# MODELLING OF THE WATER BALANCE AND NUTRIENT DYNAMICS OF MHLANGA ESTUARY

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# ABSTRACT


Wastewater discharge into a temporary open/closed estuary (TOCE) system introduces two main concerns namely (1) the effects on the water balance of the system (quantity) and (2) the effects on the nutrient dynamics (water quality). Changes to mouth breaching patterns can severely impact the hydrological and ecological functioning of TOCEs, while excessive nutrient loading can lead to eutrophic conditions and algal blooms. Algal blooms occur when residence times during closed mouth conditions exceed the time scale for growth of the microalgal community.

The aim of this study was to formulate a model in order to predict eutrophication events using the Mhlanga Estuary as a case study. The Mhlanga Estuary is situated approximately 19 km northeast of Durban and has a small catchment (<100km$^2$). The Phoenix and Mhlanga wastewater treatment works (WWTW) collectively discharge approximately 20Ml of treated effluent into the Mhlanga River per day. A simple daily-time-step water balance model was selected to model the hydrodynamics of the system. The model included various inputs and outputs of the system, residence time, storage, breaching water levels and time for mouth closure to occur. The result of the water balance model was a daily prediction of the mouth state and volume, and an indication of the breaching frequency. Observed mouth state data and measured water levels were used to test the model.

In order to predict eutrophication events and trends at the Mhlanga Estuary, it was required that the conditions at which this would occur be investigated. This included the collection of samples (physico-chemical and chlorophyll-*a*) on a weekly basis for three months, a period that included three breaching events. Due to the complexity required in developing a nutrient dynamics model, a simpler approach was selected. The grey water index (GWI) was formulated in order to account for nutrient loadings into the estuary. WWTW discharge data were provided by eThekwini Municipality Water and Sanitaion (EMWS). Initial results showed that under ideal conditions, an algal bloom would occur approximately fourteen days following re-closure of the Mhlanga mouth. The eutrophication index ($E_i$) was then formulated to account for both residence time and nutrient concentrations. The $E_i$ at which eutrophic conditions can be expected was found to be about 50 %. It is important to note that this value for $E_i$ is expected to be site specific and only accounts for the Mhlanga Estuary, but the concept can be generalized to other similar estuaries.

Water levels simulated using the water balance model and observed mouth state data produced similar levels to those measured by DWAF. Following simulations of different flow scenarios (75% and 150% increase in WWTW discharges), it was found that an increase in capping flows resulted in more frequent breaching events and longer open mouth conditions. The risk of eutrophic conditions also increased with an increase in WWTW capping flows. Algal blooms are predicted to continue despite more frequent breaching events induced by an increase in capping flows.

# PREFACE

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I, Robynne Angela Lawrie, hereby declare that the whole of this dissertation is my own work and has not been submitted in part, or in whole to any other University. Where use has been made of the work of others, it has been duly acknowledged in the text. This research work was carried out in the Centre for Research in Environmental, Coastal and Hydraulic Engineering, School of Civil Engineering, Surveying and Construction, University of KwaZulu-Natal, Durban, under the supervision of Professor D. D. Stretch.

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As the candidates supervisor I have approved this dissertation for submission

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# ABBREVIATIONS

| | |
|---|---|
| A | Surface area |
| B | Length of the sand berm |
| $C_D$ | Diluted nutrient concentration |
| Chl-*a* | Chlorophyll-*a* |
| $C_S$ | River nutrient concentration |
| CSIR | Council for Scientific and Industrial Research |
| $C_W$ | Wastewater nutrient concentration |
| Daily-Inflow | The daily inflow into the estuary |
| dFDC | Daily Flow Duration Curve |
| DIN | Dissolved inorganic nitrogen |
| DIP | Dissolved inorganic phosphorous |
| DO | Dissolved oxygen |
| DWAF | Department of Water and Forestry |
| $E_{i\ crit}$ | Critical eutrophication index |
| $E_i$ | Eutrophication index |
| $E_p$ | Potential for eutrophication to occur |
| EPA | United States Office of Water Environmental Protection Agency |
| FDC | Flow Duration Curve |
| GWI | Grey water index |
| $GWI_{crit}$ | Critical grey water index |
| hc | Hydraulic conductivity |
| H | Water level |
| $H_{max}$ | Maximum water level |
| $H_{min}$ | Minimum water level |
| i | Hydraulic gradient |
| $I_z$ | Intensity of light at depth, z |
| k | Hydraulic conductivity |

| | |
|---|---|
| $K_d$ | Vertical attenuation coefficient |
| KZN | KwaZulu-Natal |
| L | The length of flow over which head difference occurs |
| mFDC | Monthly Flow Duration Curve |
| N | Nitrogen |
| NTU | Nephelometric turbidity units |
| P | Phosphorous |
| PAR | Photosynthetically available radiation |
| $Q_S$ | River discharge |
| $Q_W$ | Wastewater discharge |
| S | Storage |
| $S_{max}$ | Maximum storage |
| $T_B$ | Time required for an algal bloom |
| $T_{close}$ | The number of days until closure after a breaching event |
| TOCE | Temporarily open/closed estuary |
| $T_Q$ | Filling time |
| $T_R$ | Residence time |
| WWTW | Wastewater treatment works |
| $Z_{eu}$ | Euphotic depth |
| $z_{tot}$ | Total depth |

# 1. INTRODUCTION

## 1.1 Introduction

Estuaries are ecologically important as they are home to an environment of high biodiversity and serve as a nursery for juvenile fish. Estuaries are utilised by humans for recreation, subsistence fishing and tourism. Temporarily open/closed estuaries (TOCE) make up the vast majority of South Africa's 258 estuaries. These are not permanently open to the sea and the mouth dynamics are generally driven by the river inflows into the system.

The South African National Water Act (Act. 36) of 1998 aims to ensure the sustainable management of water resources in order to meet the needs of humans and the environment. The National Water Act brought about an improvement to regulations due to the inclusion of groundwater, sediment, riparian habitat and estuaries in the definition of *water resources* (Wepener *et al.*, 2006). This included the requirement that the quantity and quality of water needed by all aquatic systems be determined (Perissinotto *et al.*, 2004). Intervention by human activities with the natural quantity and quality of river water delivered to an estuary influences the size and functioning of the estuarine system involved. Consequently, the estuarine environment and the biodiversity are impacted on.

The discharge of sewage into an estuary system introduces two main concerns. Firstly, the added quantity of inflow interferes with the water balance of the system, and secondly the nutrient dynamics (water quality) are affected. For example, changes to mouth breaching patterns can severely impact the physico-chemical environment, while high nutrient loads may lead to algal blooms. Blooms occur when the time required for the estuary to fill during closed mouth conditions exceeds the time scale for growth of the microalgal community. In situations where nutrient supplies are plentiful, there is an ideal situation for blooms to occur.

Due to the sensitive and dynamic nature of temporary open/closed estuaries, it is imperative that the impact of proposed changes on the environment be investigated. Insights into the processes occurring within the estuarine environment are required in order to specify mitigation and/or preventative measures. This study should therefore assist in implementing best management practice decisions with regard to minimizing impacts on water resources.

## 1.2 Objectives

The objectives of this study were:

1. To investigate the water balance of a temporary open estuarine system and the factors influencing the mouth dynamics.
2. To investigate eutrophication processes and how they develop.
3. To formulate a model (incorporating 1 and 2) in order to predict future eutrophication events at the Mhlanga Estuary, and which can be used at other similar estuaries.
4. To test the model using observed data collected from the Mhlanga Estuary.

## 1.3 Outline of the Dissertation

The following chapters included in this dissertation are outlined as follows:

*Chapter Two* presents a review of literature concerning temporarily open/closed estuaries. The hydrodynamics and water balance of temporarily open/closed estuaries are discussed, followed by previous relevant investigations. The abiotic and biotic components of the estuarine environment are discussed, including nutrient dynamics. Eutrophication and the policy framework in South Africa concerning it are introduced.

*Chapter Three* outlines the characteristics of the case study: Mhlanga Estuary. Laboratory procedures and equipment associated with the collection of samples and water quality analysis are discussed. The various parameters included in the hydrodynamic and eutrophication prediction models are introduced.

*Chapter Four* presents the results of the fieldwork, including physico-chemical and chlorophyll-*a* samples and flow measurements. Nutrient concentrations and phytoplankton biomass are analysed. Various scenarios are simulated using observed and simulated mouth state data.

*Chapter Five* presents the conclusions of the study followed by recommendations for future research.

# 2. LITERATURE REVIEW

## 2.1 Introduction

Temporarily open/closed estuaries (TOCE) comprise 71% of South Africa's 258 estuaries. These estuaries generally remain closed during the dry season, while open phases are more prevalent during the rainy season (Perissinotto *et al.*, 2004). Water exchange between the sea and the estuary is governed by the mouth state of the estuary. The estuary mouth state is in turn governed by opposing forces: closing forces due to wave- and flood-tide driven sediment suspension and deposition in the mouth, and scouring forces due to ebb-tidal currents govern the state of mouth (Schumann et al., 1999). TOCEs are generally characterised by small river catchments, thus making them sensitive to external weather patterns and flow conditions (Perissinotto *et al.*, 2004).

Periodic mouth breaching of TOCEs create a dynamic system in which rapid physico-chemical fluctuations occur. When compared to permanently open estuaries, the biological functioning of the estuary is affected and a complex environment is created. Prolonged closed mouth conditions are reflected in water levels and the water column structure (Schumann *et al.*, 1999). TOCEs create an ideal environment for temperature and salinity stratification as well as high nutrient loadings and depleted dissolved oxygen concentrations to occur (Perissinotto *et al.*, 2004).

Perched TOCEs create major disturbances when they breach, as the estuary essentially empties, exposing formerly submerged areas of the estuary bed. This may result in scouring of substantial quantities of accumulated sediments in the estuary during high outflows (Stretch and Parkinson, 2006). Large sediment deposits near the mouth of the estuary follow mouth-breaching events brought on by high river flows. These are subsequently conserved upon closure (Cooper, 2001). These processes continuously alter the morphology of the estuary. The hydrodynamics of an estuarine system is therefore the driving force behind the functioning and sustainability of TOCEs.

## 2.2 Hydrology and Water Balance

The hydrology of estuarine systems is complex and variable. Abstractions and discharges as a result of anthropogenic activities result in changes to flows, which invariably affect the dynamics of the mouth. This can significantly impact the overall functioning of the system (Perissinotto *et al.*, 2004).

In order to simplify the complex hydrological functioning of a TOCE system, it is conceptualised in terms of a dynamic storage system with variable inputs and outputs, as illustrated in Figure 2.1 (Perissinotto *et al.*, 2004). Equation 2.1 equates the storage system, where the TOCE signifies a storage volume, S(t), with variable inputs I(t) and outputs O(t) as a function of time.

$$\frac{dS}{dt} = I(t) - O(t) \qquad (2.1)$$

During closed mouth conditions, inputs into the system comprise mainly stream inflow, rainfall, overwash and groundwater inflow. Outputs comprise losses due to evaporation and seepage (Perissinotto *et al*, 2004). Both seepage and evaporation depend on storage, as seepage increases as the water level rises and evaporation increases as the surface area increases (Perissinotto *et al.*, 2004). Seepage losses tend to dominate evaporation for small perched estuaries, such as those that occur in South Africa (Perissinotto *et al.*, 2004). The storage system is further influenced by anthropogenic activities, such as abstractions for domestic use and irrigation, runoff from roads and farmlands, discharges from wastewater treatment works and dams. When the mouth is open, tidal exchange flows can occur.

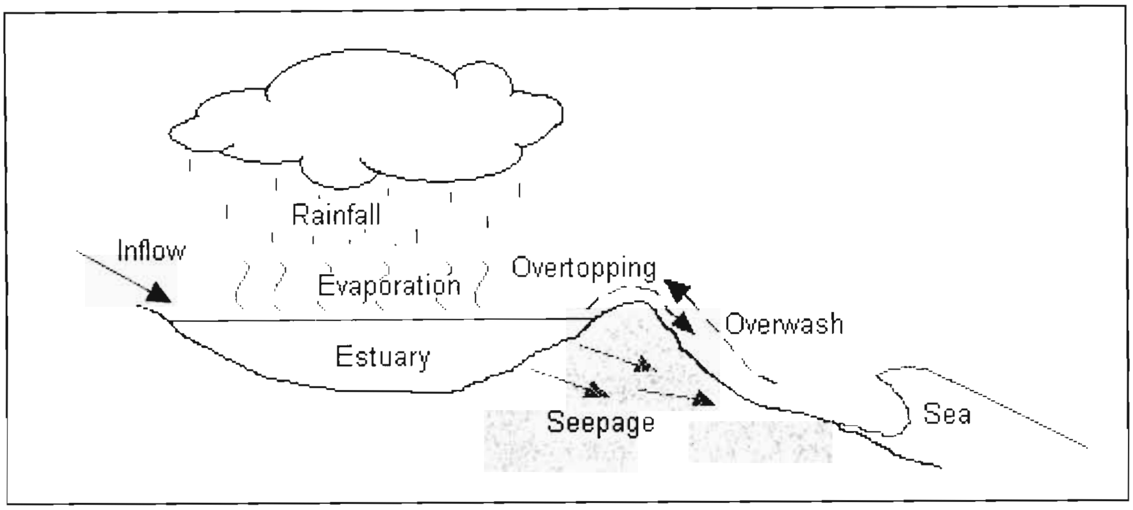


**Figure 2.1 Schematic of a perched TOCE conceptualised in terms of a dynamic storage system.**

Residence time is a broadly used concept that expresses how fast something moves through a system in equilibrium. It is the average time a substance spends within a specified system. The term has been used in plug flow reactors, determining the susceptibility of groundwater to pollution and the characterization of suspended sediment (Stewart *et al.,* 2005). Monsen *et al.* (2002) comment on the use of residence time for transport time scales and the necessity in defining the term in its context. Residence time is an important factor when considering the hydrological functioning of an estuary as it can have significant implications on the ecology of the system (Hagy *et al.*, 2000, Perissinotto *et a*l., 2004). In the case of TOCEs, residence time is defined as the period of time between breaching events that the water is stored in the estuary (Perissinotto *et al.*, 2004).

Mouth dynamics are an important aspect of the overall functioning of TOCEs. In order to model the process, an understanding of the breaching and closing mechanisms is required.

### 2.2.1 Breaching mechanisms

Mouth breaching is a natural phenomenon and is essential for the functioning of TOCEs. There are numerous natural breaching mechanisms including scour due to overtopping of the berm, seepage through the berm and wave action (Stretch and Parkinson, 2006). Artificial breaching has been used to increase the occurrence of open mouth conditions where river flow reduction has resulted in a severe decline in breaching events (Perissinotto *et al.*, 2004). There has been controversy surrounding the subject of artificial breaching as frequent breaching in this manner can have serious adverse effects on estuarine ecosystems.

The overtopping breaching process generally occurs under high river flow or flood conditions and/or during spring high tide. It occurs when the sand-bar is overtopped by the water level within the estuary subsequently scouring a channel to the sea. Under low flow conditions, the estuary is more likely to breach due to seepage. In perched estuaries, such as the Mhlanga Estuary (about 0.7m above MSL), water is able to seep through the porous sand berm from the estuary to the sea. This results in "sand-piping" erosion to the beach face, as illustrated by Plate 2.1, eventually resulting in failure of the sand-bar. Perissinotto *et al.* (2004) suggest that this mechanism requires a time scale longer than the typical tidal half cycle in order to develop. Therefore, breaching generally occurs during neap tide phases where the hydraulic gradients are more consistent over the tidal period.

**Plate 2.1 Sand-piping erosion of the beach face as a result of seepage through the sandbar.**

Stretch and Parkinson (2006) performed model experiments designed to test the influence of storage volumes on the breaching process. The results indicated that the breach width scales on the third power of the total volume of water that flows through the breach. It was further established that this relationship was consistent with breaching events at actual estuaries with storage volumes orders of magnitude higher than those for the models.

### 2.2.2 Closing mechanisms

After a breaching event, a TOCE remains open for a period of time until the sand berm is restored again. This period of time is dependent on two opposing forces. The forces maintaining an open state include river flows and/or tidal exchange inflows that scour sediment from the mouth. Physical processes that allow for sediment transport and deposition cause closure by rebuilding the sand berm (Schumann *et al.*, 1999). These include longshore and cross-shore sediment transport. A concise description of these processes and their effects on mouth closure of small and large estuaries is given in Zeitsman (2003).

### 2.2.3 Tidal flows during open mouth conditions

Many characteristics of estuaries are attributed to the tidal exchanges that are responsible for the input and output of water through an open estuary mouth. The manner in which the exchanges occur is strongly dependent on the mouth configuration. Tidal variations in the sea follow sinusoidal curves. Shallow depths of coastal seas and mouth constrictions of estuaries distort the tidal signal resulting in asymmetric tidal exchanges. Flood dominant systems are characterised by a longer ebb tide in the estuary and a shorter, but more intense flood tide, while ebb dominant systems are characterised by a longer flood tide and a shorter ebb tide (Schumann *et al.*, 1999). The tidal prism is the volume of water that is exchanged between the sea and an estuary over each tidal cycle. The tidal prism is important because it controls the salinity regime of the estuary, as well as biological exchanges with the ocean. It also plays a role in the reclosure of the mouth.

## 2.3 Studies Relating Flow Rate and Mouth State

The potential of a TOCE to reach an eutrophic state is a function of both residence time and nutrient availability. Prolonged residence times, as a result of persistent closed mouth conditions, often provide sufficient time periods for algal growth to occur. It is therefore important that the mouth state be known in order to predict eutrophic conditions. Several statistical and water balance models have been used to predict the mouth state of TOCEs, however the complex hydrological functioning of TOCE systems introduces several difficulties (Smakhtin, 2004, Stretch and Zeitsman, 2004, Londal, 2005). These include the lack of quantifiable knowledge of many of the factors (e.g. flow and storage) involved in the hydrological functioning of estuaries in South Africa. The following sections give a brief overview of some of the models previously used in the prediction of mouth state.

### 2.3.1 Flow duration curves

Smakhtin (2004) noted that the major water balance component of an estuary is often the continuous inflow from the catchment. Catchments of a scale smaller than 500 km$^2$ generally respond quickly to rainfall events and therefore models for the flows should have a time resolution of at least one day (Smakhtin, 2004). Smakhtin provides a method based on flow duration curves (FDCs) for generating continuous daily inflow time series to small-ungauged TOCEs by using flow/rainfall records from the closest source to the estuary.

Flow duration curves (FDCs) provide a graphical representation of the variability of flow at a site, thereby demonstrating the relationship between the magnitude of the flow and its frequency (Smakhtin, 2004). FDCs are represented as log-normal plots of flow versus the percentage of time that a specific flow is either exceeded or equalled. The chronological order in which flows occur is not considered in the frequency distribution of the flows. A FDC may be constructed from either daily (dFDC) or monthly data (mFDC). These may be estimated from the entire record period available or from all of the same calendar months for a selected period (Smakhtin, 2000). Stretch and Zietsman (2004) analysed normalized annual dFDCs for the Mhlanga and Mdloti Estuaries in South Africa, and used them to quantify changes in the functioning of those systems due to WWTW discharges.

### 2.3.2 Statistical flow-mouth state models

The National Water Act (Act 36 of 1998) aimed to ensure that South African water resources were sustained and fairly/deservingly distributed. In order to implement these requirements, Resource Directed Measures (RDM), as explained by Wepener *et al.* (2006), were established. An RDM study aims at providing the flow ranges of every state (open, closed and partly open) for a particular estuary and incorporates both the abiotic and biotic functions. In order to analyse the effects of flow variations on mouth dynamics, statistical flow-duration analysis is linked with a model that describes the relationship between mouth state and flow. This usually consists of specifying flow thresholds that define different mouth states.

Stretch and Zietsman (2004) developed a conceptual model to relate mouth dynamics to flow in combination with FDCs. The Mhlanga and Mdloti Estuaries were used as the case studies. Results were compared with results from RDM studies by the CSIR (2002, 2003) on the same two estuaries. Stretch and Zietsman (2004) found that these flows were only useful in determining the mouth state in high and low flow regimes and did not reflect the observed situation at intermediate flows. Zietsman (2003) explained that for intermediate flows the mouth state is intermittently open and may be classified as in an open/closed regime. Both the RDM method and that proposed by Stretch and Zietsman (2004) do not consider the water level of the estuary, which observations suggest plays a key role in the mouth dynamics.

### 2.3.3 Water balance models

Smakhtin (2004) selected a daily time-step reservoir water-balance model developed by Hughes and Ziervogel (1998) in order to simulate estuarine mouth conditions, by equating an estuary to a reservoir. This was done by equating the sand berm of an estuary to the spillway of a reservoir. The estuarine reservoir water-balance model is defined by the change in storage equating to the inputs (inflow and precipitation) minus the outputs (outflow to the sea, seepage through the berm and evaporation). It is assumed that the inflow/seepage from the sea is negligible and that mouth openings are driven solely by inflow from the feeding stream.

Smakhtin's model would not be suitable for perched TOCEs as it does not account for the catastrophic breaching events observed at perched estuaries as a result of high outflows due to scouring and tidal exchange. Smakhtins model does not attempt to incorporate details of the mouth opening and reclosure processes such as breach size and sediment transport processes that drive the reclosure process.

An investigation by Londal (2005) into the proposed statistical mouth state model by Stretch and Zeitsman (2004) produced a daily time step water balance model specifically calibrated for the Mhlanga Estuary. The model includes the effects of the water level of the estuary as well as the flows on the breaching process (Londal, 2005 after Stretch and Zeitsman, 2004). In order to establish the water level of the estuary (given inflow, evaporation, seepage and storage) the water balance method was applied to the daily inflow and used to generate daily mouth state predictions (Londal, 2005). These predictions were later translated into the percentage of time the estuary remained open and closed and also the period between breaching events.

Calibration of the water balance model was performed using mouth state data and river inflows from March 2002 to March 2005. The average time taken for the estuary mouth to close was estimated as 9 days. This includes the time required for the berm to redevelop and recover sediment lost due to scouring. The maximum seepage through the berm was established as 0.15 $m^3/s$. The sensitivity of the water balance model to key parameters was investigated. The maximum storage and the time for mouth closure were found to have the greatest effect on the mouth state predictions.

