | 0.15 | 5 | 5 |
| 1951 | 0.15 | 0.15 | 0.15 | 0.15 | 0.19 | 0.21 | 0.21 | 0.27 | 0.29 | 0.29 | 0.29 | 0.15 | 2 | 2 |
| 1952 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 0.16 | 0.25 | 0.15 | 2 | 2 |

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total breach | High Flow Breach |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1953 | 0.15 | 0.15 | 0.40 | 0.41 | 0.41 | 0.15 | 0.21 | 0.15 | 0.33 | 0.36 | 0.15 | 0.34 | 5 | 3 |
| 1954 | 0.42 | 0.15 | 0.21 | 0.23 | 0.27 | 0.28 | 0.28 | 0.29 | 0.29 | 0.29 | 0.29 | 0.31 | 1 | 0 |
| 1955 | 0.33 | 0.35 | 0.36 | 0.36 | 0.40 | 0.42 | 0.42 | 0.43 | 0.43 | 0.43 | 0.43 | 0.15 | 1 | 0 |
| 1956 | 0.18 | 0.20 | 0.24 | 0.25 | 0.32 | 0.34 | 0.35 | 0.35 | 0.35 | 0.35 | 0.35 | 0.15 | 1 | 0 |
| 1957 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.27 | 0.15 | 0.15 | 0.16 | 0.17 | 0.17 | 2 | 2 |
| 1958 | 0.17 | 0.17 | 0.23 | 0.26 | 0.26 | 0.29 | 0.32 | 0.34 | 0.34 | 0.34 | 0.34 | 0.34 | 0 | 0 |
| 1959 | 0.34 | 0.34 | 0.34 | 0.35 | 0.35 | 0.35 | 0.35 | 0.41 | 0.44 | 0.44 | 0.44 | 0.15 | 1 | 0 |
| 1960 | 0.21 | 0.23 | 0.23 | 0.24 | 0.25 | 0.26 | 0.26 | 0.29 | 0.30 | 0.30 | 0.30 | 0.30 | 0 | 0 |
| 1961 | 0.30 | 0.32 | 0.33 | 0.33 | 0.33 | 0.15 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 1 | 0 |
| 1962 | 0.38 | 0.44 | 0.44 | 0.15 | 0.17 | 0.15 | 0.15 | 0.15 | 0.16 | 0.21 | 0.27 | 0.27 | 4 | 3 |
| 1963 | 0.27 | 0.27 | 0.28 | 0.28 | 0.15 | 0.34 | 0.34 | 0.34 | 0.41 | 0.43 | 0.43 | 0.15 | 2 | 2 |
| 1964 | 0.39 | 0.42 | 0.42 | 0.42 | 0.42 | 0.42 | 0.42 | 0.15 | 0.20 | 0.22 | 0.23 | 0.24 | 1 | 0 |
| 1965 | 0.27 | 0.15 | 0.34 | 0.34 | 0.34 | 0.34 | 0.34 | 0.37 | 0.38 | 0.38 | 0.38 | 0.40 | 1 | 1 |
| 1966 | 0.40 | 0.41 | 0.41 | 0.41 | 0.43 | 0.43 | 0.45 | 0.15 | 0.15 | 0.38 | 0.15 | 0.20 | 3 | 2 |
| 1967 | 0.21 | 0.22 | 0.23 | 0.24 | 0.24 | 0.25 | 0.27 | 0.28 | 0.15 | 0.15 | 0.17 | 0.24 | 2 | 2 |
| 1968 | 0.26 | 0.26 | 0.26 | 0.26 | 0.27 | 0.35 | 0.38 | 0.38 | 0.38 | 0.38 | 0.38 | 0.38 | 0 | 0 |
| 1969 | 0.38 | 0.38 | 0.38 | 0.38 | 0.38 | 0.38 | 0.38 | 0.38 | 0.39 | 0.39 | 0.15 | 0.15 | 2 | 2 |
| 1970 | 0.15 | 0.26 | 0.15 | 0.15 | 0.18 | 0.19 | 0.15 | 0.29 | 0.31 | 0.32 | 0.37 | 0.39 | 4 | 4 |
| 1971 | 0.42 | 0.43 | 0.44 | 0.15 | 0.25 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 0.30 | 1 | 0 |
| 1972 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 0.31 | 0.31 | 0.31 | 0.31 | 0.34 | 0.35 | 0 | 0 |
| 1973 | 0.35 | 0.40 | 0.41 | 0.44 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 8 | 7 |
