WRC PROJECT K5/1679/0/1

INTEGRATED ECOLOGICAL-ECONOMIC MODELING AS AN ESTUARINE MANAGEMENT TOOL: A CASE STUDY OF THE EAST KLEINEMONDE ESTUARY

MODEL REPORT

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March 2008

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1 INTRODUCTION

1.1 Background and rationale

The economic values derived from estuaries are determined by its delivery of goods and services and its attributes, all of which are a function of ecosystem health. Estuarine health is, in turn, strongly influenced by the quantity and quality of freshwater inflows entering the system, as well as by management of the estuary.

At present, expert opinion is used in a process called Resource Directed Measures (RDM) to predict changes in the physical and biotic characteristics estuaries under different flow scenarios in order to inform decisions regarding the freshwater reserve. This is based on varying amounts of data, depending on whether the process followed is rapid, intermediate or comprehensive. The predictions are made by specialists based on existing understanding of the way in which the biophysical components of the system interact. However, the rationale used as the basis of these predictions is seldom quantified or made explicit. Thus the results generated may not always be repeatable by different experts.

Similarly, management decisions, such as if and when to breach an estuary artificially, are also largely based on gut feel, which is in turn based on the experience of managers and the information available to them. Invariably, management decisions must rely on certain assumptions which are generally not made explicit.

Modelling the interactions between different estuary components forces scientists to be explicit about the relationships that are assumed in predicting the consequences of different water flow or management scenarios. Quantifying the assumptions behind reserve and management decisions is critical to furthering our understanding of the way in which estuaries respond to changes. At the very least, it helps to set up testable hypotheses in cases where information is imperfect.

The classification process by which the future health of estuaries will be decided, and the reserve determined, will also require estimates of the economic implications of alternative flow scenarios. This involves estimating the relationships between estuarine health and attributes and a suite of economic values, using available data and assumptions that again should be made explicit. Systems-modelling provides an excellent platform for the integration of socio-economic variables that are relevant to the decision process. Understanding such relationships will not only inform decisions about water allocation, but about management options within particular systems.

1.2 Choice of estuary type and study area

South Africa has some 279 estuaries, of which 185 are classified as temporarily open/closed estuaries, or intermittently open estuaries (IOEs). While most research has taken place on the larger and permanently open systems, relatively little research has focused on IOEs in the past. These systems are particularly interesting from a management perspective because they are understood to be fairly sensitive to water inflows and other management issues which affect their mouth condition and therefore their biota.

The 35.7 ha East Kleinemonde Estuary provides the ideal IOE study area for the development of the model because of the unusually high amount of data available following a recent three-year WRC research programme (K5/1581). The programme included intensive data collection and analysis of all major physical and biotic components over a 19 month period, and culminated in the carrying out of an RDM study on the estuary in which the participating scientists predicted how the different components would change under different flow scenarios.

1.3 Purpose of the model

The aims of the modelling study were to:

- Improve synergy among disciplines in approach to RDM, management;
- Highlight assumptions and knowledge gaps in predicting changes;
- Estimate impacts of changes in flow on output of estuary goods and services;
- Develop an integrated ecological-economic model of the East Kleinemonde Estuary;
- Estimate ecological and economic implications of alternative scenarios; and
- Evaluate the model as a tool for classification and estuary management.

1.4 Scope of this report

This report outlines the development of the model of the East Kleinemonde estuary, the data and assumptions used in the model, and the outputs obtained. The detailed structure and operation of the model is described in the accompanying User Manual. In the final phase of the study, this draft report will be expanded to include a discussion on the use of the model for RDM and for management, and its applicability for use on other estuaries.

2 APPROACH

2.1 Overall approach

This study was a collaborative effort which included most of the scientists that had been involved in the recent 19-month field study of a number of aspects of the East Kleinemonde Estuary (WRC Project K5/1581). The specialists provided the necessary inputs in a series of workshops during which much of the initial construction of the model took place. Construction of the model relied on data collected during and prior to the WRC research programme, information on similar systems from the literature, professional opinion and on-the-spot analysis of data, as required. The team attempted to reduce the complexity of the various components by concentrating on key drivers and relationships, and in the absence of hard data, to articulate the assumptions they made about the influence of those drivers on their particular component.