Table 2.1 presents a comparison of the results using the water balance model with those from the statistical model (Section 2.3.2). The model indicated that should the WWTW discharges not exist (natural state), the mouth state would predominantly be closed and if the WWTWs increased discharge levels by 75 %, the breaching frequency would increase.

**Table 2.1 Percentage time that mouth phase is closed using the water balance model and the proposed statistical model (Londal, 2005).**

| | **Water Balance Model (% closed)** | **Statistical Model (% closed)** |
|---|---|---|
| Natural state (no WWTWs) | 86 | 87 |
| Present state (with WWTWs) | 71 | 64 |
| WWTWs + 75 % increase | 64 | 45 |

For the period used to calibrate the models, it was observed that the mouth state was closed 63 % of the time. This agreed well with the result found using the statistical model. The water balance model and statistical model agreed well when no WWTWs were considered. Since the water balance model accounts for the maximum water level, it was assumed to be the more accurate model. Therefore, the daily time step water balance model by Londal (2005) was used as the basis for the present study.

## 2.4 Eutrophication

Eutrophication is a natural process in which enrichment of water by nutrients occurs, resulting in an increased production of organic material. It is associated with the natural ageing process of estuaries, lakes and other water bodies over tens of thousands of years (natural eutrophication). Anthropogenic activities have accelerated this process, resulting in many undesirable conditions, such as changes in primary production, biological structure and the water quality of many estuarine environments (Walmsley, 2000). This is referred to as cultural eutrophication. By affecting primary production, long term changes in the food web structure may occur which in turn may have an effect on the feeding strategies of consumers (Livingston, 2003).

Excessive cultural eutrophication caused by anthropogenic activities has become a worldwide pollution concern. The increase in the human population, along with urbanisation and an influx of people to coastal areas has put tremendous pressure on the environment and especially on natural water resources. The quantitative and qualitative effects caused by human activities on aquatic ecosystems have highlighted the importance of the application of sound waste management principles. Walmsley (2000) suggests that the main strategy is for nutrient reduction achieved by waste minimisation and source control. This requires the establishment of environmental objectives and is, to a large degree, dependent on the involvement of stakeholders and water users.

Vollenweider (1970) provides a concise representation of the history of eutrophication and how there already existed signs in biblical passages, B.C. The study indicates that it was only after the 1940s that literature involving eutrophication became available. Furthermore is the indication of widespread eutrophication around that time especially in highly developed countries, including mention of South Africa. Vollenweider (1970) states the importance of research and the need for prevention and mitigation.

### 2.4.1 Waste management and eutrophication of TOCEs

In the past, wastewater disposal has been based on the premise that "the solution to pollution is dilution" (Peavy *et al.*, 1985). Despite dilution being a powerful self-cleaning method, its efficiency is dependent on the principle that relatively small quantities of waste are discharged into large bodies of water. However, population growth and industrial activities have put pressure on dilution capacity. This has disqualified the option of diluting raw or poorly treated wastewaters into rivers (Peavy *et al.*, 1985).

Algal blooms and premature mouth breaching are among some of the problems caused by excessive nutrient loadings into TOCEs from wastewater discharge (Perissinotto *et al.*, 2004). Excessive nutrient enrichment together with prolonged residence times, during closed mouth conditions, may have adverse effects on the inorganic chemistry of the system. Furthermore is the concern of escalating biological oxygen demand throughout the water-column and ultimately the establishment of anoxic conditions in deeper layers of TOCEs.

### 2.4.2 Problems associated with eutrophication in TOCEs

Eutrophication caused by excessive nutrient loadings can destroy the natural balance of an aquatic ecosystem. Walmsley (2000) and Livingston (2003) highlighted the numerous long and short term problems associated with the eutrophication of TOCEs. These are summarised in Table 2.2.

**Table 2.2 Long and short term problems associated with the eutrophication of TOCE (Walmsley, 2000 and Livingston, 2003).**

| Impacts | Eutrophication Problems |
|---|---|
| Ecological | changes to the ecological community structure and loss of biodiversity |
| | nuisance algal blooms |
| | changes in algal dominance |
| | increasing dominance by blue-green algae |
| | increasing occurrence of toxic algae blooms |
| | fish migration interruptions |
| | fish and invertebrate kills due to hypoxia/anoxia |
| Water quality | water quality problems |
| | deoxygenation of bottom waters with associated chemical effects |
| | increased turbidity |
| | decreased light availability |
| Anthropogenic (recreaction, health, economic) | interference in recreation activities (swimming, fishing etc) |
| | human health problems (skin complaints, gastroenteritis etc) |
| | undesirable aesthetic conditions |
| | decrease in property values |
| | interference with irrigation and livestock water systems (may even cause livestock mortality). |
| | clogging of reticulation systems |

### 2.4.3 Eutrophication and policy in South Africa

Although eutrophication was recognised as a major water resource management concern in South Africa over 30 years ago, it is still prevalent throughout the country (Walmsley, 2000). It is of particular concern in populated industrialised areas.

The National Water Act (Act 36 of 1998) and the National Environmental Management Act (Act 107 of 1998) include new perspectives on waste management and eutrophication control (Walmsley, 2000). Both Acts require that environmental integrity is protected and waste discharges are minimised and treated accordingly. Walmsley (2000) states that there is however, still a need to remobilize and redevelop the country's capacity to manage eutrophication. The importance of immediate research is outlined by Walmsley (2000). The emphasis should be on: qualitatively assessing the eutrophication issue in terms of its extent and trends; the source of nutrients and levels entering aquatic systems; and the actual social and economic costs of the problem on a national basis is outlined by Walmsley (2000).

## 2.5 The Abiotic Environment

The abiotic environment is dependent on the river flow, hydrodynamics and the tidal activity within a TOCE (Allanson *et al.*, 1999). The effects of light, temperature, salinity, dissolved oxygen concentration and nutrients can have significant impacts on the biota of the system collectively and/or individually.

### 2.5.1 Light

The distribution and abundance of algae and submerged vegetation is dependent on light availability and thus is an important consideration when investigating the ecology of a system (Scheffer, 2004). Irradiance is the flux of solar radiation falling on a unit area (Tett, 2003) and often includes all ranges of the solar energy spectrum. However, the amount and quality of light available for algal photosynthesis is photosynthetically available radiation (PAR, 400 nm – 700nm). In TOCEs the PAR is a function of weather conditions, light intensity and the concentration of suspended solids in the water column. The intensity of light decreases exponentially with depth due to absorption and scattering. Turbidity, measured in nephelometric turbidity units (NTU), is a result of suspended and colloidal matter and is representative of the ability of water to scatter and absorb light (Gama *et al.* 2005). Turbidity is related to clarity as it affects the aesthetic quality of water. A decrease in clarity may be attributed to algal blooms and suspended sediment (Clesceri *et al.*, 1998; Scheffer, 2004). The discharge of wastes, such as sewage and storm water runoff, can contribute significantly to the turbidity of a water body (DWAF, 1998).

The vertical attenuation coefficient, $K_d$, as described by Kirk (1994) is "the best single parameter in terms of which to compare the light attenuation properties of one water-body with another." It can be estimated by using the equation (Scheffer, 2004):

$$K_d = \frac{\ln\left(\frac{I_{z_1}}{I_{z_2}}\right)}{z_2 - z_1} \tag{2.2}$$

Where $I_{z2}$ is the intensity of light at depth, $z_2$, and $I_{z1}$ is the intensity of light just below surface, at depth, $z_1$. In a study conducted by Thomas *et al.* (2005), the maximum light attenuation was found during open mouth conditions. The depth below which algae are considered to be incapable of maintaining a positive net photosynthesis is called the euphotic depth (Scheffer, 2004). This is an approximate depth, where the light level decreases below 1% of the surface irradiation. Therefore, when substituting a ratio of 100:1 of $I_{z1}$:$I_{z2}$ into Equation 2.2, there is a fixed inverse proportionality between euphotic depth, $Z_{eu}$, and the vertical attenuation coefficient, $K_d$ as shown in Equation 2.3:

$$Z_{eu} \approx \frac{4.6}{K_d} \tag{2.3}$$

### 2.5.2 Temperature and salinity

Variation in temperature and salinity of influent water from the sea and/or river compared with that of an estuary defines the resultant density of the water mass.

A stratified water column generally occurs when less dense water is positioned above more dense water. This creates stratification where temperature decreases and salinity increases with depth.

Water column temperatures of TOCEs are significantly affected by the seasonal and regional climate. Since the main input of water in TOCEs is through river inflow, the water temperature is primarily dependent on river water temperatures and flows. However, during open mouth phases, cooler marine waters are introduced into the estuary resulting in vertical and horizontal stratifications in temperature and salinity. Turbulence due to wind action and new river/sea water inflow can however, homogenise the whole water column in a short period of time (Gama *et al*., 2005). Temperature is one of the most important driving factors responsible for the seasonal cycle in ecosystems and also has a direct impact on the phytoplankton – zooplankton system (Scheffer, 2004).

During closed mouth conditions, TOCEs generally exhibit lower salinities, a more stratified water column and dominance by estuarine or freshwater taxa (Perissinotto *et al*., 2003). Where freshwater flows into an estuary system, during closed mouth conditions, reduce salinity levels, horizontal and vertical salinity gradients tend to preside during open mouth conditions (Gama *et al*., 2005). The different levels of salinity stratification are presented in Table 2.3 (Livingston, 2003).

**Table 2.3 Various levels of salinity stratification (Livingston, 2003).**

<table>
<tr><td>Highly stratified</td><td rowspan="4">If difference in top and bottom salinity is</td><td>>= 10 ppt</td></tr>
<tr><td>Partially stratified, strong</td><td>>= 5 ppt and < 10 ppt</td></tr>
<tr><td>Partially stratified, weak</td><td>>= 2 ppt and < 5 ppt</td></tr>
<tr><td>Vertically homogoneous</td><td>< 2 ppt</td></tr>
</table>

### 2.5.3 Dissolved oxygen

Dissolved oxygen is a fundamental requirement of fish and other aquatic biota. Most estuarine biota are capable of tolerating short exposures to low dissolved oxygen (DO) concentrations without apparent adverse effects. However, prolonged exposures to moderate hypoxia (DO less than 5 mg/L) may have severe consequences. In order to avoid these conditions, increased predation and decreased movement into certain feeding areas by some aquatic animals may occur. Severe hypoxia, on the other hand (DO below 2 mg/L) will kill most aquatic animals (EPA, 1998).

Oxygen is required for the decomposition of dead algae. This in turn decreases the quantity of dissolved oxygen available for aquatic life. As temperature increases, the rate of decomposition increases thus reducing the concentration of available DO. Vertical stratification in estuarine waters characterized by warmer, fresher water situated above colder, saltier water, during spring and summer periods can prevent reoxygenation of bottom waters. Salinity and temperature play an important role in the amount of oxygen available to estuarine organisms. The dissolved oxygen content increases with a decrease in water temperature and salinity (EPA, 1998).

Reoxygenation of water usually occurs through turbulence due to wind and increased flows (Peavy *et al.*, 1985). Oxygen due to photosynthetic byproducts may also contribute to the reoxygenation of water bodies (EPA, 1998).

### 2.5.4 Nutrients

Nutrient over-enrichment, followed by rising phytoplankton concentrations, has become a global environmental threat (Thomas *et al.*, 2005). Walmsley (2000) defines a nutrient as a "chemical compound or element that can be used directly by plant cells (algae and aquatic macrophytes) for growth. In the context of eutrophication, nutrients are mostly inorganic elements that are assimilated by plants and, in conjunction with the process of photosynthesis, are utilized to produce and accumulate organic material in aquatic ecosystems."

The nutrient elements for the growth and reproduction of micro-organisms, plants and animals are nitrogen and phosphorous. Trace elements can also be classified as nutrients, however those most needed for biological growth are carbon, nitrogen and phosphorous. Since nitrogen and phosphorous are generally labelled as the limiting nutrients, they generally receive the most attention (Metcalf and Eddy, 2003; Peavy *et al.*, 1985).

**Nutrient limitation**

Howarth (1988) stated that estuaries receive a significantly larger quantity of nutrient inputs than any other type of ecosystem. Estuaries receive nutrient inputs per unit area of up to 1000 times the quantity applied to heavily fertilized agricultural land. This poses a serious threat on the potential of an estuary to reach an eutrophic state. Phytoplankton requires nutrients in a ratio of 16 moles nitrogen to 1 mole phosphorus. This ratio of 16:1 is known as the Redfield ratio.

Nutrient limitation of primary production in aquatic ecosystems has generated much discussion. This is due to the varying definitions/understandings of *nutrient limitation* and the generalizations of all aquatic ecosystems. Since this investigation deals mainly with eutrophication, the limitation of net primary production was selected.

Three main factors determine whether an aquatic system is nitrogen or phosphorous limited (Howarth, 1988):

1. The ratio of nitrogen to phosphorous
2. The preferential loss, recycling or adsorption of one of these nutrients in the ecosystem and
3. The extent of nitrogen fixation.

These three factors indicate that estuaries and other coastal marine ecosystems tend to be nitrogen-limited when compared to lakes. The discharge of sewage into estuaries often contributes to a low N:P ratio, which is often below the Redfield ratio of 16:1. Sewage is typically characteristic of very low N:P ratios (Howarth, 1988).

**Nutrient dynamics**

Nitrogen and phosphorous concentrations in the water column are useful measures of the potential for nuisance plant growths. Plants are also able to derive nutrients from other sources, such as the sediment and suspended particulate matter. Therefore, the concentration of nutrients present in the water column will not necessarily be an accurate representation of the nutrients available for plant growth (Wepener *et al.*, 2006). Ecosystems are complex and therefore require that the whole ecosystem dynamics be considered. This includes identifying the major sources and sinks of nutrients. Grobelaar (1992) disputes the use of over-simplified models of nutrient loadings for estuaries and other ecosystems since factors such as high turbidity, poor light availability and the hydrodynamics can influence these effects.

Various mechanistic and empirical models have been formulated in order to model excessive nutrient loading in aquatic environments. Advances in scientific understanding, methods of analysis and cross-media modelling have increased model size and complexity (Reckhow and Chapra, 1999). Most nutrient loading models have large data requirements and limited observational databases are generally insufficient to support them. It is important that both theory and observation be incorporated into predictive models thereby providing necessary improvements (Reckhow and Chapra, 1999, Stow *et al.*, 2003). Examples include the AQUATOX model (Release 2, 2004) developed by the United States Office of Water Environmental Protection Agency (EPA), the nutrient budgeting procedure developed by the United Nations Environmental Programme (UNEP, Smith and Hawaii, 2000) project: Land-Ocean Interactions in the Coastal Zone (LOICZ) and other models including those developed by O'Donohue and Dennison (1997), Schladow and Hamilton (1997a, b) and Roelke *et al.* (1999) to mention a few. These models were not suitable for the present application due to the complexity involved and the large data requirements.

**Inorganic nitrogen**

Nitrogen is either dissolved or particulate bound and can occur as inorganic or organic nitrogen (Table 2.4). Elevated levels of inorganic nitrogen tend to alter the trophic status of rivers, lakes and estuaries and are accompanied by the growth of algae and other aquatic plants. Therefore, they are often used as indicators of the effect of inorganic nitrogen on aquatic ecosystems. Site-specific conditions, specifically the availability of phosphorus, are able to modify the influence of inorganic nitrogen on eutrophication. Inorganic nitrogen concentrations below 0.5 mg/L are generally low enough to limit eutrophication (DWAF, 1998). DWAF (1998) states that eutrophic conditions are often related to inorganic nitrogen concentrations of 2.5 to 10 mg/L and above (provided that other environmental conditions are favourable).

Inter-conversions of the different forms of inorganic nitrogen are part of the nitrogen cycle and are important in the determination of N availability in surface waters (Walmsley, 2000). Ammonification, nitrification, denitrification, and the active uptake of nitrogen compounds are regulated by water temperature, oxygen availability and pH (DWAF, 1998).

**Table 2.4 Nitrogen forms present in surface and wastewaters (adapted from Vollenweider, 1970)**

| TOTAL NITROGEN | | | | |
|---|---|---|---|---|
| **Dissolved Nitrogen** | | **Nitrogen in Suspension** | | **Gaseous Nitrogen** |
| Inorganic compounds eg. $NH_4$, $NO_3$, $NO_2$ | Organic Compounds eg. amino acids, polypeptides&peptides, dissolved albumin etc | Organisms | Organic detritus and/or inorganic&organic compounds adsorbed on particles | $N_2$, $N_2O$, NO |
| ← Total N content of unfiltered water → | | | | |
| ← Total N in filtrate → | | | | |
| ← DIN → | | | | |

Dissolved inorganic nitrogen (DIN, mg/L) is the sum of ammonia ($NH_4^+$), nitrite ($NO_2^-$) and nitrate ($NO_3^-$) concentrations. In the aquatic environment, nitrate is more stable and generally far more abundant than nitrite. However, all these forms are generally measured and considered together due to their co-occurrence and rapid inter-conversion (DWAF, 1998).

Perissinotto (1995) explains that nitrate is responsible for the rate of new production whereas ammonia and urea are responsible for fuelling regenerated production. The order of preferential uptake by all phytoplankton size classes is ammonia, urea, nitrite and then nitrate.

**Phosphorous**

Phosphorous is essential for the growth of algae and all other organisms and is therefore actively utilized. This is highlighted by the scarcity of high P concentrations in unpolluted surface waters. Elevated phosphorous levels may however occur due to point-source discharges such as domestic and industrial wastewater effluents and also diffuse/non-point sources such as stormwater runoff from urban and agricultural catchments (DWAF, 1998). Municipal wastewaters may contain phosphorous concentrations from 4 to 16 mg/L. These levels can have a major impact in the trophic status of a water body. DWAF (1998) states that inorganic phosphorous levels ranging from 25 to 250 µg/L are generally associated with eutrophic conditions.

As illustrated in Table 2.5, phosphorous is either dissolved or particulate bound and occurs as inorganic orthophosphates, polyphosphates and organic phosphates (Walmsley, 2000). Orthophosphates are the only form of dissolved inorganic phosphorus (DIP) directly utilizable by aquatic biota. The phosphorus cycle is characterised by the exchange of phosphorous between the water column and sediments. This is influenced by physical, chemical and biological factors such as water pH, sorption processes and the activities of living organisms (DWAF, 1998).

**Table 2.5 Forms of phosphorous present in surface water and wastewater (adapted from Vollenweider, 1970).**

| TOTAL PHOSPHOROUS | | | | |
|---|---|---|---|---|
| **Dissolved Phosphorous** | | **Phosphorous in Suspension** | | |
| Orthophosphate $PO_4$ | As organic colloids and/or combined with an adsorptive colloid | As mineral particles and/or adsorbed on inorganic complexes | Organisms | Adsorbed on detritus and/or present in organic compounds |

← Total P content of unfiltered water →

← Total P in filtrate →

← DIP →

## 2.6 Biotic Component

Eutrophication and its effects on the aquatic environment are reflected by pressures placed on the natural biodiversity (Walmsley, 2000). The natural assimilative capacity of a coastal system relative to nutrient loadings is an important feature of any estuary. Changes in the contribution of nitrogen and phosphorous compounds can cause transformations in the composition of species (Livingston, 2001). Therefore, it is an important consideration in the evaluation of the conversion from natural eutrophication processes to hypereutrophication (Livingston, 2001). The suggested deterioration of associated biodiversity caused by the loss in assimilative capacity is a major concern. The residence time of the estuary is of particular importance in terms of the utilization of nutrients. During short retention times (days) there is insufficient time for the phytoplankton to trap nutrients as they are washed out to sea during open mouth conditions (Adams & Bate, 1999).

The extreme complexity of phytoplankton response to changes in the abiotic environment of coastal systems makes the effects of cultural eutrophication inconceivable (Livingston, 2001). However, there have been extensive studies and models formulated on the response of phytoplankton assemblages to nutrient loadings (Scheffer, 2004; O'Donohue and Dennison, 1997).

### 2.6.1 Primary production and microalgae

Perissinotto (1995) summarises primary production (from an ecological perspective) as "the rate at which new biomass is added to a population or community and is expressed in terms of the change in weight or number per unit time." Primary production is generally measured as the total quantity of carbon assimilation by the system. It appears that TOCEs experience a period of biological rejuvenation subsequent to a natural breaching event, followed by a period of maximal productivity shortly after re-closure of the mouth (Anandraj *et al.*, 2006, Thomas *et al.*, 2005)

Plankton and benthos are two of the ecological levels that dominate the pelagic (upper layer) and benthic (seafloor) subsystems respectively. Both of these levels are typically subdivided into functional groups, namely phyto- (autotrophs) and zoo- (heterotrophs). Phytoplankton and benthic microalgae are important in terms of overall estuarine primary production, however they represent one of the most complex and least understood elements of coastal systems (Livingston, 2003). Various studies of South African TOCEs revealed a large discrepancy between microphytobenthos and phytoplankton biomass (Perissinotto *et al.*, 2004, Thomas *et al.*, 2005). Possible reasons for this include the prevailing conditions in these systems, including sediment nutrient supply and sinking due to favourable conditions.

Phytoplankton is estimated to comprise about 80 to 90 % of the net annual marine production (Perissinotto, 1995). The phytoplankton community is conventionally divided into different size classes, namely microphytoplankton (> 20 µm), nanophytoplankton (2–20 µm) and picophytoplankton (< 2 µm). Phytoplankton can be simply defined as a group of organisms that have chlorophyll-a as the main photosynthetic pigment. Therefore, its concentration can be used to provide an estimate of the photosynthetic capacity of phytoplankton as well as an index of phytoplankton biomass. This method is widely used to give an indication of phytoplankton communities in their trophic organisations and in an array of modelling efforts (Livingston, 2003). Since algae contain a substantial part of the total amount of phosphorous in the water column, in summer, the chlorophyll concentration is generally well correlated to the total phosphorous concentration in the water column (Scheffer, 2004). Vollenweider (1970) reported the variations found in chlorophyll-*a* for Japanese lakes with different trophic levels established by Sakamoto (1966) as summarised in Table 2.6.