| 1974 | 0.27 | 0.32 | 0.34 | 0.36 | 0.38 | 0.40 | 0.40 | 0.41 | 0.41 | 0.42 | 0.43 | 0.15 | 1 | 1 |
| 1975 | 0.15 | 0.16 | 0.16 | 0.16 | 0.17 | 0.21 | 0.23 | 0.23 | 0.23 | 0.15 | 0.15 | 0.18 | 3 | 3 |
| 1976 | 0.33 | 0.43 | 0.45 | 0.45 | 0.15 | 0.33 | 0.34 | 0.15 | 0.28 | 0.32 | 0.33 | 0.34 | 2 | 1 |
| 1977 | 0.35 | 0.42 | 0.15 | 0.24 | 0.24 | 0.27 | 0.15 | 0.15 | 0.15 | 0.30 | 0.39 | 0.15 | 5 | 4 |
| 1978 | 0.43 | 0.15 | 0.23 | 0.25 | 0.26 | 0.26 | 0.27 | 0.28 | 0.28 | 0.15 | 0.15 | 0.15 | 4 | 3 |
| 1979 | 0.27 | 0.31 | 0.33 | 0.34 | 0.36 | 0.37 | 0.37 | 0.38 | 0.39 | 0.40 | 0.40 | 0.40 | 0 | 0 |
| 1980 | 0.41 | 0.43 | 0.44 | 0.15 | 0.19 | 0.15 | 0.15 | 0.38 | 0.15 | 0.23 | 0.15 | 0.31 | 5 | 3 |
| 1981 | 0.40 | 0.42 | 0.43 | 0.44 | 0.44 | 0.44 | 0.15 | 0.17 | 0.18 | 0.19 | 0.19 | 0.19 | 1 | 0 |
| 1982 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.31 | 0.35 | 0.35 | 0 | 0 |
| 1983 | 0.36 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.15 | 0.23 | 0.24 | 0.25 | 1 | 0 |
| 1984 | 0.25 | 0.25 | 0.25 | 0.26 | 0.26 | 0.26 | 0.26 | 0.26 | 0.26 | 0.26 | 0.26 | 0.26 | 0 | 0 |
| 1985 | 0.34 | 0.15 | 0.15 | 0.17 | 0.18 | 0.20 | 0.21 | 0.21 | 0.21 | 0.21 | 0.25 | 0.27 | 2 | 2 |
| 1986 | 0.29 | 0.30 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.31 | 0.35 | 0.36 | 0.37 | 0.37 | 0 | 0 |
| 1987 | 0.38 | 0.38 | 0.38 | 0.38 | 0.42 | 0.44 | 0.45 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 1 | 0 |
| 1988 | 0.17 | 0.17 | 0.17 | 0.17 | 0.17 | 0.17 | 0.21 | 0.23 | 0.23 | 0.23 | 0.23 | 0.23 | 0 | 0 |
| 1989 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 0.20 | 0.21 | 0.21 | 0.25 | 0.26 | 0.26 | 0.26 | 4 | 2 |

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total breach | High Flow Breach |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1990 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.32 | 0.33 | 0 | 0 |
| 1991 | 0.38 | 0.39 | 0.39 | 0.40 | 0.40 | 0.40 | 0.40 | 0.40 | 0.40 | 0.40 | 0.15 | 0.42 | 1 | 1 |
| 1992 | 0.15 | 0.35 | 0.36 | 0.36 | 0.36 | 0.36 | 0.37 | 0.37 | 0.38 | 0.39 | 0.40 | 0.15 | 2 | 1 |
| 1993 | 0.17 | 0.17 | 0.23 | 0.27 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 0.38 | 0.42 | 0 | 0 |
| 1994 | 0.42 | 0.42 | 0.15 | 0.15 | 0.16 | 0.16 | 0.20 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 2 | 2 |
| 1995 | 0.22 | 0.24 | 0.24 | 0.24 | 0.24 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0 | 0 |
| 1996 | 0.25 | 0.15 | 0.35 | 0.36 | 0.36 | 0.36 | 0.44 | 0.15 | 0.15 | 0.25 | 0.25 | 0.25 | 3 | 2 |