The model was built using an initial simulation period which corresponded to the main period of intensive research on the East Kleinemonde Estuary. This was a 19 month period from 7 March 2005 to 4 October 2006. This period was chosen because of the fact that daily water level observations in the estuary, mouth condition (open or closed) and biological data were available for this period and could be used to validate model simulations. More detailed data on mouth condition (open, closed, overwash from the sea) and salinity were also available for part of this period (21 January – 14 November 2006). However, no flow data were available for this period. Thus the model was build using rough estimates of flow, and concentrated on achieving similar physical responses to those recorded so that the biotic components could be modelled and calibrated over the same period.

Following this the model was adapted to run using simulated flow data which was available for the period 1933 – 2002 (calculated from daily rainfall using the Pitman Model – Hughes 2006). The physical sector of the model was refined using data for the 8-year period from 1 January 1993 to 31 December 2000. Data on mouth condition (open or closed) were available for this period and were used to validate physical outputs over this simulation period. The longer time period also allowed further refinement of the biotic components of the model.

2.2 Phase I. Conceptualisation and initial modelling

The structure of the first modelling workshop is outlined in Table 2.1. The outcome of this workshop was a conceptual model, proto-models of different abiotic and biotic components, and a list of data needs to be addressed before the next workshop.

Table 2.1 Phase I. Activities and participants of the first modelling workshop

Date (2006)	Activity	Participants
Sat 8 April	Introduction to STELLA	Jane Turpie, Barry Clark, Lara van Niekerk, Guy Bate, Phumelele Gama, Janine Adams, Tommy Borman, Taryn Ridden, Paul Cowley, Steve Lamberth, Anja Terorde
Sun 9 April	Develop a conceptual model of EK (Plenary and group work)	
Mon 10 – Tues 11 April	Build a proto-STELLA model of EK (Plenary and group work)	
Wed 12 April	Evaluate data needs	

2.3 Phase II. Construction of the 19 month-model

Following some preparation by the core team, a series of modelling workshops was then held in which the component sectors of the model were built in sequence (Table 2.2).

Table 2.2 Phase II. Activities and participants of the follow-up series of modelling workshops

Date	Activity	Participants
21 – 23 February 2007	Physical dynamics	Barry Clark, Lara van Niekerk, Jane Turpie
1 – 2 March 2007	Water quality	Susan Taljaard, Barry Clark
5 – 9 March 2007	Micro-algae, macrophytes and invertebrates	Guy Bate, Phumelele Gama, Janine Adams, Tommy Borman, Anusha Rajkaran, Jane Turpie, Barry Clark
12 – 16 March 2007	Fish, birds	Paul Cowley, Steve Lamberth, Anja Terorde, Jane Turpie, Barry Clark
17 March 2007 – 20 Jan 2008	All sectors	Barry Clark, Jane Turpie

The outcome of the second workshop series culminated in a working ecological model of the East Kleinemonde Estuary for the 19-month period. This was further refined during the period up to January 2008.

2.4 Phase III. Construction of the long-term model

Economic data were collected and analysed during March-April 2007 (Turpie et al. 2007). The ecological model was refined using modelled inflow data and run over an 8-year period, allowing further refinement of the biotic parameters. A sector was added to calculate the outputs required to produce scores required by the Resource Directed Measures (RDM) process for scoring estuary health. A Management sector was added to explore the implications of artificial breaching. An economic sector was added to estimate the economic implications of modelled scenarios.

Table 2.3 Phase III. Activities and participants of the final modelling phase

Date	Activity/sector	Participants
20 Jan – 26 Mar 2008	Refinement of all ecological sectors using modelled inflow data and addition of non-ecological sectors	Jane Turpie, Barry Clark
Thurs 27 March 2008 (feedback workshop)	Overall presentation	Jane Turpie, Barry Clark,
	Physical and water quality	Lara van Niekerk, Gavin Snow,
	Microalgae	Guy Bate, Phumelele Gama
	Macroalgae	Janine Adams, Tommy Borman
	Invertebrates	Tris Wooldridge, Ticky Forbes, Nicky Demetriades
	Fish	Alan Whitfield, Nadine Strydom, , Paul Cowley
	Birds	Anja Terorde

The long-term model and results were presented to the specialist team during March 2008, during which feedback on the different components of the model was obtained.