**Table 2.6 Trophic levels and the corresponding chl-a concentrations (Sakamoto, 1966).**

| Trophic levels | mg Chl-*a* /m$^3$ |
|---|---|
| Eutrophic | 5 - 140 |
| Mesotrophic | 1 -15 |
| Oligotrophic | 0.3 – 2.5 |

In a study of the permanently open Sundays Estuary by Adams and Bate (1999), phytoplankton was found to require three spring tidal cycles, 42 days, to bloom and attain maximum biomass. Thomas *et al.* (2005) however, recorded bloom events in the Mdloti and Mhlanga Estuaries, with corresponding residence times of 10, and 9 to 11 days, respectively. This is an indication of the favourable conditions that exist within these estuaries. Declines in phytoplankton biomass following a bloom may be attributed to grazing by zooplankton, nutrient depletion, sedimentation or self-shading through cell abundance (Thomas *et al.*, 2005).

### 2.6.2 Regulation processes of algal biomass

Scheffer (2004) provides a concise description of the regulation processes of algal biomass. It is stated that when the algal population is low relative to the carrying capacity of the environment the relative growth rate is highest. The productivity is however limited by the quantity of reproducing algae. When the population approaches the carrying capacity, productivity decreases to zero due to competition.

**Losses due to sinking and flushing**

Algal cells are prone to the same physical processes as sediment particles. The cycle of sedimentation and resuspension carried out by suspended sediment particles also applies to algal cells. Where some algal species are able to regulate their buoyancy, most species suffer substantial losses due to sedimentation. These species therefore depend on turbulence for survival. In vegetated parts, algal biomass is usually lower, but the community is dominated by small species that have high growth rates and low settling velocities. Compared to nanophytoplankton, microphytoplankton have a greater tendency to sink as they are more dense. Settling and flushing are density independent processes. These processes affect slow growing species more and therefore offer a competitive advantage to fast growing species. These processes may result in a change from larger, slower growing species to smaller, faster growing species (Scheffer, 2004).

**Light limitation**

Due to the turbid and nutrient enriched nature of estuaries, light limitation has been found to control phytoplankton biomass (Cole and Cloern, 1987, O'Donohue and Dennison, 1997). Phytoplankton growth may increase to such a degree that algae limit their own growth by self-shading (Kirk, 1994). Self-shading due to cell abundance increases the turbidity of water and inhibits growth by reducing the light available to the whole phytoplankton population as well as benthic primary producers (Livingston, 2001, Alvera-Azcárate *et al*, 2003).

**Phytoplankton control by grazers**

Zooplankton is the most important consumer of phytoplankton in most freshwater systems, as algal biomass can be reduced by an order of magnitude due to zooplankton grazing (Scheffer, 2004). In a study conducted on the Mdloti and Mhlanga Estuaries (TOCEs) by Kibirige *et al.* (2005), it was found that zooplankton biomass was higher during closed mouth conditions when compared to open mouth conditions.

The impact of top-down control of phytoplankton biomass has been seen as the reason for the positive correlation found between phytoplankton chlorophyll *a* biomass and zooplankton abundance in various TOCEs (Kibirige and Perissinotto, 2003, Perissinotto *et al.*, 2003, Perissinotto *et al.*, 2004, Thomas *et al.*, 2005). Supporting this is that nanophytoplankton has been found to be the greatest contributor to phytoplankton biomass (Thomas et al., 2005, Perissinotto *et al.*, 2004). Nanophytoplankton is the preferential food source of most species of zooplankton in TOCEs (Kibirige and Perissinotto, 2003). The time scale for zooplankton to reduce phytoplankton biomass has not yet been quantified. However, Perissinotto *et al.* (2004) found that the process may take longer than three weeks following mouth closure.

## 2.7 Summary

This chapter includes a literature review of the physical, biological and chemical components of TOCEs. The hydrological functioning of a TOCE system is conceptualized in terms of a dynamic storage system with variable inputs and outputs. Breaching mechanisms are generally related to the water balance of the system whereas closing mechanisms are related to marine processes. Tidal exchange processes were not explicitly modeled for this study, but were investigated in order to understand what happens to the water level at mouth closure. The term "residence time" is defined as the period of time between breaching events that the water is stored in the estuary (Perissinotto *et al.*, 2004) and is related to the mouth state. Previous models relating flow rate and mouth state were investigated. The water balance model used by Londal (2005) was chosen for this investigation as it accounts for water levels which are a key parameter in the mouth dynamics.

Nutrient over-enrichment and consequent algal blooms have become a global environmental threat (Thomas *et al.*, 2005). Eutrophication of TOCEs can have adverse ecological, water quality and anthropogenic impacts. The abiotic environment is dependent on the hydrological functioning of TOCEs and in turn can have significant impacts on the biotic environment of such a system. Various existing models were investigated in order to model the nutrient dynamics of TOCEs, however these models have large and extensive data requirements and were therefore not suitable for the present application. The water balance model was therefore extended to incorporate the nutrient balance for eutrophication applications. Although the abiotic and biotic parameters are not included in the model, they are investigated in order to assist in interpreting the processes that underpin the changes in the eutrophication indicators.

# 3. METHODOLOGY

## 3.1 Introduction

Chapter three presents the characteristics of the case study, the Mhlanga Estuary, the model developed and the field work performed. Various models have been used to predict excessive nutrient loading and its effects in aquatic environments. Due to advances in scientific understanding models have increased in size and complexity, the disadvantage being large data requirements and limited observational data to support them. The basic approach was therefore to develop a simple coupled water and nutrient balance model incorporating the prediction of eutrophication. It is intended that this model be used as prediction tool that can be used at other similar estuaries. The various parameters included in the model are introduced.

Field measurements were collected from the case study to test the model. Low resolution (monthly) observational data was available for the case study; however this data was not helpful in investigating the eutrophication processes that occur. It was decided that chlorophyll-*a* and physico-chemical samples be taken weekly. These parameters were not used in the model, but were used to assist in interpreting eutrophication processes.

## 3.2 Case Study: Mhlanga Estuary

The Mhlanga Estuary is a perched temporarily open/closed estuary formed where the Mhlanga River meets the Indian Ocean (Plate 3.1). Harrison *et al.* (2000) classified the estuary as a medium sized, subtropical estuary with poor water quality, good fish community, and moderate aesthetic state. The poor water quality of the estuary highlights the need for correct management procedures, especially since major tourism beaches are located directly adjacent to the estuary. Poor water quality and its negative impacts can have serious repercussions on social, ecological and financial aspects of the region.

The Mhlanga Estuary is situated on the east coast of South Africa, in KwaZulu-Natal (as illustrated in Figure 3.1). The estuary is located on an exposed coastline where the median significant wave height is approximately 1.8 m (Rossouw, 1984). The dominant wave direction is from a southerly direction with an estimated 1400 $m^3$/day of sediment transported longshore in the same direction (Rossouw, 1984). A summary of the physical data for the Mhlanga Estuary, adapted from Stretch and Zietsman (2004), is provided in Table 3.1.

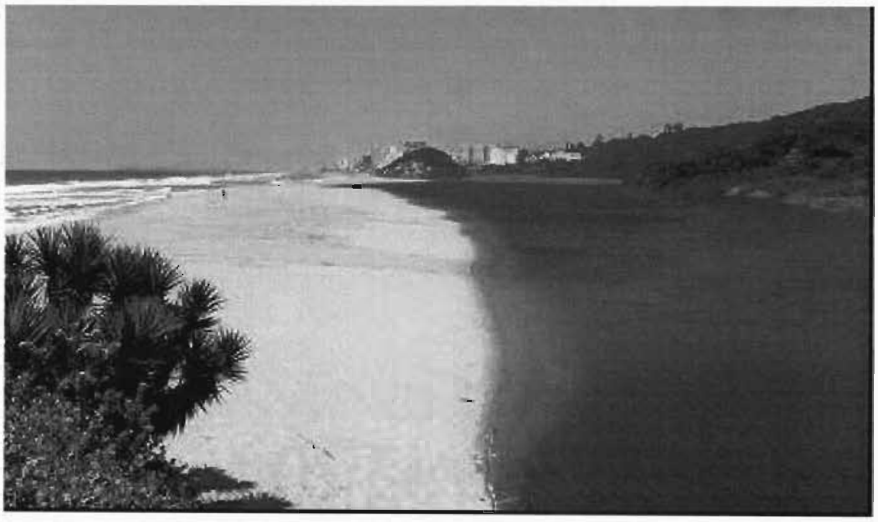

**Plate 3.1 The Mhlanga Estuary, situated 20 km North of Durban on the East Coast of South Africa.**

**Table 3.1 Physical characteristics of the Mhlanga Estuary**

| Characteristic | | Value |
|---|---|---|
| **Catchment** [1] | Catchment area | 80 $km^2$ |
| | MAP | 1000 mm |
| | MAE | 1210 mm |
| | Natural MAR | 0.4 $m^3/s$ |
| | Present annual inflows | 0.63$m^3/s$ |
| **Estuary** [2] | Estuary area | 70 ha |
| | Max water level (above MSL) | 3200 mm |
| | Min water level (above MSL) | 670 mm |
| | breaching height (above MSL) | 3000-3400 mm |
| | Live storage in estuary | 800000 $m^3$ |
| | % Closed (present) | 50% |
| | max seepage (through berm) | 0.25 $m^3/s$ |
| | Mouth closing time scale | 3-6 days |
| | Average inflow residence time | 15 days |
| **Sand Berm** [2] | Berm length | 500 m |
| | Berm width | 30-60 m |
| | Berm height (above MSL) | 3-4 m |
| | Sandbar permeability | 100-200 m/day |
| **Coastal Parameters** | Tidal range (neap) [2] | 0.5 m |
| | Tidal range (spring) [2] | 2 m |
| | Median significant wave height [4] | 1.8 m |
| | Typical beach slope [2] | 0.1-0.2 |
| | Average longshore transport [3] | 1400 $m^3/day$ |
| | Average crossshore transport [2] | 15-20 $m^3/day/m$ |
| | Tidal prism % volume [2] | 10-30 % |

[1] WR90
[2] Stretch and Zeitsman (2004)
[3] Schoones (2000)
[4] Rossouw (1984)

Flow regimes, water quality and mouth behaviour are sensitive to changes due to human activities such as abstractions, impoundments and discharges from wastewater treatment works (Perissinotto *et al.*, 2004). The Mhlanga Estuary is a good example of a TOCE that has become severely impacted by additional inputs of discharge from two wastewater treatment works (WWTW). The Phoenix and Mhlanga WWTWs are situated approximately 12 km and 2.5 km respectively upstream of the estuary (Archibald, 1995). The operators supplied daily average effluent discharge measurements from the WWTWs for this investigation. Measurements indicate that Phoenix WWTW discharges a flow of approximately 13 Ml/day and the Mhlanga WWTW 6.5 Ml/day. Stretch and Zietsman (2004) found that the actual flows of the Mhlanga WWTW vary from 1 to 20 Ml/day during the course of the day.

The Mhlanga Estuary generally breaches by means of the seepage breaching mechanism. However, when the inflows exceed outflows, failure occurs by overtopping of the berm (Perissinotto et al., 2004). Research done by Stretch and Zietsman (2004) found that the Mhlanga Estuary breaches, on average, every thirty to forty days during low flow conditions.

## 3.3 Water Balance Model of the Mhlanga Estuary

By conceptualising the Mhlanga Estuary as a dynamic storage system with variable inputs and outputs, it is possible to simplify the hydrological functioning of the system (see Figure 3.2). A daily time step water balance model was used to represent the hydrodynamics of the system. The water balance model estimates the change in storage by identifying and quantifying the inputs and outputs of the system as illustrated in Figure 3.1 and extends the model represented by Londal (2005). By estimating the change in storage and residence time, it is possible to predict the water level and ultimately the mouth state. The various components of the water balance model are discussed in the following sections.

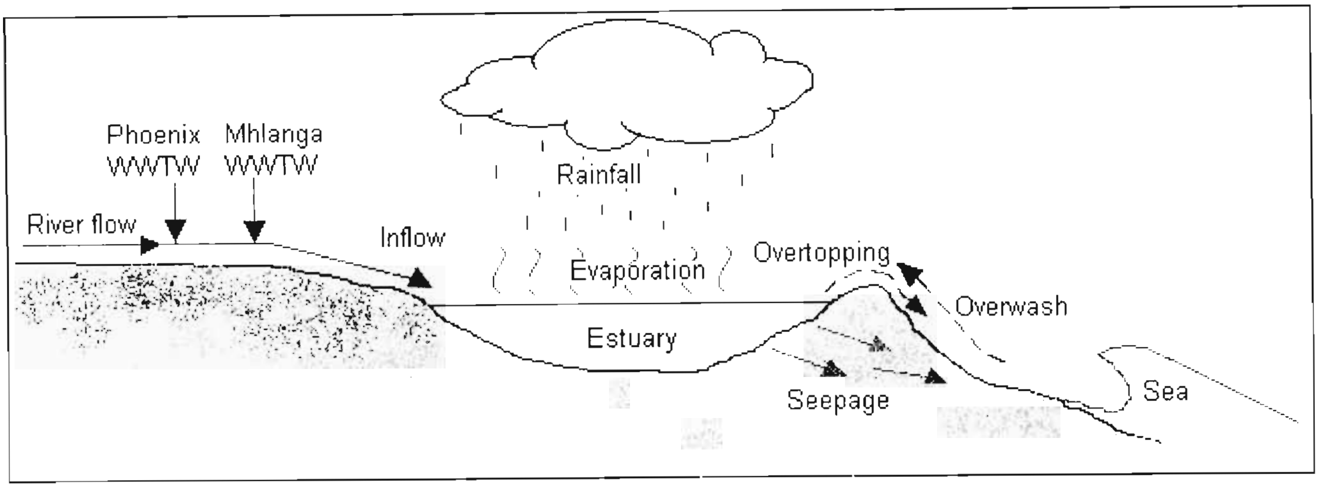


**Figure 3.1 Conceptual water balance of the Mhlanga Estuary illustrating the various inputs and outputs of the system.**

### 3.3.1 Inflows

During closed mouth conditions, inflows into the Mhlanga Estuary comprise river inflow, WWTW discharge, rainfall, overtopping/overwash and groundwater inflow. Tidal activity provides additional inflow during open mouth conditions. Groundwater inflow was assumed negligible, although there is currently no data to verify this.

#### River flow

There was no stream gauge present for the Mhlanga catchment. Daily inflows were therefore based on areal scaling of the U3H005 stream-gauge and estimates of WWTW discharges (Stretch and Zietsman, 2004). The stream-gauge monitors the daily outflow from the Hazelmere Dam, which is fed by catchment U30A (377 $km^2$), as illustrated in Figure 3.2. Mdloti estuary flows were estimated by scaling the stream-gauge data in proportion to the increased catchment area (section of catchment U30B, 484 $km^2$). Area scaling was then used to provide a rough estimate of flow into the Mhlanga Estuary, as fed by the remaining section of catchment U30B (80 $km^2$). The catchments U30A and U30B are adjacent to each other.

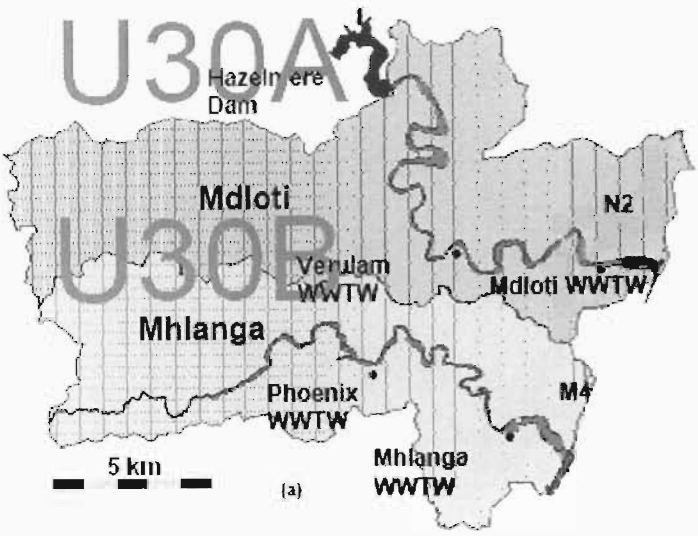


**Figure 3.2 Map of the Mdloti and Mhlanga estuary catchments. Note that the section of the Mdloti catchment upstream of the Hazelmere Dam is not shown. Scales are approximate indications.**

Field measurements performed by Stretch and Zietsman (2004) found the relationship between the instantaneous flows into the Mdloti and Mhlanga Estuaries to be described approximately by a linear regression equation with an $R^2 = 0.95$

$$Inflow_{Mhlanga} = 0.369 \times Inflow_{Mdloti} + Inflow_{WWTW} \qquad [m^3/s] \qquad (3.1)$$

However, area scaling of the gauged daily flows from U3H005 suggested much lower flows than were obtained during the field study. Although this issue remains unresolved and requires further attention, it remains outside the scope of this investigation. The accuracy of the flows is however, not critical, since (1) the aim of this investigation was to explore the nutrient dynamics and (2) observed mouth state data were used in the water balance model in order to reduce the dependency on questionable inflows .

Daily discharge flows into the Mhlanga River from the Phoenix and Mhlanga WWTWs were provided by eThekwini Municipality, Wastewater Department. The data was added to daily river flows to obtain the total inflow into the estuary.

**Rainfall**

Rainfall data from the Tongaat (Maidstone) and Mount Edgecombe (SASRI Research Site) weather stations were collected from the South African Sugar Research Institute (SASRI) website. An average of the rainfall recorded at the two weather stations was used to approximate the total precipitation falling in the Mhlanga area.

Zietsman (2003) found that rainfall events in the Mhlanga Estuary catchment area, of approximately 25 mm per day are generally associated with breaching events. The direct rainfall into the estuary was not considered as a separate component of the model as it is assumed to be included in the river inflow.

**Overwash and overtopping**

Given that overtopping or overwash generally results in breaching of the sandberm, it was assumed that when overtopping or overwash occurred, the estuary had breached or was in the initial process of breaching. Therefore, it was assumed unnecessary to quantify these inputs.

### 3.3.2 Outflows

During closed mouth conditions, the main water loss from the Mhlanga Estuary is by seepage through the berm. This is due to the perched nature of the estuary. Other losses such as evaporation are generally negligible in the case of small South African estuaries because of the small surface area of the estuary. Seepage rates increase with the rise in water level due to the increase in hydraulic head. Therefore in an estuary, seepage was simulated by assuming a single power law relationship where k = 3 (Londal, 2005). When the water level of the estuary remains constant and equilibrium is reached, the inflows equate the outflows. Therefore, inflow rates equate losses due to seepage (Perissinotto *et al.*, 2004). Research carried out by Zietsman (2003) at the Mhlanga Estuary suggested a maximum seepage rate of 0.23 $m^3/s$ when the estuary is fill.

**Seepage**

Seepage is a function of the hydraulic gradient, *i,* which is proportional to the head difference between the sea level and the water level of an estuary, $\Delta h$. Figure 3.3 shows an illustration of seepage through the sand berm of an estuary.

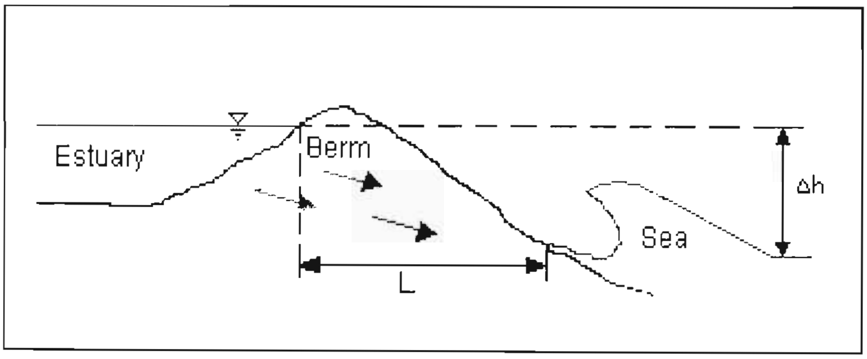


**Figure 3.3 Schematic of seepage occurring through the estuary berm.**

Seepage is estimated by multiplying the seepage area, *A*, by the hydraulic gradient, *i*, and the hydraulic conductivity, *hc*. The surface area is defined as the product of the length of the sand berm, *B*, and the difference between the present water level and the minimum water level, $\Delta H$. The hydraulic gradient is the ratio of $\Delta H$ to the length of flow over which the head difference occurs, *L*.

Whence

$$Seepage = k \cdot i \cdot A = k \cdot \frac{\Delta H}{L}(\Delta H \cdot B) \tag{3.2}$$

Assuming that $B$, $L$, and $k$ remain constant, the rate of loss due to seepage increases with increasing water levels (Londal, 2005). Therefore, the daily seepage rate is estimated by establishing a relationship between the daily water level related to the maximum water level and maximum seepage rate as shown in Equation 3.2 (Londal, 2005). Water levels are measured relative to mean sea level (MSL).

$$Seepage(t) = Seepage_{\max} \times \left( \frac{\Delta H(t)}{\Delta H_{\max}} \right)^2 \tag{3.3}$$

where $Seepage_{max}$ = the maximum seepage rate [$m^3/s$]

$\Delta H$ = the actual water level [mm]

$\Delta H_{max}$ = the maximum water level [mm]

$m$ = coefficient and is equal to $k-1$.

It is only possible for the water level to increase to the height at which breaching occurs, if the inflows exceed the maximum outflows. If the inflows are less than the maximum outflows, the water level will reach equilibrium below the breaching height. Therefore, a *critical flow rate* can be defined as equal to the maximum seepage outflow, below which breaching will not occur (Zietsman, 2003). However, inflows larger than the critical flow rate will not immediately result in a breaching event due to the influence of residence time.

**Evaporation**

The average evaporation loss, approximately 0.02 $m^3/s$, was estimated from the mean annual evaporation (MAE), 1210 mm and the maximum surface area, 70 ha (see Table 3.1). This is about ten times smaller than the maximum seepage and is therefore not relevant.

### 3.3.3 Storage

The morphology of the estuary has not been surveyed. Therefore, a geometric model was used to estimate the storage capacity of the estuary. A triangular prism, as illustrated in Figure 3.4, was chosen in order to allow for sedimentation and dead storage in the lower reaches of the estuary (Zietsman, 2003).