| 1997 | 0.26 | 0.26 | 0.26 | 0.26 | 0.26 | 0.40 | 0.45 | 0.45 | 0.45 | 0.15 | 0.16 | 0.17 | 1 | 0 |
| 1998 | 0.17 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.18 | 0.23 | 0.24 | 0.25 | 0 | 0 |
| 1999 | 0.36 | 0.40 | 0.40 | 0.15 | 0.20 | 0.15 | 0.44 | 0.44 | 0.44 | 0.44 | 0.44 | 0.44 | 2 | 1 |
| 2000 | 0.15 | 0.25 | 0.29 | 0.30 | 0.31 | 0.31 | 0.33 | 0.33 | 0.33 | 0.38 | 0.41 | 0.15 | 2 | 0 |
| 2001 | 0.17 | 0.24 | 0.26 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.38 | 0.15 | 0.15 | 2 | 2 |
| 2002 | 0.15 | 0.17 | 0.18 | 0.18 | 0.18 | 0.18 | 0.19 | 0.15 | 0.15 | 0.18 | 0.19 | 0.19 | 3 | 3 |
| | | | | | | | | | | | | Ave | 1.96 | 1.41 |

High flow breaching

Low flow breaching

Appendix 4. Simulated monthly volumes (million $m^3$) in the East Kleinemonde Estuary for Future Scenario 2 (Van Niekerk *et al*., 2008).

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breach | High Flow Breach Flow |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1920 | 0.15 | 0.15 | 0.21 | 0.22 | 0.23 | 0.34 | 0.15 | 0.35 | 0.37 | 0.37 | 0.37 | 0.38 | 1 | 1 |
| 1921 | 0.38 | 0.15 | 0.25 | 0.25 | 0.25 | 0.25 | 0.35 | 0.41 | 0.15 | 0.15 | 0.15 | 0.26 | 4 | 3 |
| 1922 | 0.27 | 0.15 | 0.15 | 0.15 | 0.19 | 0.19 | 0.19 | 0.19 | 0.20 | 0.26 | 0.28 | 0.28 | 3 | 3 |
| 1923 | 0.29 | 0.32 | 0.35 | 0.36 | 0.36 | 0.36 | 0.36 | 0.39 | 0.39 | 0.39 | 0.40 | 0.41 | 0 | 0 |
| 1924 | 0.41 | 0.41 | 0.15 | 0.18 | 0.18 | 0.15 | 0.34 | 0.35 | 0.37 | 0.37 | 0.37 | 0.15 | 3 | 1 |
| 1925 | 0.20 | 0.20 | 0.22 | 0.23 | 0.28 | 0.31 | 0.31 | 0.31 | 0.32 | 0.32 | 0.32 | 0.33 | 0 | 0 |
| 1926 | 0.15 | 0.25 | 0.25 | 0.25 | 0.26 | 0.28 | 0.29 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 1 | 1 |
| 1927 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.23 | 0.15 | 4 | 2 |
| 1928 | 0.15 | 0.34 | 0.40 | 0.40 | 0.40 | 0.41 | 0.41 | 0.41 | 0.15 | 0.15 | 0.26 | 0.15 | 4 | 4 |
| 1929 | 0.15 | 0.23 | 0.23 | 0.23 | 0.23 | 0.42 | 0.15 | 0.15 | 0.18 | 0.19 | 0.23 | 0.29 | 2 | 1 |
| 1930 | 0.15 | 0.30 | 0.30 | 0.30 | 0.30 | 0.38 | 0.15 | 0.18 | 0.18 | 0.42 | 0.15 | 0.18 | 3 | 1 |
| 1931 | 0.15 | 0.15 | 0.15 | 0.36 | 0.36 | 0.36 | 0.36 | 0.36 | 0.36 | 0.38 | 0.38 | 0.15 | 4 | 4 |
| 1932 | 0.15 | 0.35 | 0.35 | 0.35 | 0.36 | 0.36 | 0.15 | 0.20 | 0.20 | 0.20 | 0.45 | 0.15 | 3 | 1 |
| 1933 | 0.15 | 0.19 | 0.20 | 0.21 | 0.22 | 0.15 | 0.29 | 0.29 | 0.29 | 0.15 | 0.15 | 0.15 | 5 | 3 |
| 1934 | 0.15 | 0.28 | 0.31 | 0.31 | 0.31 | 0.31 | 0.15 | 0.15 | 0.15 | 0.22 | 0.23 | 0.23 | 4 | 3 |