3 CONCEPTUAL MODEL OF THE ESTUARY

One of the outputs of the first modelling workshop was a conceptual model of the East Kleinemonde Estuary (Figure 3.1). The main purpose of this exercise was to think the problem through before getting bogged down in STELLA, to define parameters and their units of measurement, and to guide the design of the model in STELLA.

The conceptual model shows the group's understanding of the main drivers of each component of the ecosystem. The parameters that influence one another are linked by arrows and the expected nature of the relationship is indicated by the shape of the hypothetical response curve on each arrow. The parameter from which the arrow comes is on the X-axis, while the parameter being influenced is on the Y-axis. States 1, 2 and 3 indicated on the diagram refer to mouth states 1 = closed, 2 = marine overwashing and 3 = open.

According to this understanding, the mouth condition of the estuary is affected by a combination of freshwater and marine inflows and the building up of the berm which is influenced by sediment availability and wave energy. The resulting balance between freshwater and marine inputs affects water levels, salinity, nutrient levels, water residence time, current speed and turbidity, all of which affect the productivity and community structure of the estuary. These conditions determine the balance between different types of vegetation communities in the estuary, as well as the production of phytoplankton. Food and habitat structure, coupled with the physical characteristics for which different organisms vary in their tolerance, determine the abundance and structure of the invertebrate, fish and bird communities.

This conceptual model formed the basis for the construction of the STELLA model as well as for the collation of relevant data required for the modelling process.