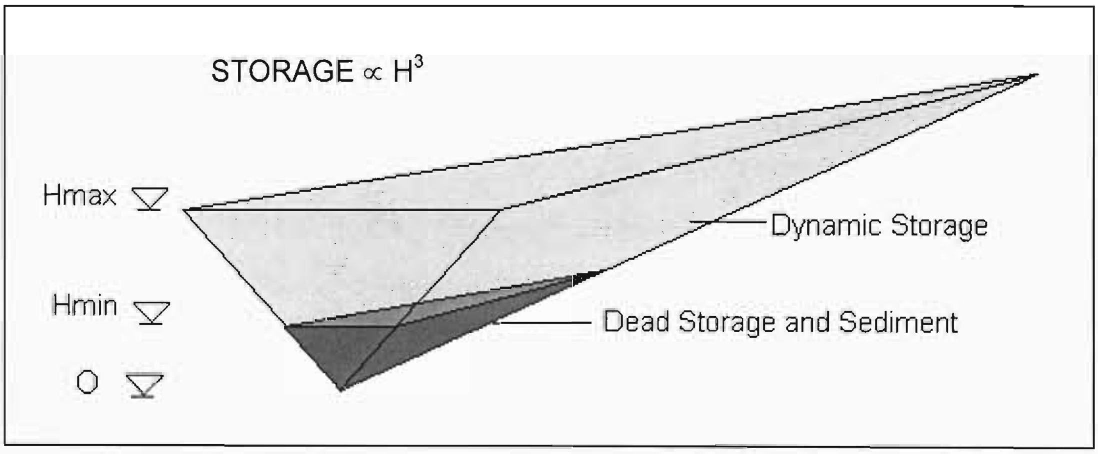


**Figure 3.4 Schematic of storage model used to represent the storage capacity of the Mhlanga Estuary (adapted from Zietsman, 2003).**

Storage is estimated for the water balance as the difference between the outflows and the inflows. During closed mouth conditions, water is accumulated and the storage is estimated as the lesser of the maximum storage or the daily resultant increase added to the storage of the previous day.

The maximum storage of the system (at breaching water levels) was therefore estimated by Stretch and Zietsman (2004) as 800 000 $m^3$.

### 3.3.4 Water levels

When the inflows exceed the outflows, the water levels rise until the maximum breaching level (maximum water level) is reached and failure of the sand berm occurs. Water level monitoring performed by Stretch and Zietsman (2004) at the Mhlanga Estuary indicated that breaching occurred at water levels of approximately 3000 mm above mean sea level. Continuous water level monitoring was required to provide a record of the changes in mouth state and to give an indication of the volume of the estuary.

**Simulated water level**

The water level was simulated by assuming a power law relationship such that $H \propto S^{1/3}$ (Londal, 2005). The daily time step water level was therefore given by:

$$\frac{(H - H_{\min})}{(H_{\max} - H_{\min})} = \left(\frac{S}{S_{\max}}\right)^{\frac{1}{3}} \tag{3.4}$$

When the predicted H exceeded $H_{max}$, a breach was assumed to occur and the water levels were reset to $H_{min}$ for the open mouth phase.

### 3.3.5 Filling time and time for mouth closure

As noted in section 2.2., the residence time $T_R$ is defined in the present context as the time that the estuary mouth has been closed. The filling time, $T_Q$, is the time required to fill an estuary to its breaching level, ignoring losses/outflows. $T_Q$ is therefore estimated as (Londal, 2005):

$$T_Q = \frac{Storage_{\max}}{Daily\_Inflow} \tag{3.5}$$

The *Daily_Inflow* is the total flow into the estuary measured in $m^3/s$,. The time required for mouth closure is defined as the time taken from when the estuary breaches until the berm is sufficiently rebuilt and the outlet closed. The number of days until closure after a breaching event is denoted $T_{close}$ (Londal, 2005).

The time it takes for the estuary berm to build, $T_{close}$, is a function of wave action and long-shore/cross-shore sediment transport. If $T_Q > T_{close}$, then the time required to fill the estuary to the maximum storage, given the inflow rate, is greater than the time it takes for the berm to rebuild. If $T_Q < T_{close}$, then the time required to fill the estuary to its maximum storage, given the daily inflow rate, is less than the time it takes for the berm to rebuild itself. If the latter occurs, the mouth is assumed to remain open. The time for closure increases by the number of days that the filling time $T_Q$ is less than $T_{close}$ (Londal, 2005).

### 3.3.6 Mouth State

The Mhlanga Estuary generally breaches as a result of the seepage breaching mechanism, but may also experience overtopping, especially when it rains. Therefore, the maximum water level is the level at which breaching occurs. The mouth state was assumed to be open under the three conditions summarized in

Table 3.2. If none of these conditions were true, the mouth state was assumed closed (Londal, 2005).

**Table 3.2 Conditions under which the mouth state would remain open.**

| Condition | Mouth State is open if: |
|---|---|
| 1 | The water level > the maximum water level |
| 2 | The mouth state of the previous day = open and $T_Q < T_{close}$ |
| 3 | The mouth state of the previous day = open and number of days to closure > 1 day |

Observed mouth state data provided by KZN Wildlife, was initially used as it makes the model less dependent on questionable inflow data. In order to compare the observed mouth state data with the simulated mouth state data and observe how the results would be affected, simulations were run using both sets of data.

## 3.4 Eutrophication Prediction Model for the Mhlanga Estuary

The eutrophication of estuaries is influenced by the nutrient input coupled with residence time. Physico-chemical parameters and chlorophyll-*a* concentrations, although not used in the model, are important in analysing eutrophic conditions. The aim was to predict eutrophic conditions and use the result of the field study to find a relationship.

### 3.4.1 Simulated nutrient concentrations

Mass balance principles and the wastewater effluent data were used to estimate the daily nutrient input into the Mhlanga Estuary, assuming conserved nutrient concentrations:

$$C_D = \frac{C_S Q_S + C_W Q_W}{Q_S + Q_W} \quad \text{[mg/L]} \tag{3.6}$$

| | | | |
|---|---|---|---|
| where $C_D$ | = | diluted nutrient concentration | [mg/L] |
| $C_S$ | = | river nutrient concentration | [mg/L] |
| $C_W$ | = | wastewater nutrient concentration | [mg/L] |
| $Q_S$ | = | river discharge | [$m^3$/s] |
| $Q_W$ | = | wastewater discharge | [$m^3$/s]. |

The estimated volume (live storage plus dead storage) of the estuary for the water balance, was then used to estimate diluted nutrient concentrations in the estuary per day. During closed mouth conditions, the potential accumulation of nutrients in the estuary could thus be estimated. These concentration levels were later compared to the field measurements. Note however that the nutrient balance did not account for losses due to biological uptake.

### 3.4.2 Grey water index

The chemistry of estuaries is dependent on a combination of hydrological, physico-chemical and biological interactions. Therefore, modelling the full nutrient balance involves interrelated systems to account for sources and sinks that are complex and difficult to validate. This was beyond the scope of this investigation. Therefore a simpler approach was selected. A grey water index (GWI) was developed in order to establish an indicator with which to predict algal blooms and nutrient concentrations. WWTW discharge is characterised by high concentrations of nutrients, and is denoted *black water*. It is assumed that the percentage of black water present in the estuary is significant due to the potential for algal blooms to occur. The natural (uncontaminated) river inflow is defined as *white water*. The river flow entering the estuary comprises a mixture of both white and black water and is therefore referred to as *grey water*.

The GWI considers the ratio of black water [$m^3$] to grey water [$m^3$] present in the estuary defined by

$$GWI = \frac{black\_water}{grey\_water} \ [\%] \qquad (3.7)$$

The volume of black water in the estuary is the quantity of black water entering the estuary per day in addition to that already present. The volume of grey water present in the estuary at any time is estimated as the dead storage in addition to the live storage (grey water) for that day. The dead storage is defined as the quantity of water remaining in the estuary after breaching of the mouth.

If the nutrient concentrations in the black water are conserved and the source concentrations are known, then the GWI can be used to give an indication of nutrient concentrations in the estuary.

A high percentage of black water present in the estuary, together with a long residence time can create the ideal environment for algal blooms to occur. Therefore, it is necessary to consider the residence time in connection with the GWI in order to be able to predict possible algal blooms.

If the time required for an algal bloom to occur, $T_B$, is greater than the residence time, $T_R$, the possibility of a bloom occurring is unlikely. However, if $T_B$ is less than $T_R$, the possibility increases. Therefore, by considering $T_R/T_B$ for each day that the estuary remains closed in conjunction with the GWI, it is possible to predict if a bloom may occur.

A *eutrophication index, $E_i$,* was defined in order to estimate the combined effects of the GWI and $T_R/T_B$ in the potential for an algal bloom to occur. Hence,

$$E_i = GWI \times T_R/T_B \qquad (3.8)$$

For blooms to occur, the residence time is expected to be greater than the bloom time $T_R/T_B \geq 1$, and the nutrient loadings greater than some critical value, i.e. $GWI \geq GWI_{crit}$. Therefore, $E_i$ would be greater than or equal to $GWI_{crit}$. Note that by truncating the $T_R/T_B$ ratio to a maximum value of unity implies that the $E_i$ has a range of zero to one. Therefore, $E_{i\,crit}$ would be equal to $GWI_{crit}$.

Alternatively, another method was used. This method defines a binary indicator of the potential for eutrophication to occur, $E_p$. The method was based on the following statement:

IF $T_R > T_B$ AND $GWI > GWI_{critical}$ THEN $E_p = 1$, OTHERWISE $E_p = 0$.

If $E_p$ equals one, an algal bloom is likely to occur and if $E_p$ equals zero, the potential for eutrophic conditions and algal blooms to occur is small.

### 3.5 Implementation of the Model

Spreadsheets were used to record, manipulate and analyse data. Data tables included the physical case study characteristics, rainfall data, mouth state data, WWTW discharge data (nutrient concentrations and flows) and river inflow data. Results obtained from sampling events were also tabulated. The various components of the water balance model and eutrophication model were then implemented. Appendix A presents the spreadsheet used for the water balance and eutrophication models as well as flowcharts for the water balance model.

The sensitivity of the water balance model to key parameters was tested by Londal (2005). These included the maximum storage, maximum seepage through the berm, maximum evaporation and time for mouth closure. Londal (2005) found that the model was most sensitive to changes in the maximum storage.

## 3.6 Field Sampling

Stretch and Zietsman (2004) found that the Mhlanga Estuary breaches, on average, every thirty to forty days during low flow conditions. Field measurements were taken on a weekly basis, between 10h00 and 13h00, for a three month period, from July 2006 to September 2006. The eutrophication of estuaries is influenced by the nutrient input coupled with residence time. Physico-chemical parameters and chlorophyll-a concentrations are useful in interpreting the processes that underpin the changes in the eutrophication indicators. Monthly data of these parameters were available for the Mhlanga Estuary, however higher resolution data were required. Therefore, chlorophyll-a concentrations and other physico-chemical parameters were measured on a weekly basis over a three month period, from July 2006 until the end of September 2006.

### 3.6.1 Flows

Flow measurements were taken to verify the inferred inflow record and examine the inaccuracies that would be expected from the inflow simulations. Weekly flow measurements were taken upstream of the Mhlanga WWTW discharge point. The same position on the Mhlanga River was used for all measurements. A fairly straight, narrow section was selected in order to minimise interference due to the riverbanks, reeds and changes in depth. Numerical integration was used to estimate flow rates. A 10mm diameter propeller type velocimeter instrument was used to obtain flow measurements. Velocity measurements were taken in conjunction with the water depth at one-metre intervals across the width of the river. Where possible the propeller was set at approximately 0.6 times the water depth in order to obtain the average vertical velocity profile. This was limited due to the shallowness of the river (approximately 300mm). The flow rate was estimated as the summation of flows for each section calculated as the product of velocity and area. Measured cross-sections of the river can be found in Appendix B. Effluent flows from the Mhlanga WWTW were added to the estimated flowrate to obtain the total river flow into the estuary (thus including both WWTWs).

### 3.6.2 Water levels

Two methods of monitoring the water level were utilised for this investigation. The first method involved a miniature submersible water level logger coupled to a pressure transducer developed by Stretch and Zietsman (2004). Details concerning the development of the device are given in Zietsman (2003). The device was calibrated (see Appendix C) before it was placed in a perforated container that was fixed under the pile cap of the M4 bridge (see Plate 3.2). The data logger can store up to 1800 data points. Logging was set at twenty-minute intervals. The installation failed to provide any useful data due to a problem with the electronics. DWAF subsequently provided a continuous water level monitor at the site (see Plate 3.3). Measurements were obtained using a Mini Orpheus pressure sensor. The data were independently recorded by DWAF and included water levels recorded at 12-minute intervals starting from August 2005.

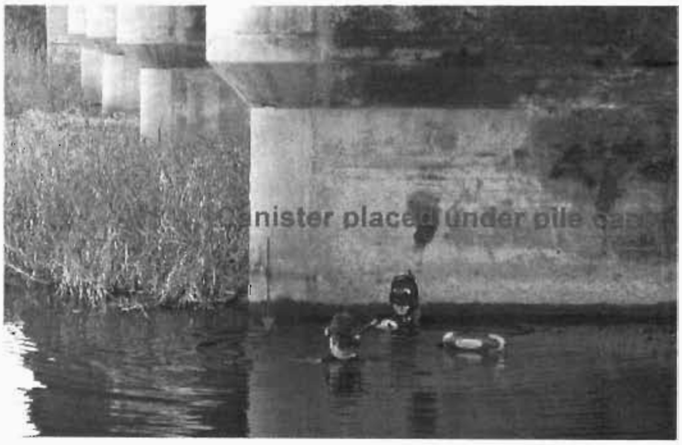

**Plate 3.2 Water level logger deployed by divers under the pile cap of the northern, seaward side of the M4 bridge.**

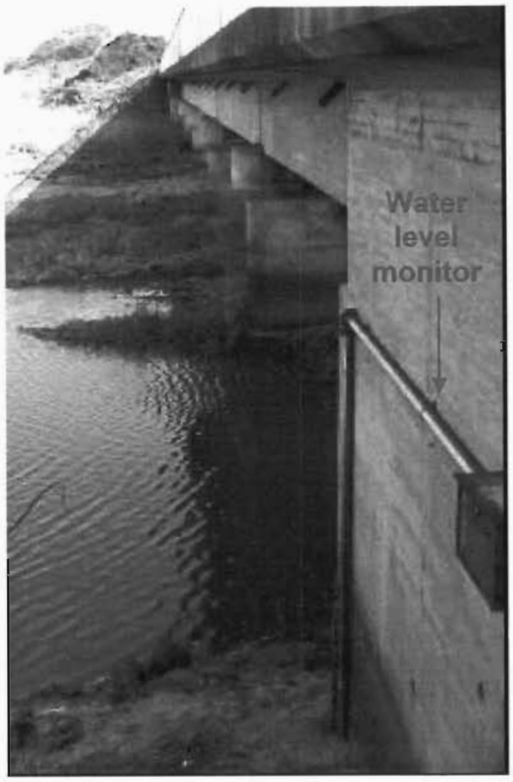


**Plate 3.3 Water level monitor installed by DWAF on the seaward side of the M4 bridge**

### 3.6.3 Physico-chemical parameters

Although the physico-chemical parameters were not included in the model, they assisted in interpreting eutrophication processes. Vertical profiles of turbidity, pH, salinity, depth, dissolved oxygen, temperature and conductivity were recorded using an YSI 6920 Water Logger. The sampling point selected was situated in the middle reaches of the estuary, on the landward side of the M4 bridge, as illustrated in Figure 3.5. Samples were taken between 11h00 and 13h00 for consistency.

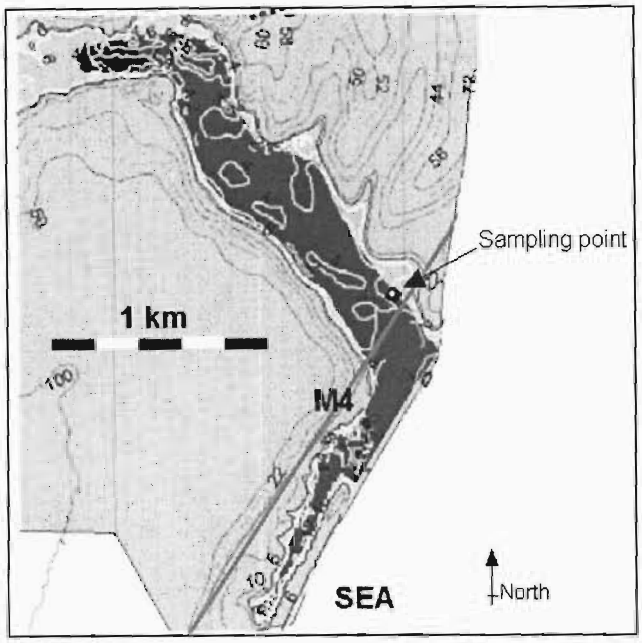


**Figure 3.5 Map of the Mhlanga estuary, showing the point at which water samples were taken.**

Irradiance (PAR, 400 to 700mm) was recorded using a LI-COR LI-189 underwater, spherical quantum sensor. Readings were taken 10 cm below the surface and 30 cm above the estuary floor. These readings were then used to estimate the diffusive attenuation coefficient, $K_d$ ($m^{-1}$) by using Equation 2.5

Water samples were collected, approximately 10 cm below the water surface and 30 cm above the bottom and placed into 250 ml acid washed polyethylene bottles for macronutrient analysis. Bottom samples were taken using a 1 litre pop bottle sampler and surface samples were collected manually. Dissolved inorganic nitrogen (DIN; ammonia, nitrate and nitrite) and dissolved inorganic phosphorous (DIP; orthophosphate) were analysed at the CSIR-Environmentek, Durban, in the Analytic Laboratory. This was done using the Automated Segmented Flow Analysis method to obtain ammonia and orthophosphate concentrations and the Automated Flow Injection Analysis method to obtain the nitrate and nitrite concentration.

Treated wastewater effluent data for both the Mhlanga and Phoenix WWTWs, including discharge flow rates, nutrient concentrations (ammonia, nitrates, nitrites and orthophosphates) and pH were supplied by the eThekwini Municipality, Wastewater Department. The data supplied comprised samples taken once to five times monthly.

### 3.6.4 Measurement procedures

Although phytoplankton biomass was not included in the model, it assisted in determining the bloom time for the case study and interpreting eutrophication processes. Chlorophyll-a concentration was used as an indication of phytoplankton biomass. It was measured by collecting water samples, 10 cm below the surface (manually) and 30 cm above the bottom (using a 1 litre pop-bottle sampler). The water samples were then placed into 500 ml acid pre-washed polyethylene bottles. Samples of 200 ml were then filtered through 45µm RGF glass-fibre filters. Chlorophyll-*a* (chl-*a*) extraction was done in 90 % acetone (10 ml) over 48 hours in polyethylene tubes at a low temperature (4°C). Chl-*a* was measured using the 10-AU Turner Designs fluorometer with the non-acidification, narrow band technique and calibrated using pure extracts of chl-*a* (Sigma Products). The fluorescence readings were then converted using Equation 3.10 in order to obtain an estimation of phytoplankton chl *a* biomass (Perissinotto, 2006).

$$[mgChl - a.m^{-3}] = fluorescence\_reading \times \frac{volume\_acetone[ml]}{water\_filtered[ml]} \qquad (3.9)$$

# 4. RESULTS AND DISCUSSION

## 4.1 Introduction

Chapter Four presents the results for physico-chemical and phytoplankton samples collected during the investigation period of three months (July 2006 to September 2006). The relationship between measured and estimated flows and rainfall are presented. Water level, wastewater discharge and observed mouth state data were collected and used in the analysis of results and the verification of model parameters. Four possible scenarios were defined and hydrological and eutrophication simulations run.

## 4.2 Flows

Measured instantaneous flows, simulated daily averaged flows and rainfall were plotted against time, as illustrated in Figure 4.1. Simulated flows remained fairly constant during the study period despite significant rainfall events on three occasions. The increase in flows due to rainfall was however observed in the measured data. Since the readings taken at gauge U3H005 are daily averages, short-term variations during rainfall events are averaged out and the data are therefore not an accurate representation of instantaneous flow rates. During the drier periods of the study, the measured flows ranged from 14 to 28 % more than the simulated daily average flows. This indicates that the conversion factor used to convert flows from the Mdloti to the Mhlanga Estuary requires further calibration. Since the aim of this investigation was to explore the nutrient dynamics, refinement was not pursued. As the inaccuracy in the simulated daily averaged flows would affect mouth state predictions, observed mouth state data was used thereby allowing the model to be less dependent on questionable inflows.

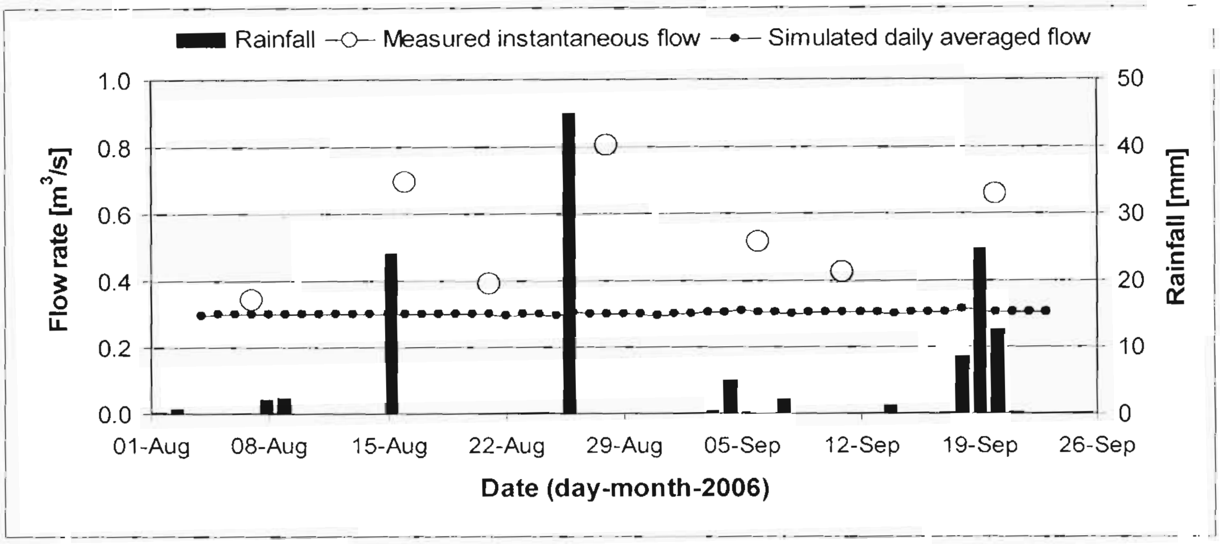


**Figure 4.1 Comparison of the measured and estimated inflows into the estuary and rainfall over the period of the study.**

An alternative approach would be to use the method outlined by Smakhtin (2004) by which continuous daily inflow time series to small ungauged TOCEs can be generated by the use of readily available rainfall records. These records would firstly have to be converted into some continuous function of rainfall to be used in the spatial interpolation algorithm (Smahktin, 2004). A key characteristic of the method would be river FDCs. Since the aim here was to explore the nutrient dynamics, refinement of this parameter was not pursued.