| 1935 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.26 | 0.26 | 0.26 | 0.26 | 0.26 | 0 | 0 |
| 1936 | 0.15 | 0.15 | 0.33 | 0.33 | 0.33 | 0.15 | 0.31 | 0.31 | 0.31 | 0.32 | 0.32 | 0.32 | 3 | 3 |
| 1937 | 0.32 | 0.34 | 0.15 | 0.15 | 0.15 | 0.25 | 0.36 | 0.38 | 0.38 | 0.38 | 0.38 | 0.38 | 3 | 2 |
| 1938 | 0.38 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.22 | 0.22 | 0.22 | 0.22 | 0.23 | 0.30 | 5 | 4 |
| 1939 | 0.15 | 0.19 | 0.15 | 0.15 | 0.15 | 0.15 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 5 | 3 |
| 1940 | 0.30 | 0.15 | 0.22 | 0.22 | 0.23 | 0.23 | 0.15 | 0.15 | 0.16 | 0.16 | 0.16 | 0.16 | 3 | 2 |
| 1941 | 0.16 | 0.20 | 0.15 | 0.32 | 0.34 | 0.34 | 0.34 | 0.42 | 0.44 | 0.44 | 0.45 | 0.15 | 2 | 0 |
| 1942 | 0.41 | 0.15 | 0.16 | 0.33 | 0.37 | 0.37 | 0.15 | 0.21 | 0.21 | 0.21 | 0.27 | 0.28 | 2 | 0 |
| 1943 | 0.28 | 0.15 | 0.31 | 0.31 | 0.32 | 0.15 | 0.23 | 0.30 | 0.31 | 0.31 | 0.31 | 0.15 | 3 | 2 |
| 1944 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.41 | 0.43 | 0.43 | 0.43 | 0 | 0 |
| 1945 | 0.15 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 | 1 | 0 |
| 1946 | 0.23 | 0.24 | 0.25 | 0.25 | 0.25 | 0.15 | 0.32 | 0.33 | 0.39 | 0.15 | 0.21 | 0.21 | 2 | 1 |
| 1947 | 0.21 | 0.23 | 0.24 | 0.24 | 0.24 | 0.25 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 2 | 2 |
| 1948 | 0.15 | 0.28 | 0.28 | 0.29 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 0.30 | 0.31 | 0.32 | 1 | 1 |
| 1949 | 0.32 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.16 | 0.16 | 5 | 4 |
| 1950 | 0.15 | 0.15 | 0.15 | 0.15 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.29 | 0.29 | 0.15 | 5 | 5 |
| 1951 | 0.15 | 0.15 | 0.15 | 0.16 | 0.27 | 0.31 | 0.32 | 0.15 | 0.20 | 0.20 | 0.20 | 0.15 | 3 | 2 |

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breach | High Flow Breach Flow |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1952 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 0.17 | 0.43 | 0.15 | 2 | 2 |
| 1953 | 0.15 | 0.15 | 0.26 | 0.26 | 0.26 | 0.15 | 0.25 | 0.15 | 0.31 | 0.34 | 0.15 | 0.31 | 5 | 4 |
| 1954 | 0.38 | 0.15 | 0.21 | 0.23 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.36 | 1 | 0 |
| 1955 | 0.39 | 0.15 | 0.17 | 0.17 | 0.29 | 0.34 | 0.35 | 0.37 | 0.38 | 0.38 | 0.38 | 0.15 | 2 | 0 |
| 1956 | 0.22 | 0.27 | 0.40 | 0.42 | 0.15 | 0.22 | 0.22 | 0.22 | 0.23 | 0.23 | 0.23 | 0.15 | 2 | 1 |
| 1957 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.29 | 0.42 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 2 | 2 |