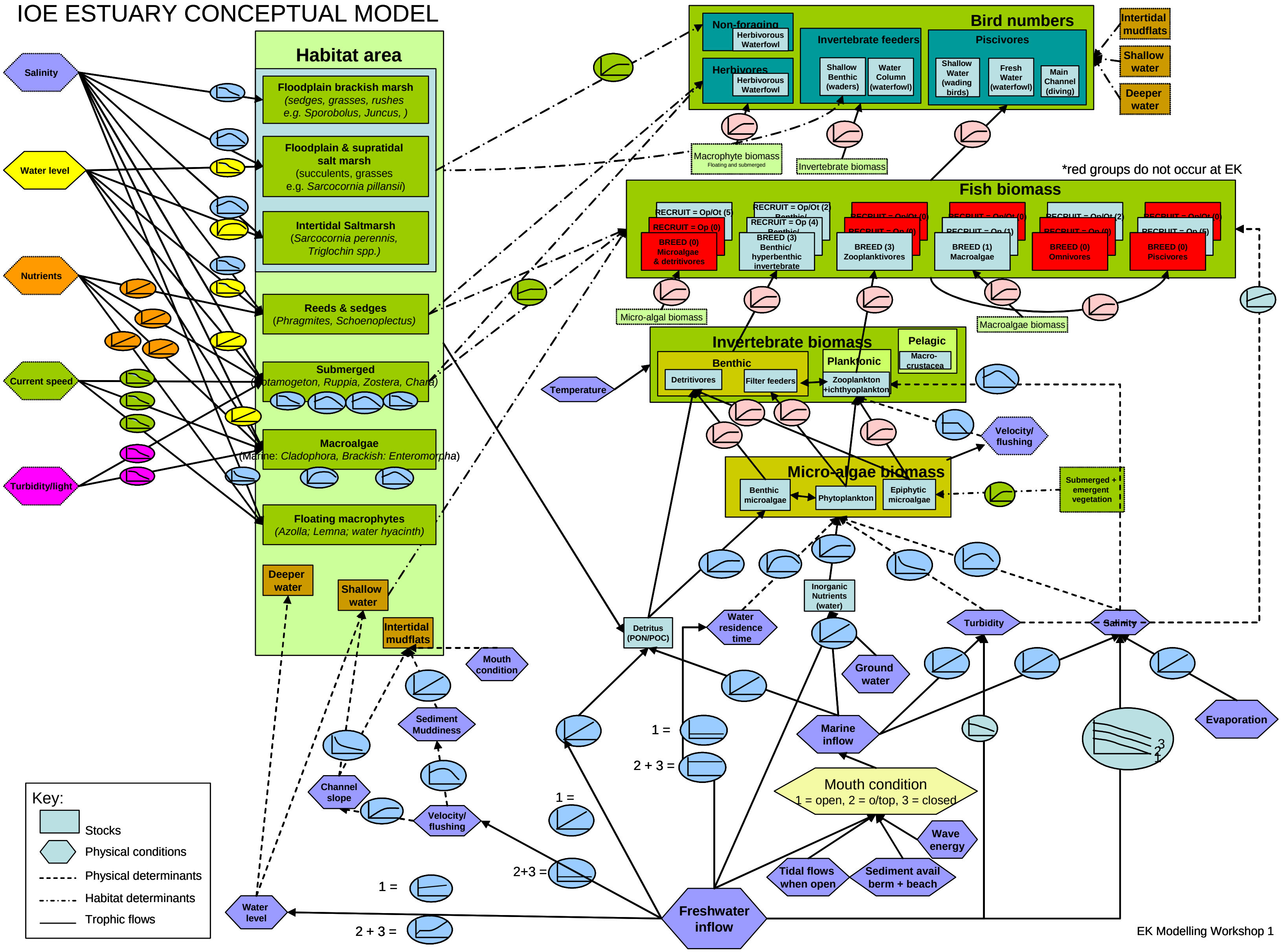

Figure 3.1. Conceptual model of the East Kleinemonde Estuary

4 DATA, ASSUMPTIONS AND DEFINITION OF MODEL PARAMETERS

4.1 Introduction

The aim of the model was to show how the physical and biotic components of the estuary change in response to a change in freshwater inflow into the estuary or other management interventions. From a physical perspective, the model had to simulate the changes in mouth condition, water level and salinity in response to freshwater inflow, taking other physical processes such as tidal cycles into account. This was then translated into the effects on broad habitat types and the primary productivity of the system. The model then had to estimate changes in faunal groups ranging from zooplankton and other invertebrates to fish and birds. Finally the model outputs had to be translated into measures useful for assessing estuarine health and economic value.

The model was constructed as a series of sub-models (called ‘sectors’ in STELLA), each of which dealt with a different physical, biotic or management aspect of the system. The data, understanding and assumptions used in the construction of each of these sub-models is described below.

4.2 Physical dynamics, salinity and nutrients

4.2.1 Sub-model overview

The aim of the physical model was to simulate as closely as possible changes in physical (hydrodynamic) and water quality parameters in the estuary using a daily time-step. Input parameters for the model included daily freshwater inflow (derived either from water level in the estuary or from disaggregated monthly flow data), daily tidal height maxima, and daily mean wave height data. Key hydrodynamic parameters we simulated included changes in water level in the estuary, total water volume in the system, and the height of the berm across the mouth of the estuary. Key water quality parameters simulated included average salinity and average concentration of dissolved inorganic nitrogen (DIN) in the system. A brief description of the nature and source of each of the input and output parameters for the model are described below.

4.2.2 Freshwater inflow

Daily freshwater inflows into the East Kleinemonde Estuary were derived from different sources for the two simulation periods. In the case of the initial 18 month simulation period (7 March 2005 to 4 October 2006), data on freshwater inputs into the estuary were back calculated from daily observations on water level in the estuary. Using a bathymetry model of the estuary (Bornman 2007, Figure 4.1), the relationship between depth (water level) and volume was derived (Figure 4.2). It was assumed that any change in water depth in the estuary was attributable to freshwater inflow except when the mouth of the estuary was open and tidal influences were assumed to predominate. During the period when the mouth was open, freshwater inflows were held constant for a short period (2-3 days) and then dropped to zero. A major shortcoming of this approach was that inputs from the sea in the form of overwash that affected water level in the estuary were also counted as freshwater inputs, which in turn created anomalous trends in salinity within the system (i.e. salinity dropped when it should actually have increased). This was unfortunately unavoidable as rainfall-derived freshwater inflow data were not available for this period.