## 4.3 Water levels

The measured water levels show how the water level of the estuary gradually increases until the breaching water level is reached and breaching occurs. Tidal influence on the water level of the estuary during open mouth conditions is also evident. Figure 4.2 and Figure 4.3 provide an illustration of the temporal variation in water levels of the Mhlanga Estuary from August 2005 to December 2005 and from January 2006 to October 2006, respectively, as provided by DWAF.

The recorded data from DWAF indicates that the majority of minimum water levels generally ranged from 549 to 561 mm. The lowest water level reordered is 322 mm. The recorded data from DWAF also indicates that the average breaching level was 2690 mm with minimum and maximum breaching levels of 2384 and 3006 mm respectively. Stretch and Zietsman (2004) recorded a maximum water level of 3390 mm. Many of these breaching events were influenced by heavy rainfall events. When heavy rains increase the water level rapidly, premature breaching is likely to occur, especially if the sandbar has not been allowed sufficient time to completely rebuild itself.

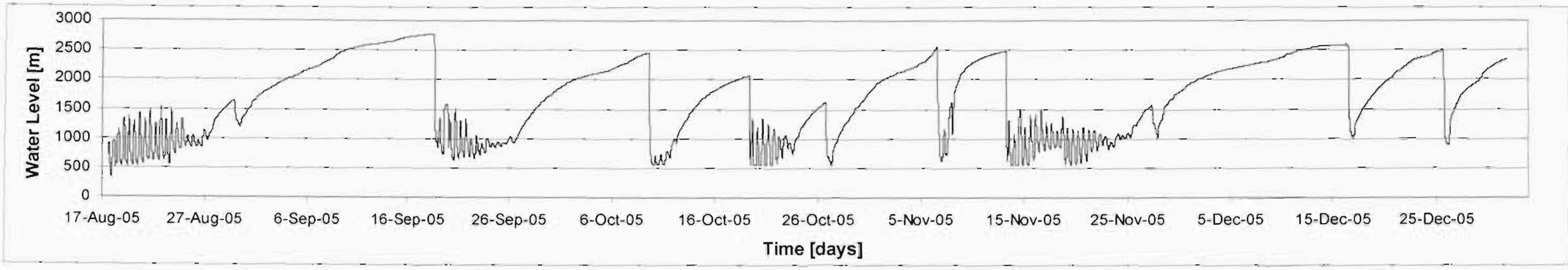


**Figure 4.2 Temporal variations in measured water levels (provided by DWAF) of the Mhlanga Estuary from August to December 2005.**

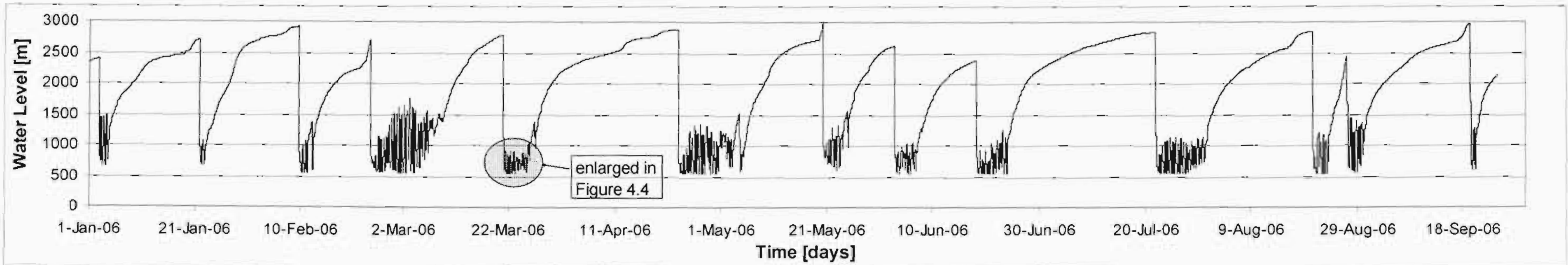


**Figure 4.3 Temporal variations in measured water levels (provided by DWAF) of the Mhlanga Estuary from January to September 2006.**

A section of Figure 4.3 has been enlarged (see Figure 4.4) to examine water level changes during open mouth conditions. Despite the perched nature of the Mhlanga Estuary, evidence of tidally driven water level fluctuations during the open mouth phase is present. From point A to B of Figure 4.4, there is a decrease in the difference between water levels at high and low tide. This illustrates the rebuilding process of the berm. The estuary mouth begins to close, but is followed by a subsequent partial breach at point C before the berm is completely rebuilt. The minimum water level recorded at point C is therefore the water level at closure from which the estuary begins to fill. Therefore, the minimum water level, $H_{min}$, used in the water balance model, was taken from this point. The average water level at this point was 930mm.

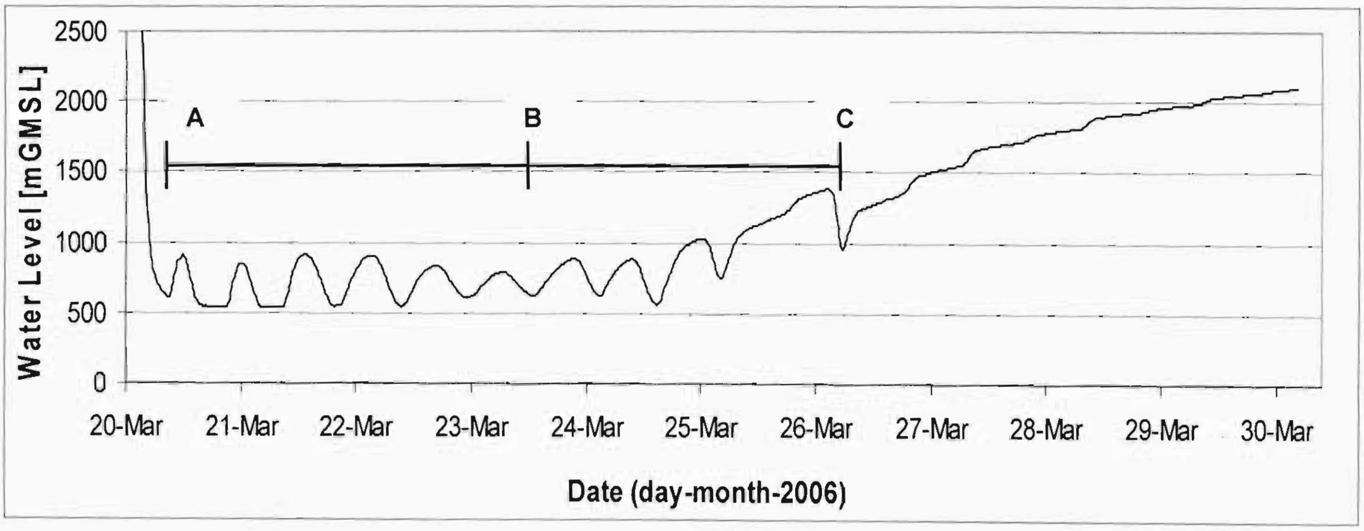


**Figure 4.4 Open mouth conditions of the Mhlanga Estuary, showing water levels and the corresponding tidal heights (mGMSL).**

## 4.4 Phytoplankton

Phytoplankton samples were taken in order to establish a bloom time for the model. Figure 4.5 illustrates the influence of mouth state and rainfall on phytoplankton biomass during the study period. The points are joined by dashes lines for clarity. The different closed mouth states are labelled 1 to 5 in Figure 4.5 and will be referred to as *phases*. Two exceptional phytoplankton blooms (greater than 100 mg chl-*a*.$m^{-3}$, after Adams and Bate, 1999) were recorded during the course of this study with a maximum reading of 375 mg chl-*a*.$m^{-3}$ (surface; 333 mg chl-*a*.$m^{-3}$ bottom) and the second highest reading of 280 mg chl-*a*.$m^{-3}$ (bottom; 32 mg chl-*a*.$m^{-3}$ surface). The maximum reading exceeded the maximum value recorded at the Mhlanga Estuary by Thomas *et al.* (2005), of 303 mg chl-*a*.$m^{-3}$, in October 2002. The majority of minimum values were recorded during open mouth conditions and ranged from 2.9 and 13.35 mg chl-*a*.$m^{-3}$. The decline in phytoplankton biomass during open mouth conditions is a result of flushing of the estuary by breach outflows and tidal activity.

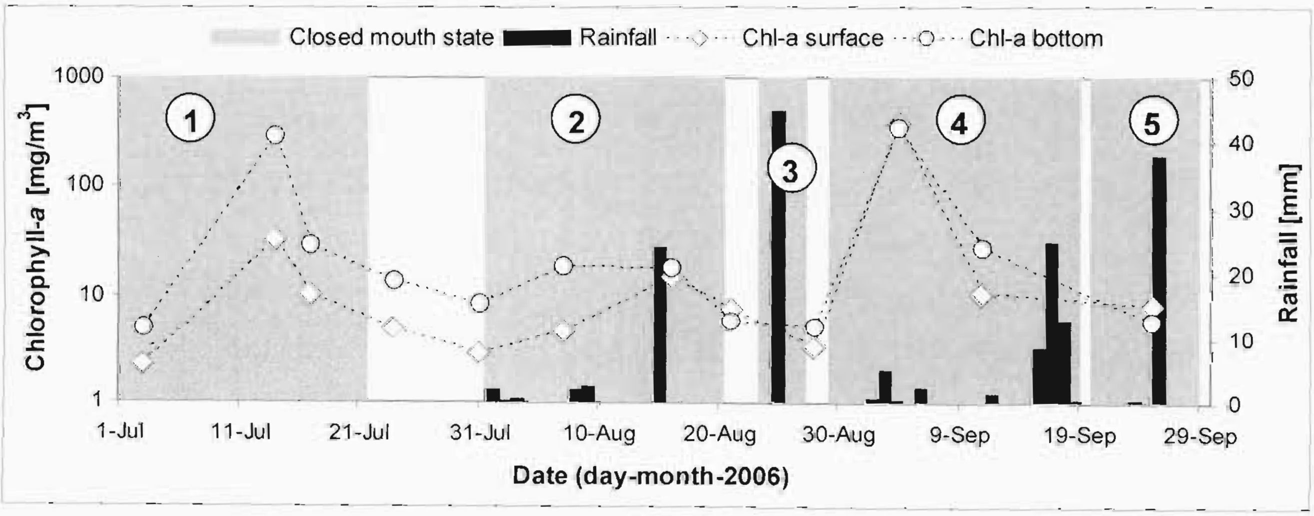


**Figure 4.5 Measured phytoplankton, rainfall and mouth state vs. time at the Mhlanga Estuary, during the study period.**

The biomass of phytoplankton at bottom waters was generally greater than that of the phytoplankton near the surface. The study by Perissinotto *et al.* (2002) at the Mpenjati Estuary (TOCE) found larger concentrations of microphytobenthos than surface phytoplankton and suggested that this was a result of settled-out phytoplankton that had become part of the epibenthic flora. Therefore a more even distribution of plankton would be expected during open mouth conditions due to re-suspension. In this study, bottom phytoplankton biomass was consistently greater than surface phytoplankton biomass. It is suggested that biological processes, such as different productivity or grazing rates, were responsible for the difference in biomass between the two subsystems as opposed to vertical physical processes.

Subsequently, Perissinotto *et al.* (2003) found that the impact of zooplankton on phytoplankton was substantial, even exceeding its availability at times. It was suggested that the metabolic needs of the zooplankton community is met entirely by phytoplankton, at least during open mouth conditions, and when there the phytoplankton is insufficient to supply the metabolic requirements of the zooplankton, large proportions of benthic microalgae (microphytobenthos) are consumed. In addition, Thomas *et al.* (2005) found that at the Mhlanga Estuary, the minimum recorded phytoplankton chl-*a* biomass corresponded to the second highest zooplankton recording, taken during the study. It is suggested that zooplankton grazing may be responsible for the regulation of algal biomass.

### 4.4.1 Bloom time

Closed mouth conditions were characterised by a sharp increase in biomass followed by a sudden decline. This is evident in phases one, two and four (refer to Fig. 4.5). The residence time of phase three and five, were not long enough to allow biomass to follow the same pattern. The bloom that occurred in the first phase included higher concentrations of bottom phytoplankton compared to surface phytoplankton. This is an indication of an ageing bloom that is decaying towards the bottom. During the second closed mouth state, biomass increased and remained consistent before declining. The nine day interval between the sixth and seventh chl-*a* sample (phase two in Figure 4.5) coupled with the corresponding decrease in DO concentrations (see Figure 4.8) suggests that a bloom probably occurred between the sampling times and was therefore missed. This would be consistent with the biomass activity recorded in the first and fourth closed mouth periods.

In order to establish the bloom time, $T_B$, the time taken from mouth closure for an exceptional algal bloom to occur was considered. Biomass samples were taken weekly and therefore did not allow an accurate bloom time to be established. Therefore, from the results from phase one and three (Figure 4.9) a bloom time, $T_B$, ranging from approximately 12 to 16 days for the Mhlanga Estuary was considered. The mouth state only remained open for a short period after the third event, therefore preventing a complete flush of phytoplankton biomass. Consequently, this may have contributed to the sudden increase in algal biomass during the fourth closed mouth state, thereby allowing a faster bloom time. Assuming that the mouth effectively remained closed between the third and fourth event, a more reasonable bloom time of 13 days was obtained.

A representative time of 14 days (2 weeks) is therefore suggested as a current best estimate. It is important to note that this value is not necessarily the same for all similar TOCEs as it is likely to depend on site-specific parameters such as temperature, mixing, etc.

## 4.5 Physico-chemical results

The measurements of the physico-chemical parameters, including irradiance, temperature, salinity, dissolved oxygen and nutrient concentrations are summarized in Appendix C2. It is important to note that although these parameters were not used in the model, they assist in interpreting the processes that underpin the changes in the eutrophication indicators. Physico-chemical parameters of the Mhlanga Estuary, investigated by Perissinotto *et al.* (2004) and Begg (1984) in 1964, are summarised in Appendix C1.

### 4.5.1 Irradiance

The minimum light attenuation coefficient ($K_d$) was 0.84 $m^{-1}$ (16 August 2006) and the maximum $K_d$ was 4.89 $m^{-1}$ (21 August 2006). As illustrated in Figure 4.6, the lowest $K_d$ values coincided with closed mouth conditions and the highest values with open mouth conditions. These results are consistent with results obtained from other studies of TOCEs (Perissinotto *et al.*, 2003, Gama *et al.*, 2005) and results from the study of the Mhlanga Estuary (from 2002 to 2003) by Thomas *et al.*, (2005).

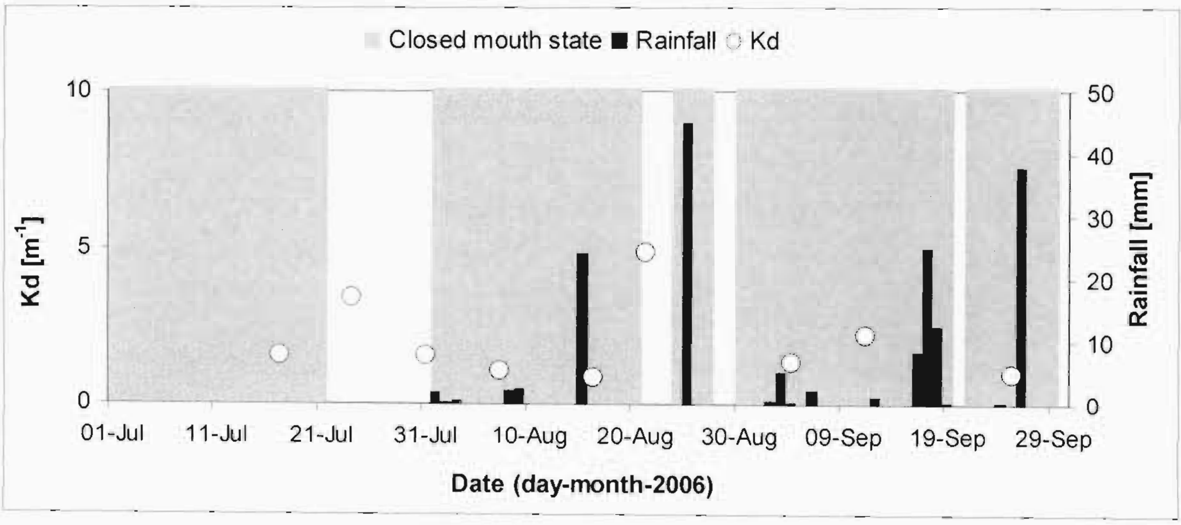


**Figure 4.6 Temporal variations in the diffusive attenuation coefficient, $K_d$ and rainfall in the Mhlanga estuary during the sampling period**

During closed mouth conditions, phytoplankton growth may increase the attenuation of light with depth to such an extent that by self-shading they limit their own growth (Kirk, 1994). This does not seem to be the case as $K_d$ remains fairly constant throughout closed mouth conditions and when algal blooms were recorded. The lower values recoded during closed mouth conditions are as a result of calmer conditions (Perissinotto *et al.*,2003). The slight increase in $K_d$ during open mouth conditions may be as a result of tidal mixing in the estuary. Rainfall occurred on several occasions, as illustrated in Figure 4.6, however, little if any, influence was shown. The euphotic depth, $Z_{eu}$, was greater than the total depth, $Z_{tot}$, for each sampling event indicating that light was probably not a limiting factor of phytoplankton growth.

### 4.5.2 Temperature and Salinity

Measured top and bottom water column temperature and salinity were plotted in Figure 4.7. Fairly consistent temperatures within the water column were recorded throughout the sampling period, with a maximum difference of 2.5 °C between top and bottom temperatures. Overall temperatures varied slightly from 16.7 °C to 22.8 °C throughout the sampling period. Warm, evenly distributed temperatures provide ideal conditions for phytoplankton growth.

Significant differences in top and bottom salinity indicate salinity stratification, with two incidences showing highly stratified conditions. These occurred during open mouth conditions, with the most saline water at the bottom and fresh water running over the top. As more saline water enters the estuary, it accumulates in the deeper parts of the estuary. The decay in salinity stratification is evident during the closed mouth phase as mixing occurs. It is speculated that the mixing is driven primarily by wind (e.g. Thomas *et al.*, 2005).

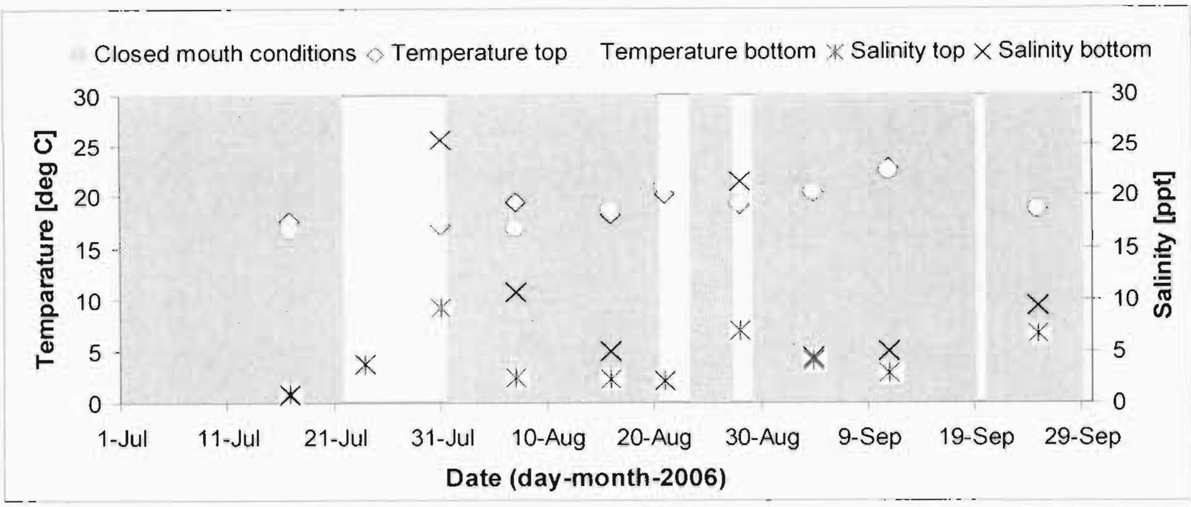


**Figure 4.7 Temporal variations in top and bottom water temperature and salinity in the Mhlanga Estuary during the study period.**

Phytoplankton concentrations appear to be better related to salinity stratification than irradiance. The absence of salinity stratification, especially during closed mouth conditions may account for the higher concentrations of phytoplankton observed. The absence of stratification allows the phytoplankton to remain suspended in the water column rather than sink towards the bottom. This allows the phytoplankton to attain higher productivity through the optimal utilization of available light (Thomas et al., 2005).

### 4.5.3 Dissolved Oxygen

During the sampling period, the Mhlanga Estuary exhibited dissolved oxygen (DO) concentrations ranging from 5.21 to 17.13 mg/L. Figure 4.8 presents the DO concentrations measured at the top and bottom of the water column during the study period. The drop in DO concentration is consistent with the consequence of an algal bloom and the subsequent decrease in biomass that was present in the fourth phase. This suggests that a bloom may have also occurred in the second phase, especially because there was a nine day interval between sampling events. The relatively high DO concentrations recorded on the 3rd of September 2006 were probably as a result of the strong winds experienced on that day.