| 1958 | 0.15 | 0.15 | 0.34 | 0.42 | 0.42 | 0.15 | 0.24 | 0.27 | 0.27 | 0.27 | 0.27 | 0.28 | 1 | 0 |
| 1959 | 0.28 | 0.29 | 0.29 | 0.30 | 0.30 | 0.30 | 0.30 | 0.15 | 0.21 | 0.21 | 0.21 | 0.27 | 1 | 0 |
| 1960 | 0.44 | 0.15 | 0.15 | 0.16 | 0.20 | 0.21 | 0.21 | 0.30 | 0.32 | 0.33 | 0.33 | 0.33 | 1 | 0 |
| 1961 | 0.33 | 0.37 | 0.39 | 0.39 | 0.39 | 0.15 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 1 | 1 |
| 1962 | 0.15 | 0.33 | 0.33 | 0.15 | 0.19 | 0.15 | 0.15 | 0.44 | 0.44 | 0.15 | 0.20 | 0.20 | 5 | 3 |
| 1963 | 0.20 | 0.20 | 0.20 | 0.20 | 0.15 | 0.39 | 0.39 | 0.39 | 0.15 | 0.20 | 0.20 | 0.15 | 3 | 2 |
| 1964 | 0.39 | 0.42 | 0.42 | 0.42 | 0.42 | 0.43 | 0.43 | 0.15 | 0.30 | 0.35 | 0.38 | 0.39 | 1 | 0 |
| 1965 | 0.15 | 0.15 | 0.34 | 0.34 | 0.34 | 0.34 | 0.34 | 0.41 | 0.43 | 0.43 | 0.15 | 0.19 | 3 | 1 |
| 1966 | 0.21 | 0.22 | 0.22 | 0.22 | 0.27 | 0.29 | 0.33 | 0.15 | 0.15 | 0.34 | 0.43 | 0.45 | 2 | 2 |
| 1967 | 0.45 | 0.45 | 0.45 | 0.45 | 0.45 | 0.15 | 0.22 | 0.23 | 0.15 | 0.15 | 0.15 | 0.18 | 4 | 2 |
| 1968 | 0.18 | 0.18 | 0.18 | 0.18 | 0.21 | 0.45 | 0.15 | 0.15 | 0.16 | 0.16 | 0.16 | 0.16 | 1 | 0 |
| 1969 | 0.16 | 0.16 | 0.16 | 0.16 | 0.16 | 0.16 | 0.16 | 0.16 | 0.17 | 0.17 | 0.15 | 0.15 | 2 | 2 |
| 1970 | 0.15 | 0.22 | 0.15 | 0.40 | 0.41 | 0.41 | 0.15 | 0.28 | 0.28 | 0.28 | 0.33 | 0.34 | 3 | 3 |
| 1971 | 0.38 | 0.38 | 0.41 | 0.41 | 0.15 | 0.20 | 0.20 | 0.20 | 0.20 | 0.22 | 0.23 | 0.24 | 1 | 0 |
| 1972 | 0.24 | 0.24 | 0.24 | 0.24 | 0.24 | 0.25 | 0.26 | 0.27 | 0.27 | 0.27 | 0.34 | 0.37 | 0 | 0 |
| 1973 | 0.37 | 0.15 | 0.19 | 0.28 | 0.41 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.38 | 7 | 6 |
| 1974 | 0.38 | 0.38 | 0.38 | 0.38 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 | 0.41 | 0.15 | 1 | 1 |
| 1975 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.29 | 0.33 | 0.33 | 0.33 | 0.15 | 0.15 | 0.17 | 3 | 3 |
| 1976 | 0.29 | 0.37 | 0.37 | 0.37 | 0.15 | 0.36 | 0.37 | 0.15 | 0.26 | 0.29 | 0.29 | 0.29 | 2 | 1 |
| 1977 | 0.29 | 0.42 | 0.15 | 0.23 | 0.23 | 0.30 | 0.15 | 0.15 | 0.37 | 0.40 | 0.40 | 0.40 | 3 | 3 |
| 1978 | 0.15 | 0.18 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.21 | 0.21 | 0.15 | 0.15 | 0.15 | 4 | 3 |
| 1979 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.17 | 0.17 | 0.17 | 0.20 | 0 | 0 |
| 1980 | 0.21 | 0.28 | 0.30 | 0.15 | 0.27 | 0.15 | 0.15 | 0.33 | 0.39 | 0.39 | 0.15 | 0.23 | 4 | 3 |
| 1981 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.42 | 0.15 | 0.17 | 0.18 | 0.18 | 0.18 | 1 | 0 |