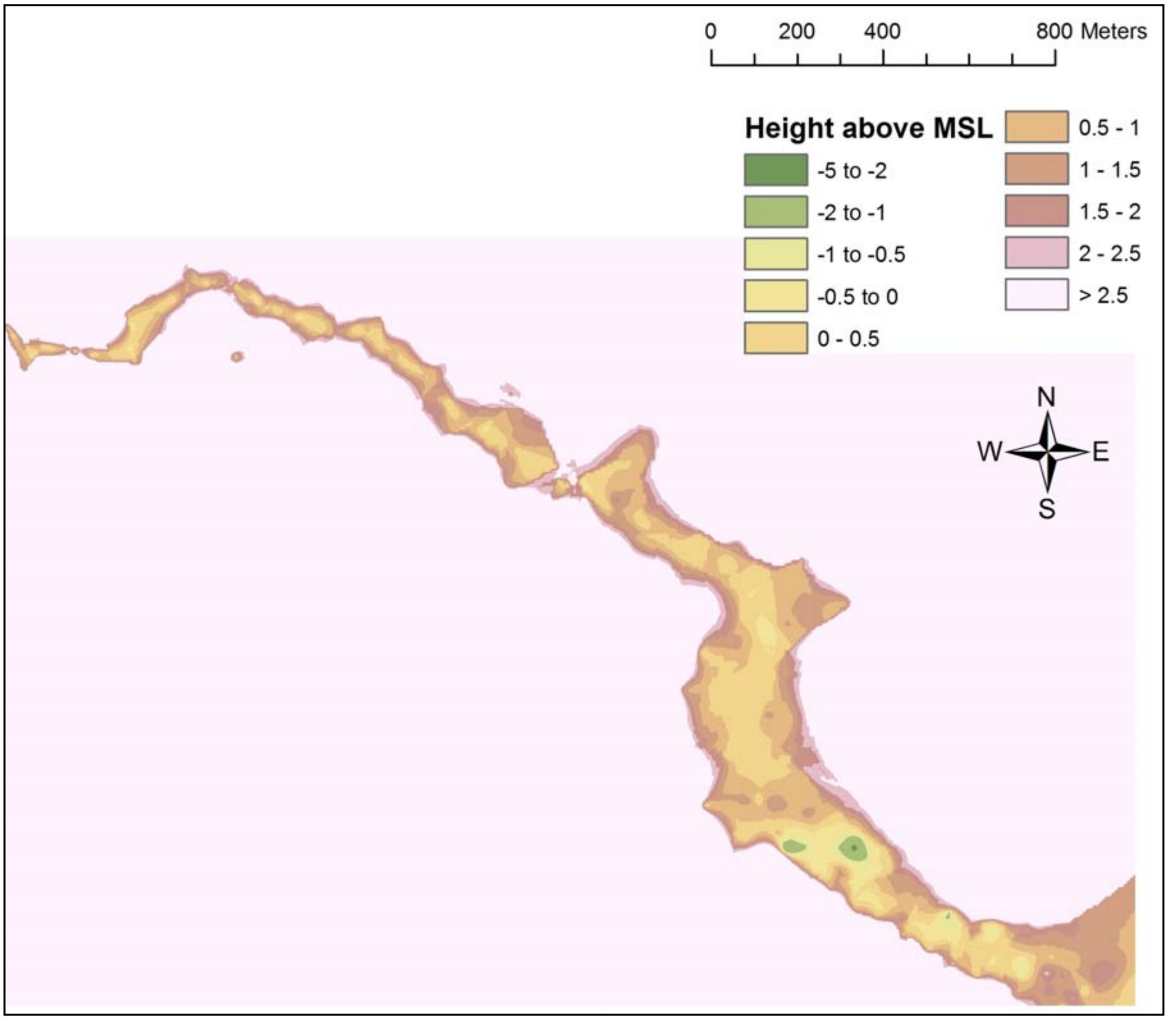

Figure 4.1. Bathymetry of the East Kleinemonde Estuary from which depth-volume and volume-surface area relationships were derived (Source: Bornman 2007).

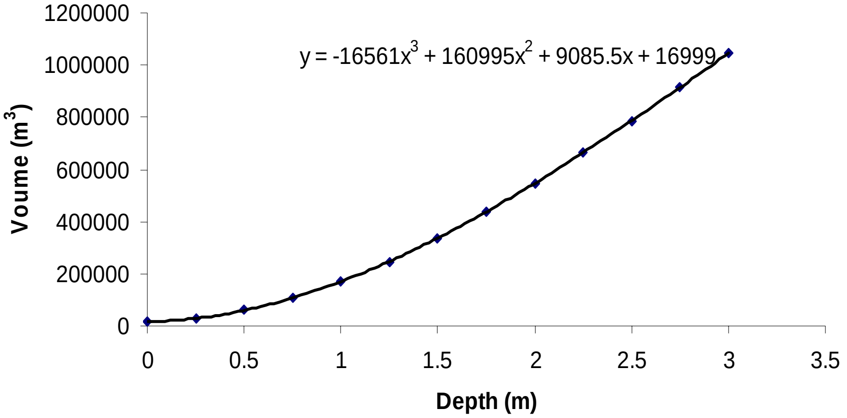

Figure 4.2. Relationship between water depth and volume in the East Kleinemonde Estuary derived from the bathymetry model prepared by Bornman (2007).