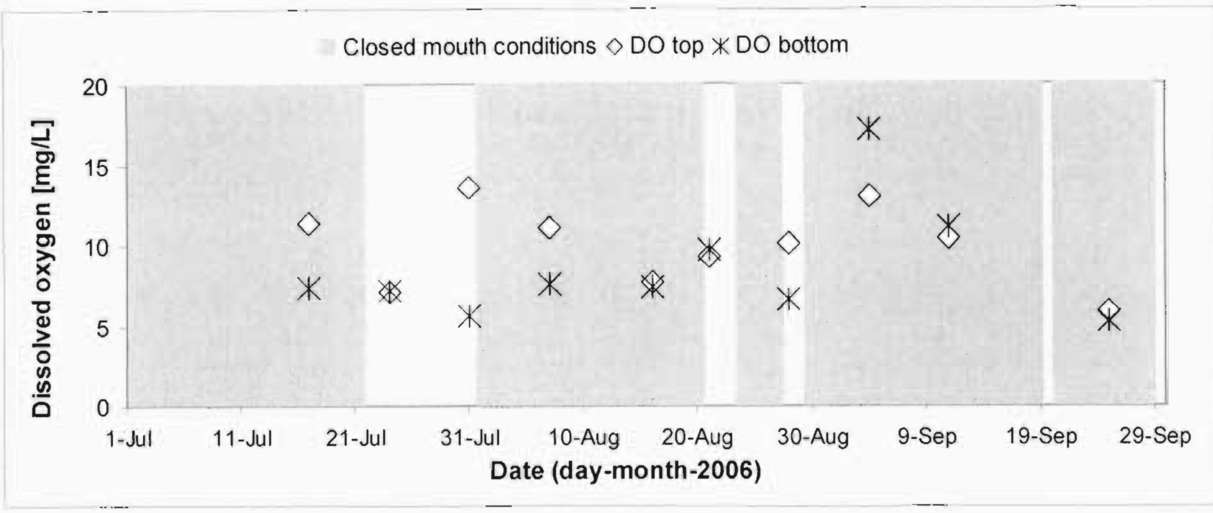


**Figure 4.8 Temporal variations in dissolved oxygen concentrations for the Mhlanga Estuary during the study period.**

Findings by Perissinotto *et al.*, 2004 (Appendix C1) indicate a common seasonal trend displayed by DO levels and temperature. The warmer months are associated with low DO concentrations and high temperatures whereas the colder months have higher DO concentrations. Therefore, one would expect lower DO concentrations than those recorded here in the summer months, thereby increasing the likelihood of severely anoxic conditions.

### 4.5.4 Macronutrients

Discharge data from the Phoenix and Mhlanga WWTWs, from 2002 to 2006, indicate average DIP concentrations of 5.3 mg/L and 5.1 mg/L respectively. DIN concentrations from the Phoenix WWTW increased from 2002 to 2004 and then decreased significantly in 2005 and then again in 2006. In 2002 and 2004, the Mhlanga WWTW discharged significant concentrations of DIN, of up to 27 mg/L, otherwise averaging about 8mg/L. Seasonal variations in DIN concentrations occurred, however DIP concentrations remained more consistent. DIN:DIP ratios of the treated wastewater averaged 2.9 for the Phoenix and 1.9 for the Mhlanga WWTWs.

Measured water column DIN and DIP concentrations were plotted against time, as illustrated in Figure 4.9. Bottom DIN and DIP concentrations are significantly less than surface concentrations. DIP tends to increase during closed mouth conditions and decrease during open mouth conditions. This may be an indication of the accumulation of nutrients within the estuary. The maximum DIP concentration was 2.7 mg/L (closed mouth conditions, top of the water column) and the minimum, 0.7 mg/L (closed mouth conditions, bottom of the water column). According to DWAF (1998), DIP concentrations exceeding 0.25 mg/L are often related to eutrophic conditions (refer to section 2.5.4). All measured top and bottom DIP concentrations exceeded this value by more than three times.

Top and bottom DIN concentrations tended to decrease during closed mouth conditions, with a long term downward trend in top concentrations observed over the study period. This is possibly due to the decrease in DIN concentrations discharged from the WWTWs into the river over the study period. The maximum DIN concentration was 6.4 mg/L (open mouth conditions, top of the water column) and the minimum, 1.4 mg/L (closed mouth conditions, bottom of the water column). The majority of surface DIN concentrations exceeded 2.5 mg/L, which according to DWAF (1998) are often related to eutrophic conditions (refer to section 2.5.4).

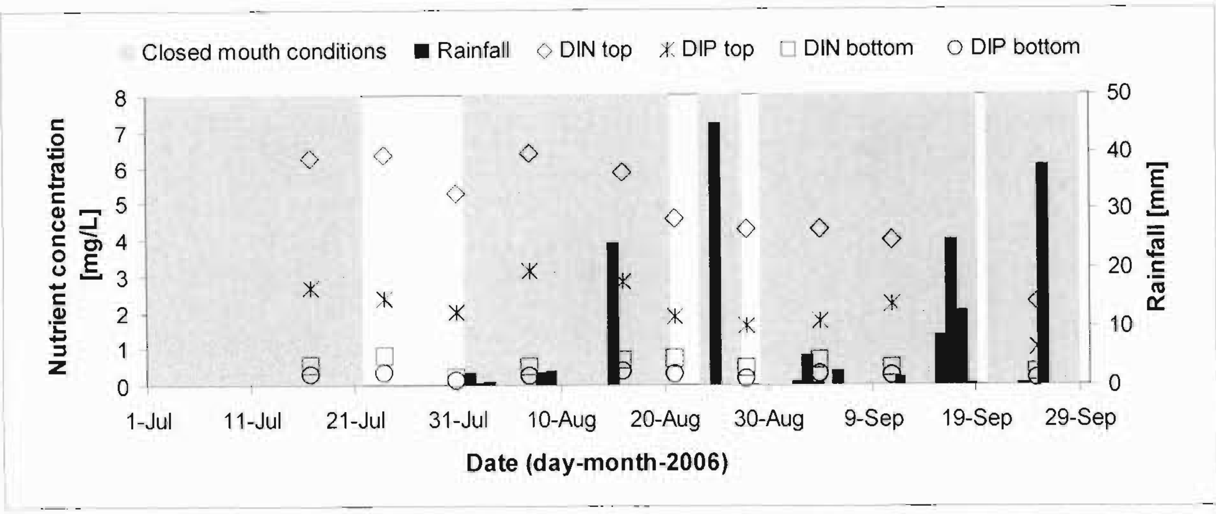


**Figure 4.9 Temporal variations in DIN and DIP concentrations at the Mhlanga estuary during the study period.**

Measured DIN:DIP ratios in the estuary ranged from 1.87 to 2.66, suggesting nitrogen limitation, however, both DIN and DIP concentrations were above the values characteristic of eutrophic conditions. The over-abundance of nutrients present in the estuary provide the ideal conditions for algal blooms to occur (provided that other environmental conditions are favourable).

## 4.6 Simulation Using Observed Mouth State Data

Observed mouth state data were initially used in the water balance and eutrophication models in order to reduce the dependency of the model on questionable inflow data. The data were recorded by field rangers from KZN Wildlife. Various values for the model parameters, such as the maximum storage and minimum and maximum water levels were tested, using observed mouth state data and recorded water levels (DWAF).

### 4.6.1 Maximum storage

The maximum storage was assumed equal to 800 000 m$^3$ as discussed in section 3.3.3 (Stretch and Zietsman, 2004). A sensitivity analysis of this parameter of the water balance model was performed by Londal (2005). Results showed that the mouth state was most sensitive to decreases in the maximum storage. In order to test what would happen if the maximum storage was increased, storage values were simulated using actual mouth state observations with *spillage* features. Simulated water levels were then compared to measured water levels. With the maximum storage set at 800 000 $m^3$, results showed that from mid-August 2005 until mid-November 2006, spillage occurred 3 % of the time. An increase in the maximum storage to 1000 000 $m^3$, decreased the spillage to 2 %. By increasing the maximum storage further, spillage decreased, however the simulated water levels did not correspond well with those measured. Simulated water levels corresponded best to measured water levels with a maximum storage of 800 000 $m^3$.

A rough estimate of the dead storage was estimated from an approximation of the remaining water surface area and the assumed average water level during open mouth conditions. The average water level during open mouth conditions was taken as 0.85 m (refer to Figure 4.10) and the remaining surface area was assumed to be 60 % of the maximum surface area, 70 ha (refer to Table 3.1). A rough estimation of 300 000 $m^3$ was assumed for the dead storage. The eutrophication model is sensitive to changes in the dead storage. The dead storage should therefore be verified in future studies by means of a detailed hydrographic survey. It is important to note that the aim of this investigation is the formulation of a eutrophication model and not the testing of data.

### 4.6.2 Recorded and simulated water levels

Water levels were simulated and plotted against recorded water levels using the observed mouth state data (see Figure 4.10 and Figure 4.11). The minimum water level, $H_{min}$, used in the water balance model, was taken as 930 mm. The maximum water level, $H_{max}$, was set equal to 3200 mm. Measured water levels correlated well with simulated water levels, with slight variations occurring intermittently. This is probably due to the inaccuracy involved in simulating river inflows from gauge U3H005. Simulated water levels were sensitive to changes in the assumed maximum storage and maximum and minimum water levels as previously noted by Londal (2005).

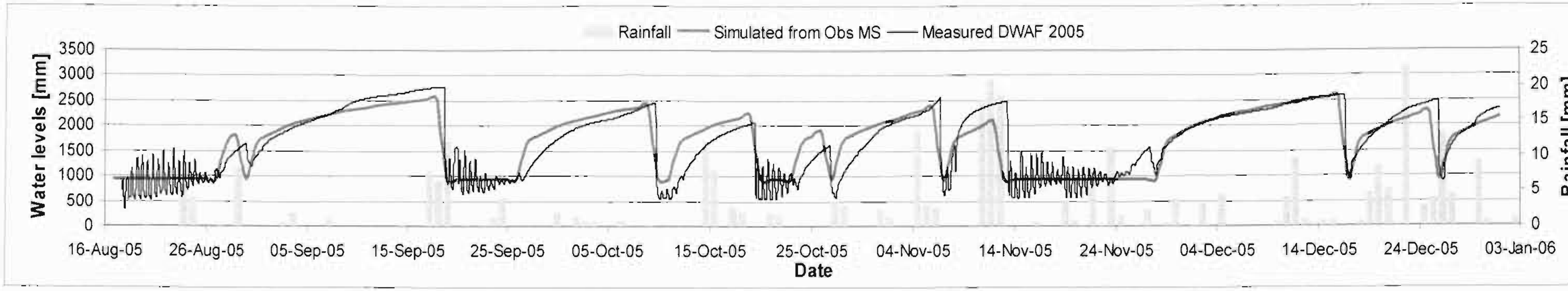


**Figure 4.10 Temporal variations in rainfall and simulated and measured water levels (provided by DWAF) of the Mhlanga Estuary from August to December 2005.**

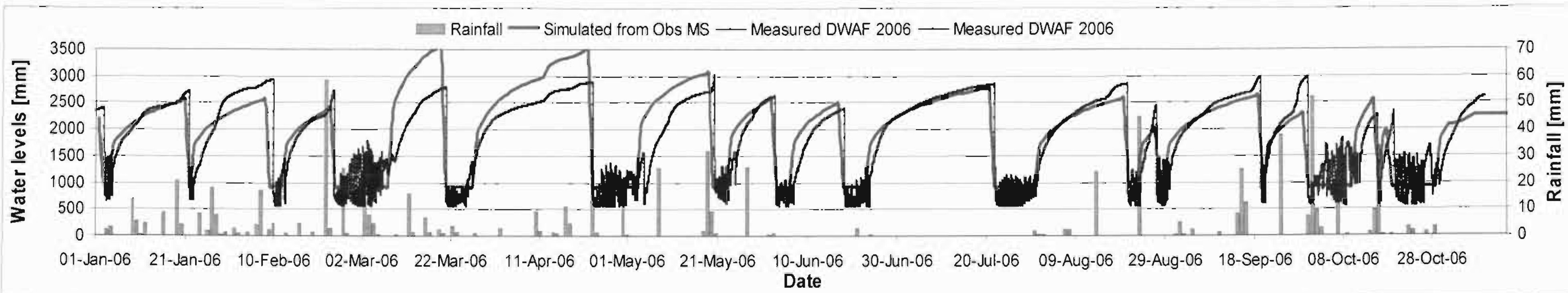


**Figure 4.11 Temporal variations in rainfall, simulated and measured water levels (provided by DWAF) of the Mhlanga Estuary from January to October 2006.**

### 4.6.3 Nutrient concentrations

By assuming that nutrients entering the estuary are conserved, nutrient concentrations within the estuary were simulated using the water balance model and the wastewater effluent data. Measured and simulated DIP and DIN concentrations were plotted in Figure 4.12 and Figure 4.13 respectively. The measured top and bottom DIN and DIP concentrations were averaged. The DIP and DIN concentrations were assumed to be 1.3 mg/L and 3.3 mg/L, respectively, during open mouth conditions. These values were assumed from the average measured DIP and DIN concentrations taken subsequent to closure of the mouth. More frequent monitoring of DIN and DIP concentrations would be required to make a more accurate assumption. It is important to note that these values were selected to test the model methodology and are therefore not necessarily accurate. These values are not critical as they do not affect the eutrophication model.

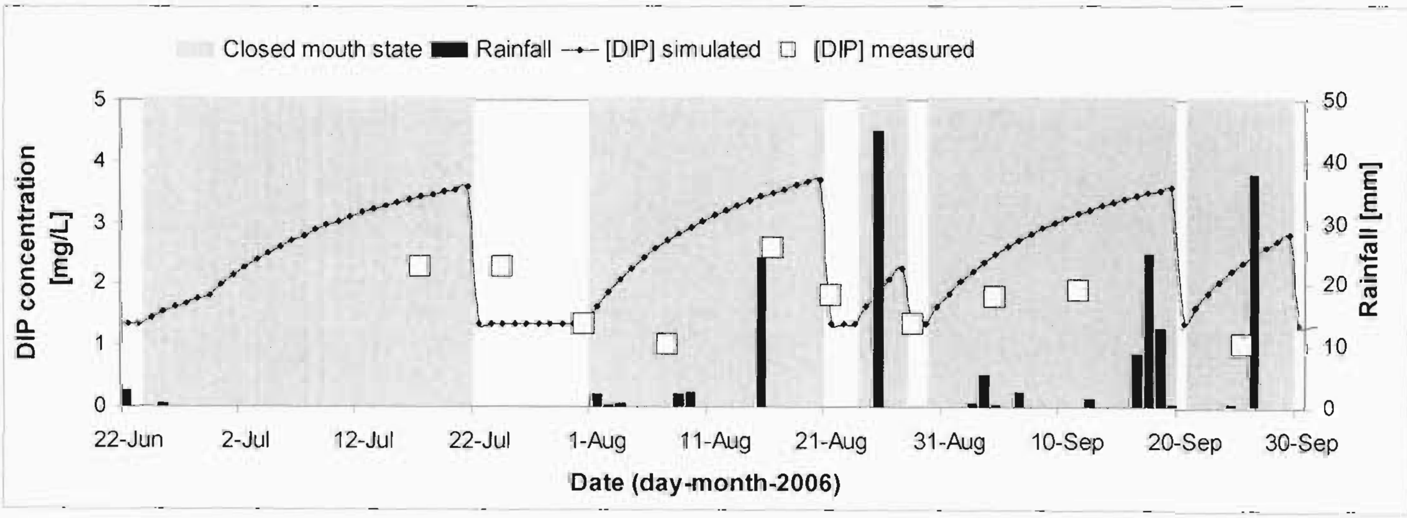


**Figure 4.12 Temporal variations in measured and simulated DIP and rainfall in the Mhlanga Estuary during the study period.**

Measured DIP concentrations illustrate a saw-tooth pattern (refer to Figure 4.12) where concentrations increase during closed mouth conditions and decrease during open mouth conditions. It is speculated that the increase in DIP during closed mouth conditions is an indication of the accumulation of nutrients within the estuary. Simulated conserved DIP concentrations gradually increase during closed mouth conditions. It is speculated that the maximum DIP concentration occurred following a rainfall event (15th August 2006), suggesting an increase in DIP due to runoff from nearby farmlands. This is also illustrated after the rainfall event on the 3rd September 2006, where the DIP concentration is relatively high considering an algal bloom occurred at that time.

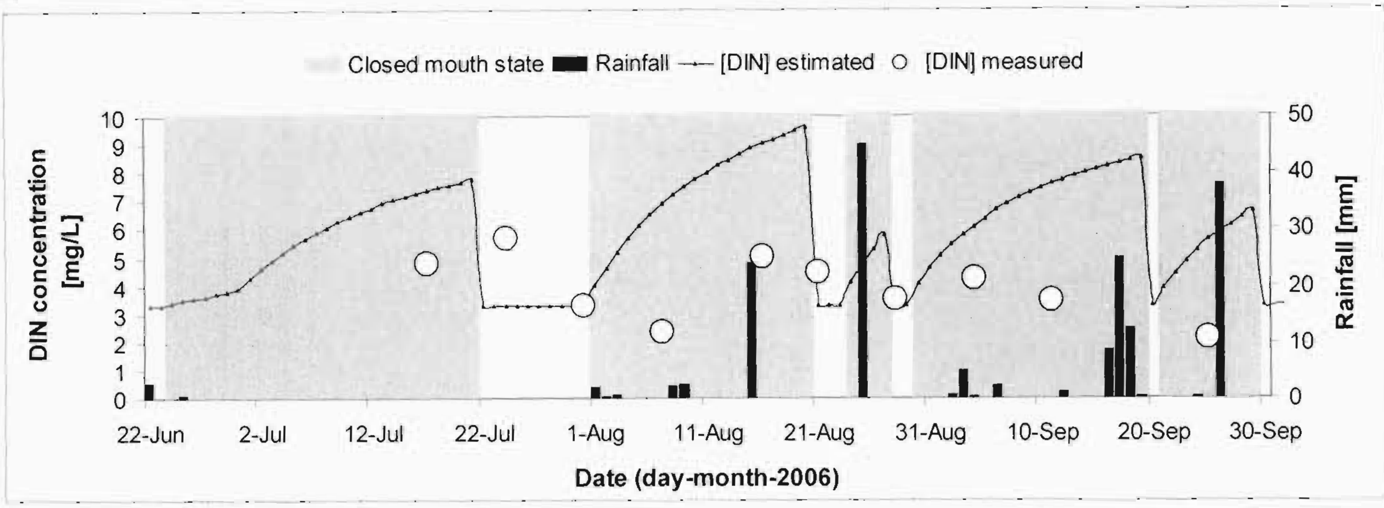


**Figure 4.13 Temporal variations in measured and simulated DIN and rainfall in the Mhlanga Estuary during the study period.**

Figure 4.13 illustrates a decrease in DIN concentrations during open mouth conditions. There is no clear pattern during closed mouth conditions. However, during the second closed mouth phase, there is a drop in DIN concentrations followed by a sharp rise. It is speculated that because this measurement occurred immediately after a rainfall event (on the 15th August 2006), it may be as a result of DIN input as a result of runoff from nearby sugarcane fields. The only other measurement taken immediately after a rainfall event was on the 4th September 2006. This value is relatively high considering that an algal bloom occurred at that time although only 5 mm occurred on that day.

Although the measured nutrient concentrations provide an indication of in-situ conditions, it is not possible to conclude any valid arguments due to the frequency of sampling (gaps in the existing results).

In order to obtain an indication of the nutrient uptake by phytoplankton for growth, the following method was considered. The differences in simulated conserved and measured concentrations of both DIN and DIP concentrations were estimated. The ratio of the difference in DIN concentrations to the difference in DIP concentrations, during closed mouth conditions, was plotted against time in Figure 4.14. The relatively significant decrease in DIN (large DIN:DIP ratio) during the second, fourth and fifth phases are consistent with the Redfield ratio, where the DIN requirement is larger than the DIP. This is an indication of the nutrient uptake by plants for growth. The DIN:DIP ratio decreases following the occurrence of an algal bloom and may be an indication of nutrient limitation, as estuaries are generally nitrogen limited due to sewage inputs or due to the declines in phytoplankton growth as witnessed in Figure 4.7. The measurements taken in the first phase were taken subsequent to an algal bloom and therefore the ratio is low.

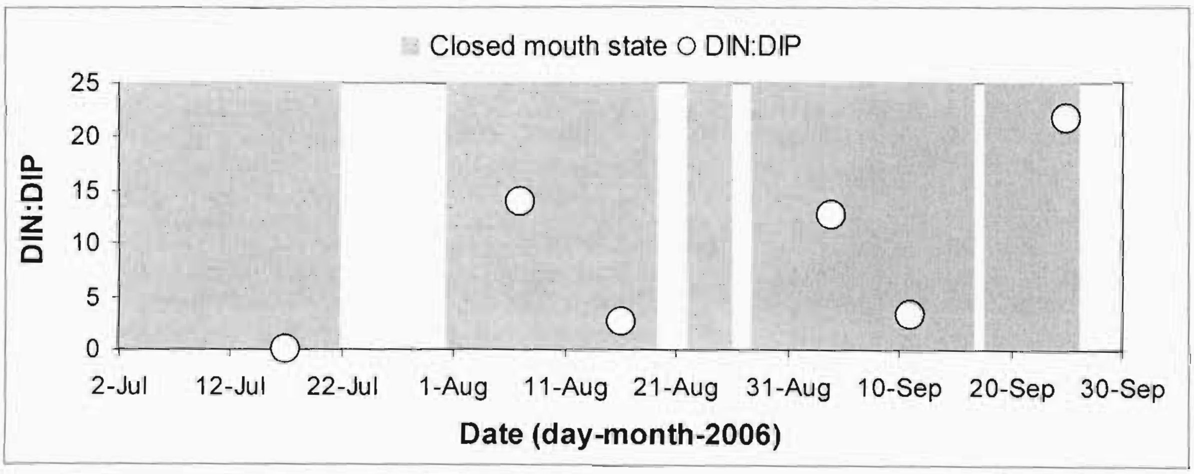


**Figure 4.14 Temporal variations in the ratio of the uptake of DIN to the uptake of DIP used by phytoplankton for growth for the study period**

### 4.6.4 Grey water index

The *GWI*, $T_R/T_B$ and chlorophyll-*a* concentration were plotted against time, as illustrated in Figure 4.15. The *GWI* was simulated on a daily time step during open mouth conditions. Since, algal blooms are a function of residence time and nutrients, a bloom is only expected to occur if $T_R/T_B$ exceeds one and the GWI attains a value large enough to ensure sufficient nutrients to supply phytoplankton growth. Assuming a bloom time of 14 days, when $T_R/T_B$ is equal to one, *GWI* is approximately 50%. The minimum *GWI* is therefore 50% in this case.