| 1982 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.15 | 0.25 | 0.25 | 1 | 1 |
| 1983 | 0.29 | 0.30 | 0.30 | 0.30 | 0.30 | 0.31 | 0.31 | 0.31 | 0.15 | 0.37 | 0.39 | 0.39 | 1 | 1 |
| 1984 | 0.39 | 0.39 | 0.39 | 0.40 | 0.41 | 0.41 | 0.41 | 0.41 | 0.41 | 0.41 | 0.41 | 0.41 | 0 | 0 |
| 1985 | 0.15 | 0.15 | 0.15 | 0.17 | 0.17 | 0.21 | 0.22 | 0.22 | 0.22 | 0.22 | 0.35 | 0.40 | 3 | 2 |
| 1986 | 0.44 | 0.15 | 0.15 | 0.15 | 0.16 | 0.17 | 0.17 | 0.17 | 0.27 | 0.30 | 0.32 | 0.34 | 1 | 0 |

| Year | Oct | Nov | Dec | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Total Breach | High Flow Breach Flow |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1987 | 0.34 | 0.34 | 0.34 | 0.34 | 0.15 | 0.22 | 0.22 | 0.23 | 0.23 | 0.23 | 0.23 | 0.25 | 1 | 0 |
| 1988 | 0.27 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.40 | 0.44 | 0.44 | 0.45 | 0.45 | 0.15 | 1 | 0 |
| 1989 | 0.15 | 0.15 | 0.15 | 0.15 | 0.20 | 0.30 | 0.32 | 0.32 | 0.43 | 0.15 | 0.15 | 0.15 | 5 | 3 |
| 1990 | 0.20 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.21 | 0.31 | 0.34 | 0 | 0 |
| 1991 | 0.15 | 0.19 | 0.19 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.15 | 0.15 | 3 | 2 |
| 1992 | 0.15 | 0.32 | 0.32 | 0.32 | 0.32 | 0.32 | 0.33 | 0.33 | 0.38 | 0.39 | 0.43 | 0.15 | 2 | 1 |
| 1993 | 0.20 | 0.20 | 0.38 | 0.15 | 0.19 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.15 | 0.24 | 2 | 0 |
| 1994 | 0.24 | 0.24 | 0.15 | 0.15 | 0.15 | 0.15 | 0.26 | 0.29 | 0.29 | 0.29 | 0.29 | 0.29 | 2 | 2 |
| 1995 | 0.29 | 0.34 | 0.36 | 0.36 | 0.36 | 0.37 | 0.37 | 0.37 | 0.37 | 0.37 | 0.38 | 0.38 | 0 | 0 |
| 1996 | 0.38 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.40 | 0.15 | 0.15 | 0.25 | 0.25 | 0.25 | 4 | 3 |
| 1997 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.15 | 0.29 | 0.29 | 0.29 | 0.30 | 0.34 | 0.36 | 1 | 1 |
| 1998 | 0.37 | 0.37 | 0.38 | 0.38 | 0.38 | 0.39 | 0.39 | 0.39 | 0.39 | 0.15 | 0.19 | 0.21 | 1 | 0 |
| 1999 | 0.15 | 0.25 | 0.25 | 0.15 | 0.27 | 0.15 | 0.42 | 0.42 | 0.42 | 0.42 | 0.42 | 0.44 | 3 | 3 |
| 2000 | 0.15 | 0.38 | 0.44 | 0.15 | 0.16 | 0.17 | 0.20 | 0.20 | 0.20 | 0.34 | 0.42 | 0.15 | 3 | 0 |
| 2001 | 0.19 | 0.31 | 0.35 | 0.35 | 0.35 | 0.35 | 0.35 | 0.36 | 0.36 | 0.15 | 0.15 | 0.15 | 3 | 3 |
| 2002 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.18 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 3 | 3 |
| | | | | | | | | | | | | | 2.34 | 1.55 |

High flow breaching

Low flow breaching