4.2.3 Water level/volume

Water volume in the estuary was treated as a stock, fed by freshwater and marine inflows into the estuary, and was drained by outflows through the mouth during a breaching event and evaporation. Daily change in water volume in the estuary was derived by summing daily flows into the system, subtracting net outflows, and adding the residual to the existing volume. Water level in turn was derived using the bathymetry model developed by Bornman (2007) and the inverse of the relationship displayed in Figure 4.2 (Figure 4.3).

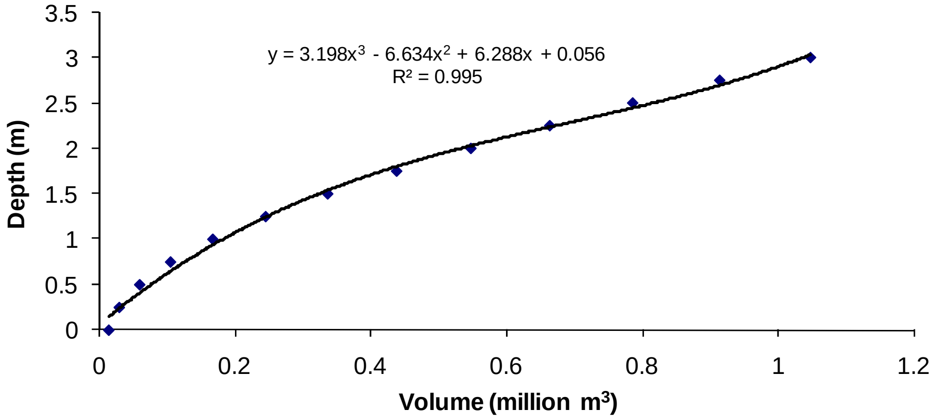

Figure 4.3. Relationship between volume water and depth in the East Kleinemonde Estuary derived from the bathymetry model prepared by Bornman (2007).

4.2.4 Sea water inflows

It was assumed that seawater inflow into the East Kleinemonde Estuary is derived from two sources - direct inflow through the mouth when it is open and over the top of the berm (termed overwash) during periods when the mouth is closed but height of the sea exceeds the height of the berm. Observations on mouth state for the East Kleinemonde Estuary were available for a ten year period - 1 March 2003 to 31 December 2006. Recorded mouth states included: open, closed, small overwash, large overwash, trickling out. These data clearly show that periods of overwash (both small and large) coincided with high sea levels (due to tidal influences) and/or when mean wave height was high.

In the model, seawater inflows into the Kleinemonde estuary were simulated by identifying periods of time when the sea level exceeded the berm height and multiplying this by the estimated width and height of the window through which these flows would enter the system and by the estimated velocity of these flows. These flows were applied only when the water level in the system was lower than the berm height (i.e. such flows would then be trapped in the system and not simply flow out again) and when daily maximum tide height (taken as measured maximum tidal heights for the Port of East London supplied by the South African Navy Hydrographic Office corrected to MSL) increased by a factor of half the measured wave height on that day (from wave rider buoy data supplied by the CSIR), exceeded the berm height on that day. The time period over which seawater was able to enter the system was calculated as the period for which the overtopping height (maximum tidal height plus 0.5 times wave height) exceeded the height of the berm at the mouth of the estuary. When the depth of the overwash is at its maximum (3.2 m) and berm is zero (i.e. overwash depth = swash height) then sea level would be higher than berm for 12 h (43 200 s) of each day. Overwash time was assumed to change with overwash depth in a non-linear fashion (Figure 4.4), declining to zero when overwash time was equal to zero. The size of the window through which seawater was able to enter the system was calculated as the mean channel width (10 m) multiplied by the difference between the average overtopping height (= overtopping height/2 since swash height varies from zero at the start of the time window up to the maximum value half way through the time window and declines to zero again thereafter) and the berm height. Velocity of water flowing into the estuary was allowed to vary with overwash depth, in accordance with the graphical function shown in Figure 4.5.