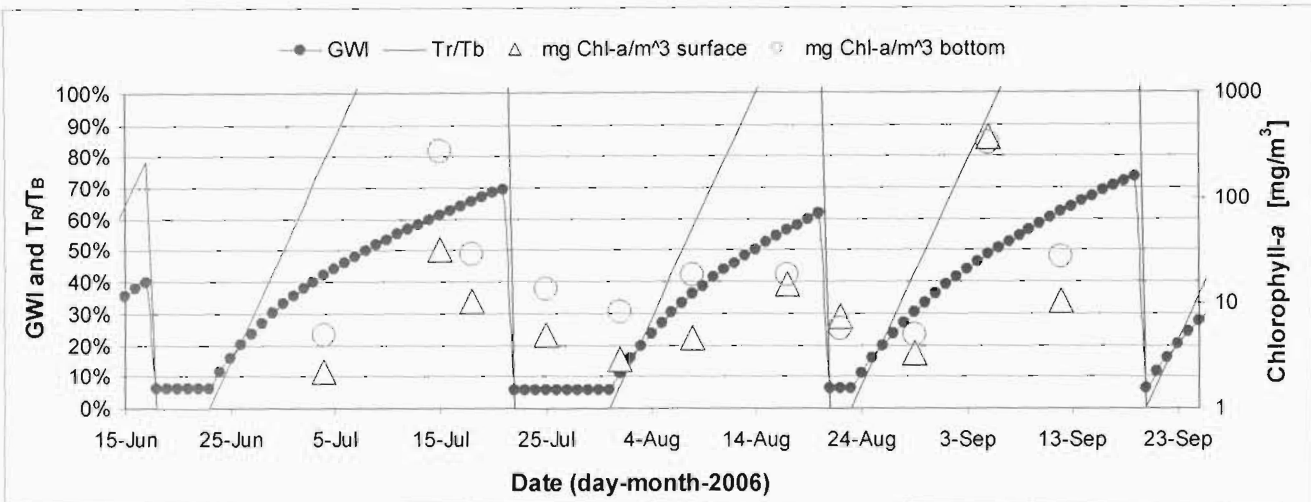


**Figure 4.15 Grey water index, $T_R/T_B$ and chlorophyll-*a* concentration at the Mhlanga Estuary during the study period.**

The GWI can be linked to nutrient loadings from the WWTWs in order to estimate nutrient concentrations in the estuary. The product of GWI (50 %, at a bloom) and nutrient concentrations of the effluent resulted in a DIN concentration of 7 mg/L and DIP concentrations of 2 to 3 mg/L. Both these values exceed the concentrations at which eutrophic conditions would be expected (Section 2.5.4).

### 4.6.5 Eutrophication index and potential

In order to establish the eutrophication index, $E_i$, at which a bloom would occur, chl-*a* concentration and $E_i$ values (see Equation 3.12) were plotted against time (see Figure 4.16). The mouth state was assumed to remain closed on the 28 and 29th August 2006, even though a brief open period was recorded. The results indicate that the $E_i$ at which a bloom may be expected is 50 %, since $E_{i\,crit} = GWI_{crit}$. This parameter may be included in the water balance model to predict eutrophication and consequent algal blooms. This method provides a temporal and quantitative indication of how eutrophic conditions occur.

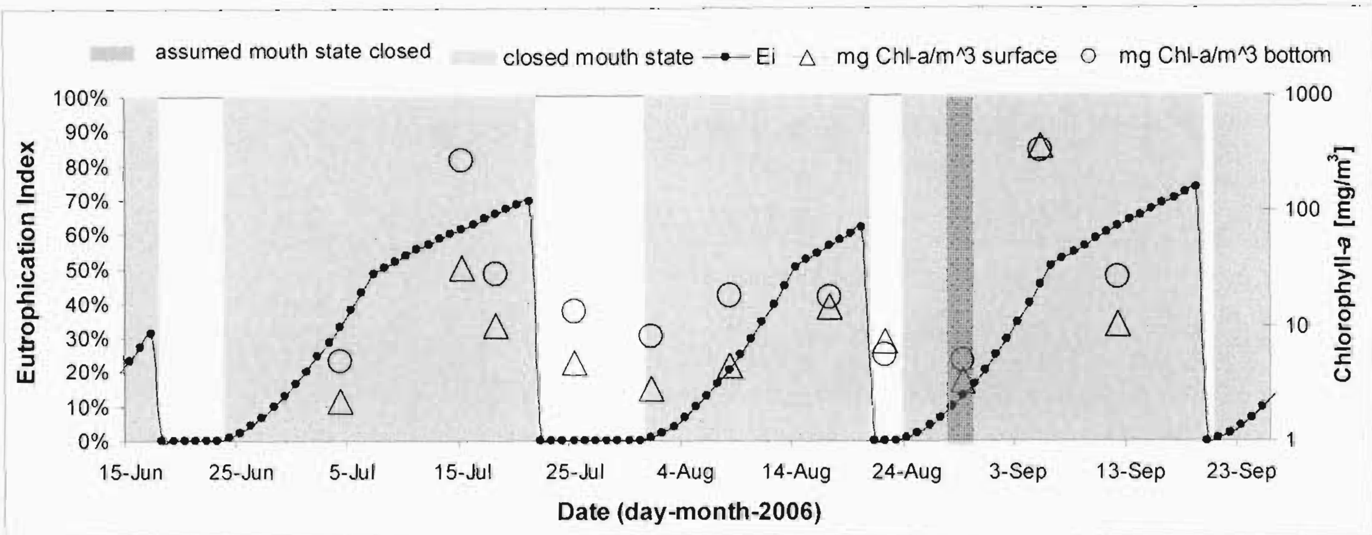


**Figure 4.16 Temporal variation in the Eutrophication index and measured chlorophyll-*a* concentration for the study period.**

Alternatively, the potential for eutrophication to occur, $E_p$, was simulated for the duration of the study period and plotted against time in Figure 4.17. The *GWI* and $T_R/T_B$ ratio and chlorophyll-*a* concentrations were also included. This method gives a simple indication of when a bloom (when chlorophyll-*a* is greater than 100 mg Chl-*a*/m$^{-3}$, Adams and Bate, 1999) may occur. From Figure 4.17 and with reference to section 4.4, it can be seen that this indicator works well for predicting the actual blooms that occurred during the study period.

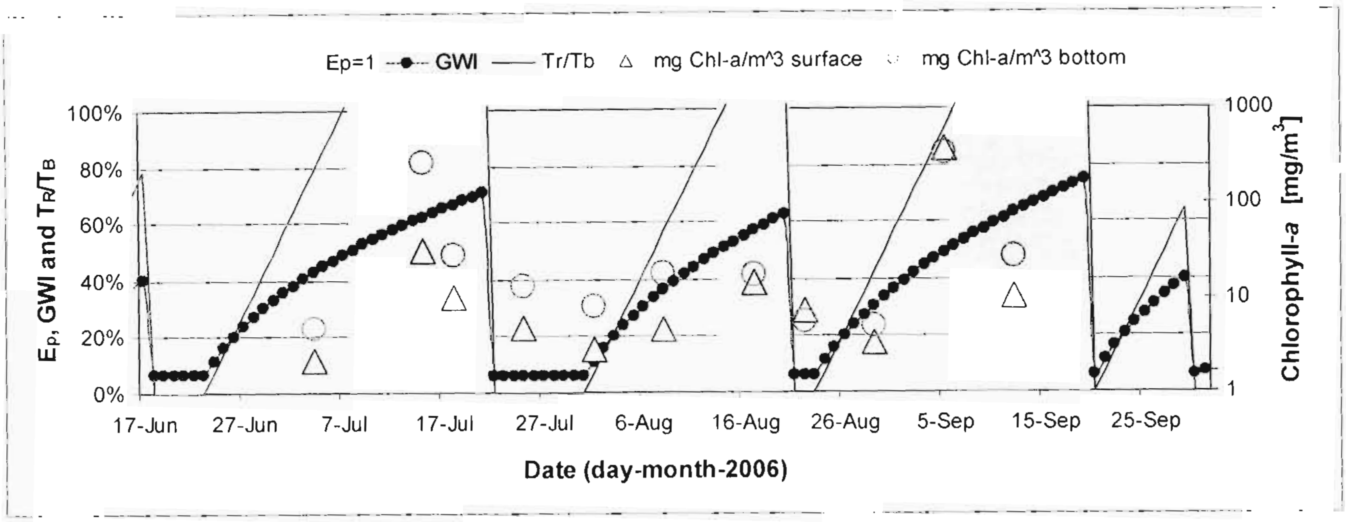


**Figure 4.17 Temporal variation in $E_p$, $GWI$, $T_R/T_B$ and chlorophyll-*a* concentration for the study period.**

## 4.7 Scenario Simulations

Simulations were run for four possible scenarios. Simulated mouth state data were used as opposed to observed mouth state data in order to make predictions and test the model.

These included:

1. natural state (excluding discharge from WWTW)
2. the present state
3. 75 % increase in both the Phoenix and Mhlanga WWTWs and
4. 150 % increase in both the Phoenix and Mhlanga WWTWs.

### 4.7.1 Mouth state

Mouth state data were simulated for each scenario, using almost five years of flow data (from 2002 to 2006), in order to quantify the influence that WWTW discharge has on the mouth state. The flow data included river inflow data and WWTWs discharge data. The results are tabulated in Table 4.1.

**Table 4.1 Simulation results concerning the mouth state (MS) for each of the possible scenarios and for comparison, the present state.**

| | Natural State (Simulated MS) | Present State (Observed MS) | Present State (Simulated MS) | 75 % increase WWTWs flow | 150 % increase WWTWs flow |
|---|---|---|---|---|---|
| **% closed** | **91%** | 68% | 77% | 67% | 57% |
| **% open** | **9%** | 32% | 23% | 33% | 43% |
| **Number of breaches** | **10** | 76 | 44 | 46 | 54 |
| **Average breaching period [days]** | **16** | 7 | 9 | 12 | 14 |

Table 4.1 gives an indication of the effect of the change of flows on the mouth state of the Mhlanga Estuary. Results indicate that as the flow increases, open mouth conditions occur more frequently and for a longer period of time. The results suggest that the present state, using observed mouth state data, has significantly more breaching events when compared to simulations of the same case where the mouth state is predicted by our model (Section 3.3.6). This is because mouth state model does not account for breaching events bought on by inadequate reconstruction of the berm and by rainfall events which are not reflected in the inflow data. If the Mhlanga Estuary experienced no treated sewage discharge, the estuary would remain closed for the majority (91%) of the time with substantially fewer breaching events. This gives an indication of how the wastewater discharge has interfered with the natural hydrological functioning of the estuary.

### 4.7.2 Eutrophication

The eutrophication index ($E_i$) was simulated and plotted for the different possible scenarios. As shown in Section 4.6.4, a bloom would be expected to occur if the

$E_i$ exceeded 50 %. Therefore, the number of bloom events expected per year was estimated.

**Table 4.2 Eutrophication events simulated from March 2002 to November 2006 for the possible scenarios and including the present state.**

| | Present State (Observed MS) | Present State (Simulated MS) | 75 % increase WWTWs flow | 150 % increase WWTWs flow |
|---|---|---|---|---|
| **Bloom events ($E_i$ >50 %)** | 15 | 24 | 41 | 50 |
| **Blooms per year** | 3 | 5 | 9 | 11 |
| **Number of closed mouth periods** | 77 | 45 | 47 | 55 |
| **Average closed time [days]** | 15 | 29 | 24 | 18 |

Table 4.2 illustrates the increase in bloom events and blooms per year expected as a result of increased wastewater flows. The increase in the number of closed mouth periods coupled with the decrease in average residence time confirms that breaching would increase with an increase in flow.

Despite the increase in flushing potential due to the increased wastewater inflows, the average residence time exceeds the time required for bloom development. However, it is apparent that further increases in WWTW discharge could decrease $T_R$ to less than $T_B$, thereby preventing algal blooms from occurring. This may have drastic adverse affects on the overall ecological functioning of the estuary as changes in the frequency, timing and duration of closed mouth conditions may interfere with migration patterns of species between the ocean and estuary (Thomas *et al.*, 2005).

# 5. CONCLUSIONS

## 5.1 Introduction

TOCEs are highly complex systems that are sensitive to change. Therefore, development of a general theory explaining the interrelationships within an estuarine system is almost impossible. The reliability of complex models is questionable due to the uncertainties associated with the numerous input parameters. Simplified modelling is therefore more desirable.

Anthropogenic activities, such as abstractions and discharges can place significant pressures on the physical, biological and chemical functioning of an estuarine system. Changes in flow can invariably affect the dynamics of the mouth. Excessive cultural eutrophication has become a worldwide pollution concern. Nutrient loading into estuarine systems, as a result of anthropogenic practices, can have severe adverse effects on the ecology of such a system. These include toxic algal blooms, anoxic conditions, fish kills and so on. Despite increases in the flushing potential of polluted waters due to increased wastewater inflows, changes in mouth dynamics may interfere with migration patterns of species between the ocean and estuary, which in turn can significantly impact the overall functioning of the system.

The aim of this study was to formulate a model in order to predict future eutrophication events and trends at the Mhlanga Estuary, which would later be used in other similar TOCEs. This included modelling the water balance of the system, in which a simple daily-time-step water balance model was selected. The model included various inputs and outputs of the system, residence time, storage, breaching water levels and time for mouth closure to occur. The result of the water balance model was a daily prediction of the mouth state and volume.

This provided an indication of the breaching frequency. Observed mouth state data and measured water levels were used to test the model.

A complete validation of this type of model requires accurate inflow data and mouth observations as these are not currently available. The observed mouth state data provided reasonable capture of the mouth state, however the precise timing of breaching events and mouth closure processes need further investigation. It is important to note that mouth state prediction was not actually relevant to this study and could be omitted.

In order to predict eutrophication events and trends at the Mhlanga Estuary, it was required that the conditions at which this would occur be established. This included the collection of physico-chemical parameters and chlorophyll-*a* concentrations. Although physico-chemical parameters and phytoplankton concentrations were recorded on a weekly basis, there were still significant gaps in the results, indicating that eutrophication processes occur at a quicker rate. The three bloom events (although one was not directly captured, but inferred from data) that occurred over the duration of the study indicate a pattern.

Due to the complexity required in developing a model of the nutrient dynamics, a simpler approach was derived. The grey water index (GWI) is a proxy variable that was formulated in order to account for nutrient loadings (DIN and DIP) into the estuary. Initial results showed that under ideal conditions, an algal bloom would occur 12 to 16 days following re-closure of the Mhlanga mouth. This was referred to as the bloom time ($T_B$). The eutrophication index ($E_i$) was then formulated to account for both residence time and nutrient concentrations. The $E_i$ at which eutrophic conditions can be expected was found to be 50 %. It is important to note that this value is sensitive to changes in the dead storage. It is important to note that although the $E_i$ and $T_B$ values are expected to be site specific, the concepts are not.

Various flow scenarios were selected and simulations of both the water balance and eutrophication models run. Water levels simulated using the water balance model and observed mouth state data reproduced those measured by DWAF. It was found that an increase in capping flows resulted in more frequent breaching events and longer open mouth conditions. The risk of eutrophic conditions also increased with an increase in WWTW discharge flows. Algal blooms continued to occur despite more frequent breaching events, induced by an increase in capping flows.

This study is an indication of the importance of the integration of multidisciplinary fields, as no ecological system can be divided into sections without having an interrelated impact.

## 5.2 Recommendations for Further Research

A complete validation of the water balance model requires that accurate inflow data and mouth observations are recorded. The limited available data on the Mhlanga Estuary reduces the accuracy of predictions and highlights the need and importance for system monitoring. Accurate continuous inflow monitoring is required as it is significant in understanding the influence of flow on mouth dynamics. It is recommended that an alternative method be used, such as the method outlined by Smakhtin (2004) including stream/river FDCs.

A detailed hydrographical survey of the estuary is required for accurate storage/volume estimations. This is important as both the water balance model and eutrophication model are sensitive to changes in the storage capacity. More detailed information of beach sediment dynamics and wave characteristics would contribute to a better understanding of the time required for mouth closure.

It is recommended that samples of phytoplankton biomass, physico-chemical parameters and nutrient concentrations be taken at a higher resolution (e.g. daily), as weekly samples were insufficient to provide an accurate representation of how eutrophication develops. Daily zooplankton samples are also recommended in order to better understand the food-web dynamics.

It is important to note that both the Phoenix and Mhlanga WWTW will be redirecting the discharge in the near future (H. Blair, eThekwini Municipality, pers comm.). This will have significant impacts on the biological, physical and chemical functioning of the Mhlanga Estuary and would provide a unique opportunity to study the recovery process and rehabilitation of this polluted system. This will in turn inform future management strategies for similar estuaries that are affected by eutrophication.

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# APPENDICES

# APPENDIX A

SPREADSHEETS AND FLOWCHARTS USED FOR THE WATER BALANCE MODEL AND EUTROPHICATION MODEL.

## A1: Spreadsheet of the water balance and eutrophication models

**MHLANGA ESTUARY**

| | | |
|---|---|---|
| Max storage | 800000 | $m^3$ |
| Dead storage | 300000 | $m^3$ |
| Hmax | 3200 | mmGMSL |
| Hmin | 930 | mmGMSL |
| Max Seepage | 0.230 | $m^3/s$ |
| Max E/T losses | 0.027 | $m^3/s$ |
| Tclose = | 8.72 | days |

| | |
|---|---|
| k = | 3 |
| m = | 2 |
| Bloom time = | 14 days |

=IF(O13=0,0,MIN(J12+(F12-E12-F12)*24*3600,$B$2))

=MINI((J13/$B$2)^(1/$F$2)*($B$4-$B$5)+$B$5, $B$4)

=IF(O13=0,(M12+1),0)

=$B$6*(K12/$B$4)^$F$3

=IF(AND(O13=0,I13>($B$8)),IF(N12="",$B$8,N12-1),IF(O13=1,"",$B$8))

=$B$7*((K12-$B$5)/($B$4-$B$5))^($F$2-1)

=$B$2/F12/(24*3600

=(IF(K12>=$B$4,0,IF(AND(O12=0,I13<($B$8*$H$8)),0,IF(AND(O12=0,N12>1),0,1))))

=S12/L12

| Date | INFLOWS: Inflow river "white water" | Inflow WWTW Phoenix | Inflow WWTW Mhlanga | "black water" | "grey water" | OUTPUTS: Evapor-ation | Seep-age | WATER BALANCE MODEL: $T_Q$ | Live Storage | Water level | Volume (live+ dead storage) | # days open | #days to closure | Observed Mouth state | | EUTROPHICATION MODEL: $T_R$ | $T_R/T_B$ | black water accumulation | GWI | $E_I$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | [$m^3/s$] | [$m^3/s$] | [$m^3/s$] | [$m^3/s$] | [$m^3/s$] | [$m^3/s$] | [$m^3/s$] | [days] | [$m^3$] | [mm] | [$m^3$] | | | | | | | [$m^3$] | [%] | [%] |
| 24-Jan-06 | 0.081 | 0.165 | 0.075 | 0.240 | 0.320 | 0.004 | 0.074 | 29 | 701222 | 3102 | 1001222 | 0 | | 1 | closed | 38 | 2.71 | 805092 | 80% | 80% |
| 25-Jan-06 | 0.074 | 0.165 | 0.075 | 0.240 | 0.314 | 0.005 | 0.083 | 30 | 722182 | 3124 | 1022182 | 0 | | 1 | closed | 39 | 2.79 | 825820 | 81% | 81% |
| 26-Jan-06 | 0.079 | 0.165 | 0.075 | 0.240 | 0.319 | 0.006 | 0.091 | 29 | 741632 | 3143 | 1041632 | 0 | | 1 | closed | 40 | 2.86 | 846548 | 81% | 81% |
| 27-Jan-06 | 0.099 | 0.165 | 0.075 | 0.240 | 0.339 | 0.007 | 0.098 | 27 | 760742 | 3162 | 1060742 | 0 | | 1 | closed | 41 | 2.93 | 867276 | 82% | 82% |
| 28-Jan-06 | 0.087 | 0.165 | 0.075 | 0.240 | 0.327 | 0.008 | 0.105 | 28 | 780988 | 3182 | 1080988 | 0 | | 1 | closed | 42 | 3 | 888004 | 82% | 82% |
| 29-Jan-06 | 0.075 | 0.165 | 0.075 | 0.240 | 0.315 | 0.009 | 0.110 | 29 | 799483 | 3200 | 1099483 | 0 | | 1 | closed | 43 | 3.07 | 908732 | 83% | 83% |
| 30-Jan-06 | 0.072 | 0.165 | 0.075 | 0.240 | 0.312 | 0.009 | 0.115 | 30 | 800000 | 3200 | 1100000 | 0 | | 1 | closed | 44 | 3.14 | 929460 | 84% | 84% |
| 31-Jan-06 | 0.071 | 0.165 | 0.075 | 0.240 | 0.310 | 0.010 | 0.119 | 30 | 0 | 930 | 300000 | 1 | 9 | 0 | open | 0 | 0 | 20728 | 7% | 0% |
| 01-Feb-06 | 0.073 | 0.152 | 0.088 | 0.239 | 0.312 | 0.011 | 0.124 | 30 | 0 | 930 | 300000 | 2 | 8 | 0 | open | 0 | 0 | 20686 | 7% | 0% |
| 02-Feb-06 | 0.070 | 0.152 | 0.088 | 0.239 | 0.309 | 0.011 | 0.127 | 30 | 0 | 930 | 300000 | 3 | 7 | 0 | open | 0 | 0 | 20686 | 7% | 0% |
| 03-Feb-06 | 0.068 | 0.152 | 0.088 | 0.239 | 0.308 | 0.012 | 0.131 | 30 | 0 | 930 | 300000 | 4 | 6 | 0 | open | 0 | 0 | 20686 | 7% | 0% |
| 04-Feb-06 | 0.071 | 0.152 | 0.088 | 0.239 | 0.310 | 0.012 | 0.134 | 30 | 0 | 930 | 300000 | 5 | 5 | 0 | open | 0 | 0 | 20686 | 7% | 0% |
| 05-Feb-06 | 0.070 | 0.152 | 0.088 | 0.239 | 0.309 | 0.013 | 0.138 | 30 | 0 | 930 | 300000 | 6 | 4 | 0 | open | 0 | 0 | 20686 | 7% | 0% |
| 06-Feb-06 | 0.077 | 0.152 | 0.088 | 0.239 | 0.317 | 0.013 | 0.141 | 29 | 0 | 930 | 300000 | 7 | 3 | 0 | open | 0 | 0 | 20686 | 7% | 0% |
| 07-Feb-06 | 0.079 | 0.152 | 0.088 | 0.239 | 0.319 | 0.013 | 0.144 | 29 | 0 | 930 | 300000 | 8 | 2 | 0 | open | 0 | 0 | 20686 | 7% | 0% |
| 08-Feb-06 | 0.076 | 0.152 | 0.088 | 0.239 | 0.316 | 0.014 | 0.147 | 29 | 0 | 930 | 300000 | 9 | 1 | 0 | open | 0 | 0 | 20686 | 7% | 0% |
| 09-Feb-06 | 0.079 | 0.152 | 0.088 | 0.239 | 0.319 | 0.000 | 0.019 | 29 | 13393.5 | 1511 | 313393 | 0 | | 1 | closed | 1 | 0.07 | 41372 | 13% | 1% |
| 10-Feb-06 | 0.083 | 0.152 | 0.088 | 0.239 | 0.322 | 0.000 | 0.019 | 29 | 39236.4 | 1761 | 339236 | 0 | | 1 | closed | 2 | 0.14 | 62058 | 18% | 3% |
| 11-Feb-06 | 0.076 | 0.152 | 0.088 | 0.239 | 0.316 | 0.000 | 0.019 | 29 | 65406.7 | 1915 | 365407 | 0 | | 1 | closed | 3 | 0.21 | 82744 | 23% | 5% |