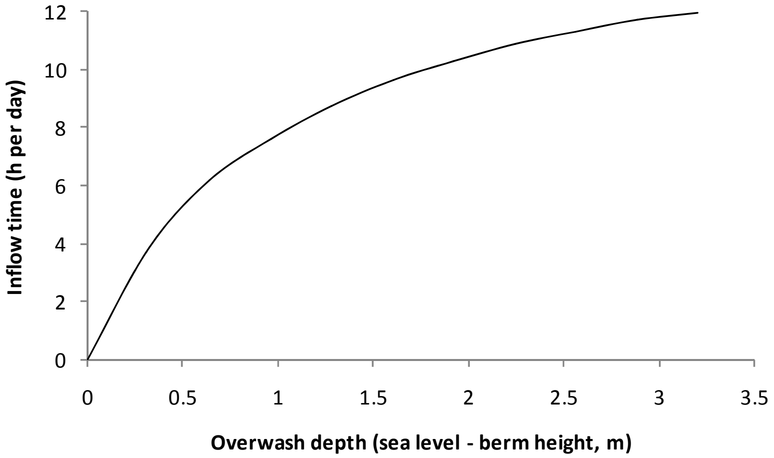

Figure 4.4. Total time per day over which it is possible for seawater to flow into the estuary as a function of the difference between maximum overwash depth (based on maximum tide height) and berm height.

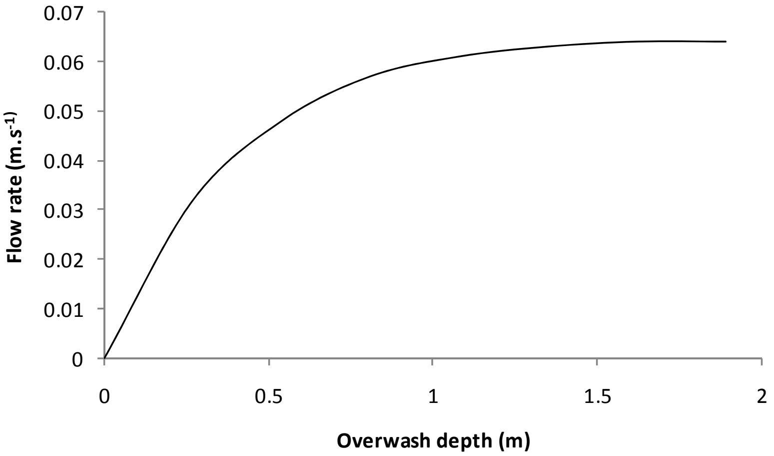

Figure 4.5. Variation in flow rate with overwash depth for seawater flowing into the estuary during overwash events.

4.2.5 Evaporation

Evaporative losses in the model were entered as monthly averages extrapolated from four months of measured evaporation rates on the East Kleinemonde (December, January, June and July;Figure 4.6). These evaporation rates were calculated for freshwater and thus needed to be adjusted in accordance with changing salinity in the estuary, in accordance with the relationship defined by Bargery *et al*. (2008; Figure 4.7)

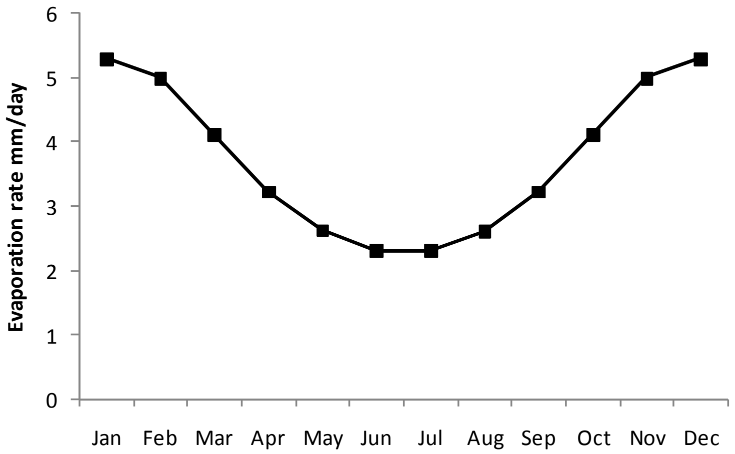

Figure 4.6. Monthly evaporation rates used in the model extrapolated from four months of measured data.