=IF(O30=1,Q29+1,0)

=T30*(IF(R30>1,1,R30)

=IF(O30=0,E30*24*3600,(E30*24*3600)+S29)

**Figure A.1 Section of the spreadsheet of the water balance and eutrophication models.**

## A2: Flow charts used for the models

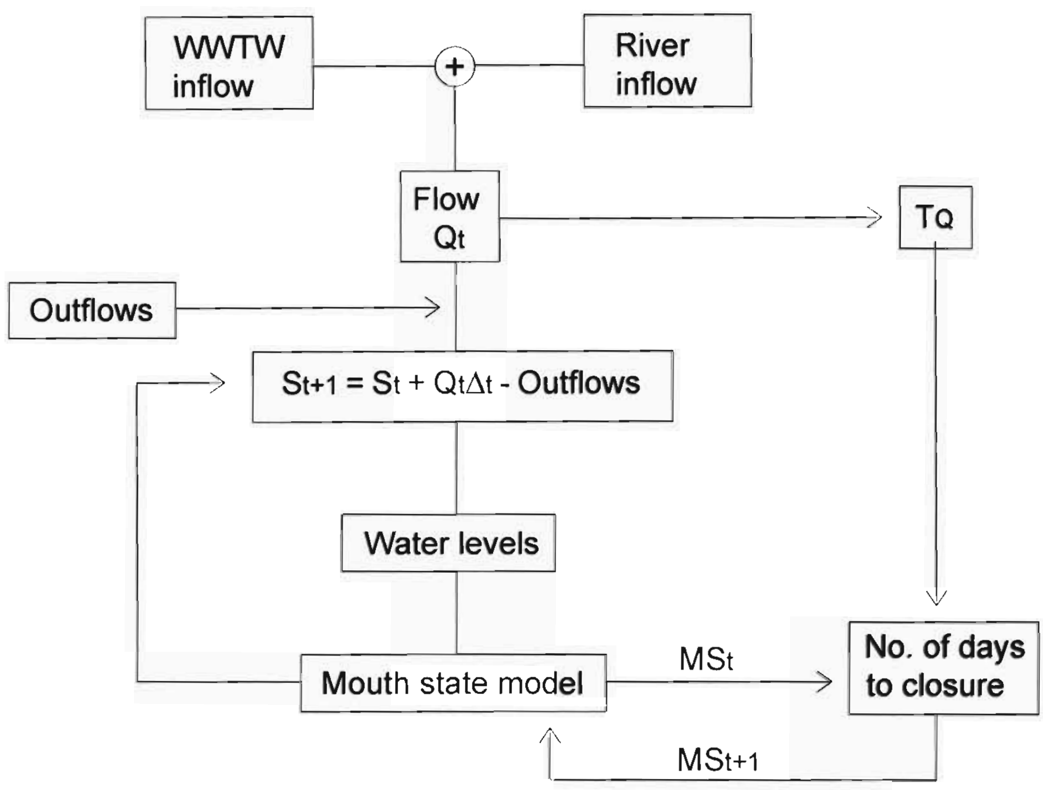


**Figure A.2 Water balance model**

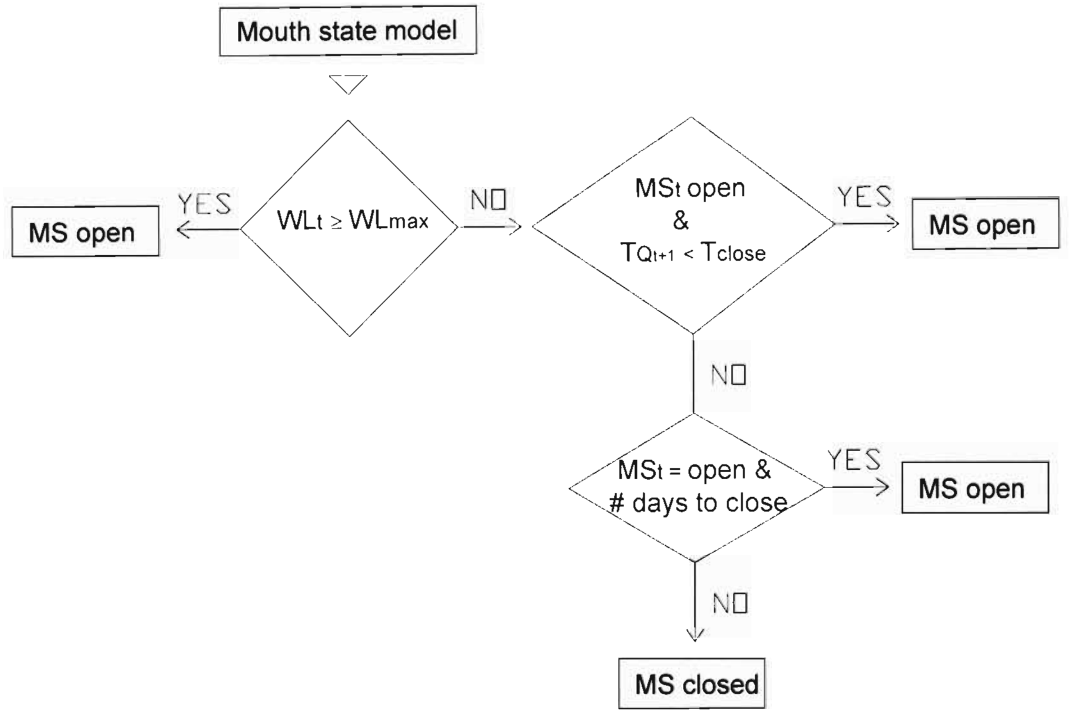


**Figure A.3 Mouth state model**

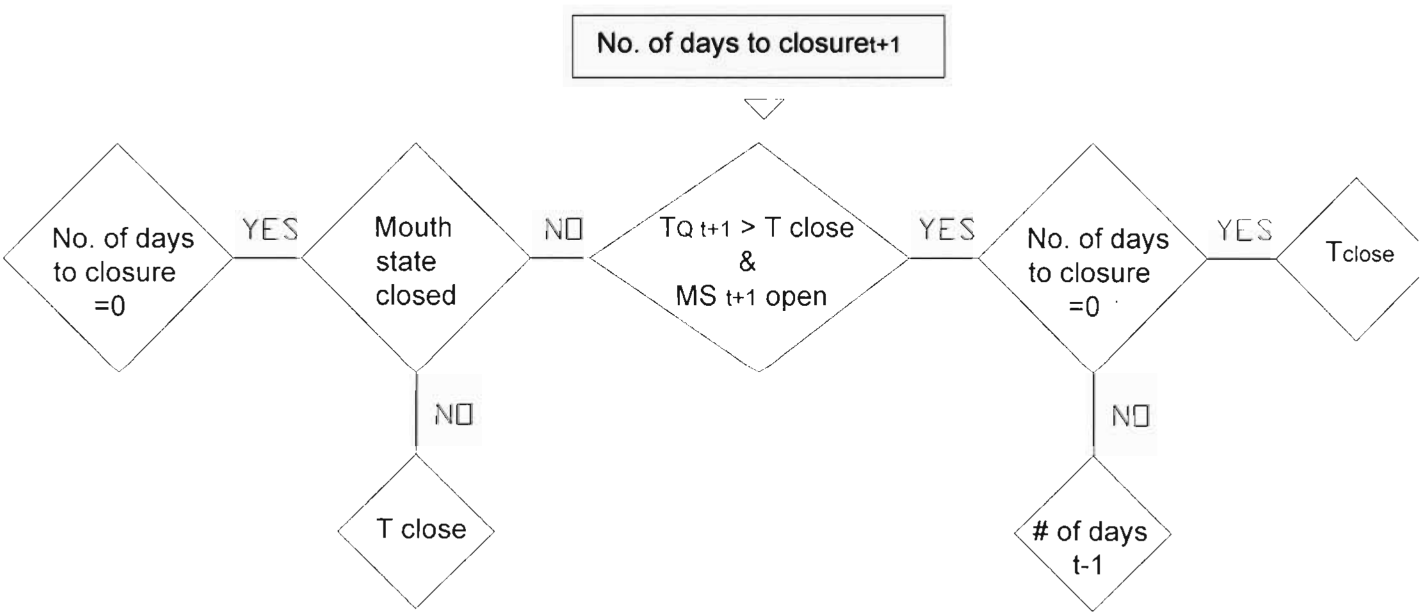


**Figure A.3 Number of days to closure model**

# APPENDIX B

## PARAMETERS USED IN FIELD MEASUREMENTS

## B1: Cross sections of the Mhlanga Estuary used to estimate river flows.

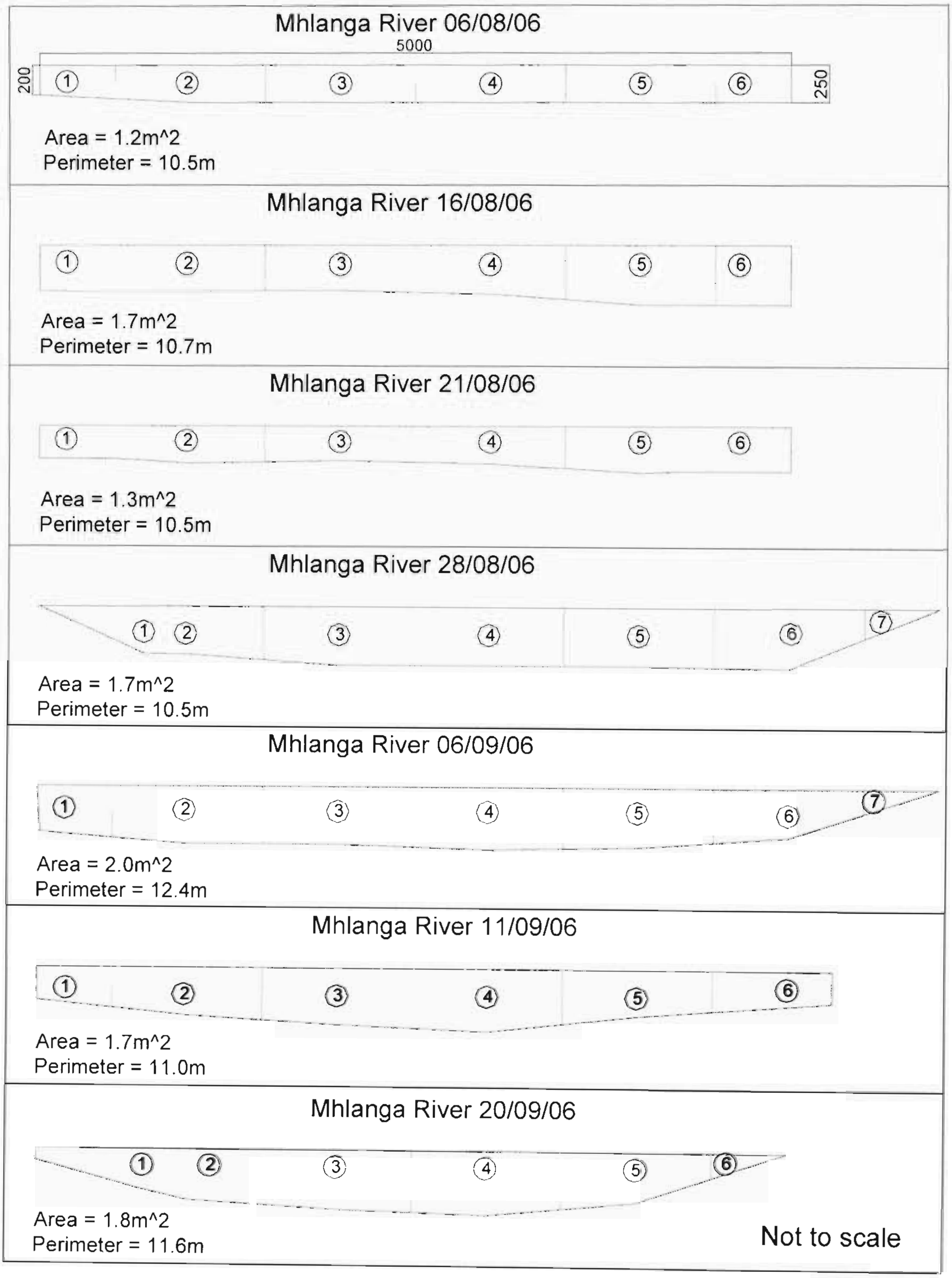


**Figure B.1** **Cross sections of the Mhlanga River at each sampling event.**

## B2: Water level sensor calibration

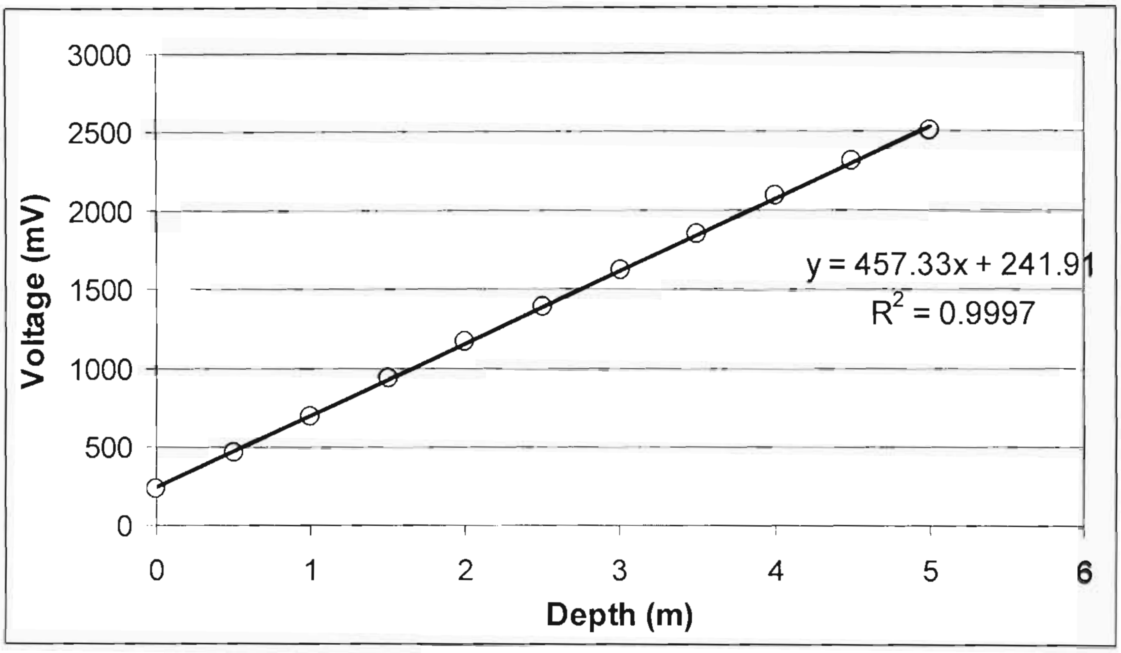


**Figure B.2 Calibration curve for water level sensor of voltage vs. depth.**

# APPENDIX C

PHYSICO-CHEMICAL PARAMETERS

Perissinotto *et al.* (2004)

Begg (1984)

**C1: Minimum and maximum physico-chemical parameters found in previous studies of the Mhlanga Estuary (UR = upper reaches, MR = middle reaches and LR = lower reaches).**

<table>
<tr><th rowspan="3"></th><th colspan="4" rowspan="2">(Perissinotto et al., 2004)</th><th colspan="3">Begg (1984)</th></tr>
<tr><th>Summer</th><th colspan="2">Winter, 1964</th></tr>
<tr><th colspan="2">Minimum</th><th colspan="2">Maximum</th><th></th><th>dry conditions</th><th>after heavy rains</th></tr>
<tr><td rowspan="2">DIN [mg/L]</td><td>0.2</td><td>(Aug ' 02, MR, open)</td><td>5.8</td><td>(July ' 02, UR, open)</td><td rowspan="2">0.73</td><td rowspan="2">0.71</td><td rowspan="2">1.04</td></tr>
<tr><td>0.6</td><td>(Oct ' 02, LR, closed)</td><td>5.1</td><td>(March ' 02, MR, closed)</td></tr>
<tr><td rowspan="2">DIP [mg/L]</td><td>0.2</td><td>(Jan ' 03, LR, open)</td><td>2.5</td><td>(July ' 02, LR, open)</td><td rowspan="2">0.12</td><td rowspan="2">0.3</td><td rowspan="2">0.23</td></tr>
<tr><td>0.4</td><td>(Sept ' 02, UR, closed)</td><td>2.3</td><td>(March ' 02, MR, closed)</td></tr>
<tr><td>$K_d$ [$m^{-1}$]</td><td>14</td><td>(Nov ' 02, MR, open)</td><td>0.21</td><td>(Jan ' 03, MR, open)</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Salinity [‰]</td><td>0.1</td><td>(Jan ' 03, UR, open)</td><td>32.0</td><td>(May ' 02, LR, open)</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Temperature [°C]</td><td>13.7</td><td>(July' 02, UR, surface)</td><td>30.0</td><td>(Feb ' 03, MR, bottom)</td><td>25</td><td>13.8</td><td>13.5</td></tr>
<tr><td>Dissolved oxygen [mg/L]</td><td>0.08</td><td>(April ' 02, UR, bottom, open)</td><td>12.20</td><td>(April ' 02, MR, surface)</td><td>3.8</td><td>9.3</td><td>8.9</td></tr>
<tr><td colspan="5">Average DIN:DIP ratio = 6.84</td><td>6.08</td><td>2.37</td><td>4.52</td></tr>
</table>

## C2: Physico-chemical parameters of the Mhlanga Estuary during the study period, July to September 2006.

| Date | Mouth State | YSI 6920 Water Logger Probe | | | | | | | | | | | | | | Light meter | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Temperature [degC] | | Conductivity [mS/cm] | | Salinity [ppt] | | DOsat [%] | | DO [mg/L] | | pH | | Turbidity [NTU] | | $K_d$ [$m^{-1}$] | Euphotic depth [m] |
| | | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 | | $4.6/K_d$ |
| 07/17/06 | closed | 17.6 | 16.7 | 1.6 | 1.3 | 0.9 | 0.8 | 120 | 77 | 11.4 | 7.5 | 9.0 | 8.3 | 1.3 | 1.8 | 0.49 | 9.45 |
| 07/24/06 | open | | | 6.3 | 6.4 | 3.7 | 3.7 | 83 | 84 | 7.1 | 7.2 | 8.2 | 8.1 | 9.3 | 11.0 | 14.55 | 0.32 |
| 07/31/06 | open | 17.2 | 17.6 | 13.3 | 34.4 | 9.2 | 25.7 | 149 | 70 | 13.6 | 5.7 | 8.0 | 7.8 | 0.7 | 6.0 | 1.33 | 3.46 |
| 08/07/06 | closed | 19.5 | 16.9 | 4.1 | 15.2 | 2.5 | 10.7 | 122 | 84 | 11.1 | 7.6 | 8.0 | 7.5 | 5.3 | 1.7 | 0.46 | 10.08 |
| 08/16/06 | closed | 18.2 | 18.6 | 3.7 | 7.9 | 2.3 | 4.9 | 83 | 85 | 7.7 | 7.5 | 8.2 | 8.0 | 2.5 | 4.6 | 0.39 | 11.75 |
| 08/21/06 | open | 20.1 | 20.9 | 3.6 | 3.8 | 2.2 | 2.2 | 106 | 106 | 9.3 | 9.7 | 8.1 | 8.1 | 10.1 | 9.3 | 15.28 | 0.30 |
| 08/28/06 | open | 19.2 | 19.2 | 10.7 | 30.2 | 6.9 | 21.3 | 114 | 82 | 10.1 | 6.7 | 8.4 | 7.9 | 14.1 | 8.1 | equipment damaged | |
| 09/04/06 | closed | 20.3 | 20.3 | 6.8 | 7.2 | 4.1 | 4.4 | 147 | 195 | 12.9 | 17.1 | 8.8 | 8.6 | 3.3 | 3.3 | 0.71 | 6.43 |
| 09/11/06 | closed | 22.8 | 22.4 | 5.2 | 8.5 | 2.9 | 5.1 | 122 | 131 | 10.4 | 11.1 | 8.0 | 8.2 | | 2.2 | 0.99 | 4.64 |
| 09/25/06 | closed | 18.6 | 18.7 | 10.4 | 13.9 | 6.7 | 9.3 | 65 | 61 | 5.8 | 5.3 | 7.9 | 7.8 | 7.1 | 6.3 | 0.49 | 9.45 |