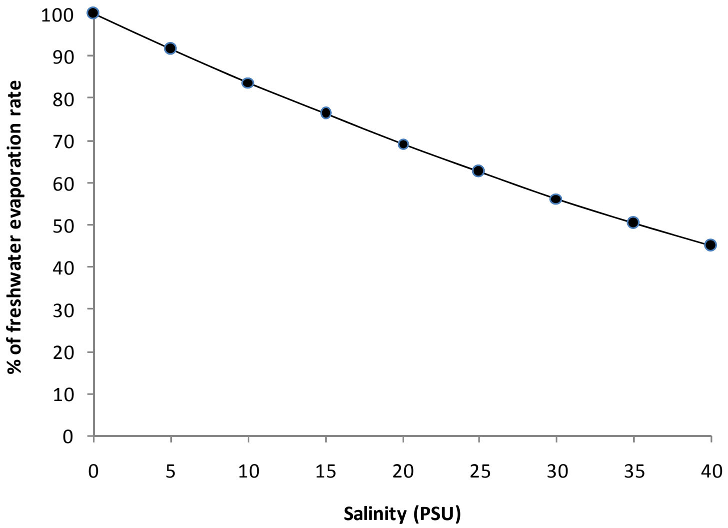

Figure 4.7. Change in evaporation rate with changing salinity as applied in the model (after Bargery et al. 2008).

4.2.6 Maximum sea level

The daily maximum sea level at the mouth of the Kleinemonde estuary was inputted to the model as maximum daily tide heights calculated from data supplied by the South African Navy Hydrographic Office (SANHO). These data were corrected to MSL using correction factor published by SANHO. Gaps in the data were patched using maximum daily tide heights calculated from predicted hourly tidal height data also supplied by SANHO or where such data were not available using data extracted from WX Tides 32 version 2.6, Jan 20, 2000 (www.Geocities.com/Sta iliconValley/Horizon/1195/) for East London. Predicted maximum daily tide heights supplied by SANHO and data from WX Tides were also corrected from chart datum to MSL using correction factor published by SANHO.

4.2.7 Berm height and breaching

The berm at the mouth of the estuary was modelled as a one-dimensional stock of sand (height only). Daily growth in berm height was assumed to occur at a rate determined by the prevailing offshore wave climate and the height of the berm, increasing with increasing wave height and decreasing as it approached the maximum observed berm height. These growth rate functions were derived from measured changes in berm height and wave height made by B. McKenzie, Rhodes University and Dr T. Bornman, Nelson Mandela Metropolitan University (Figure 4.8, Figure 4.9).

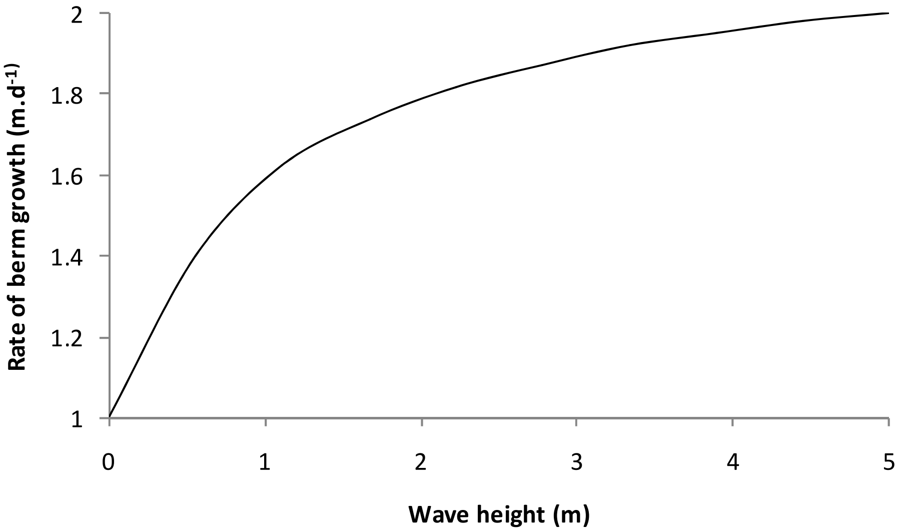

Figure 4.8. Rate of sand build up on the berm at the mouth as a function of wave height.

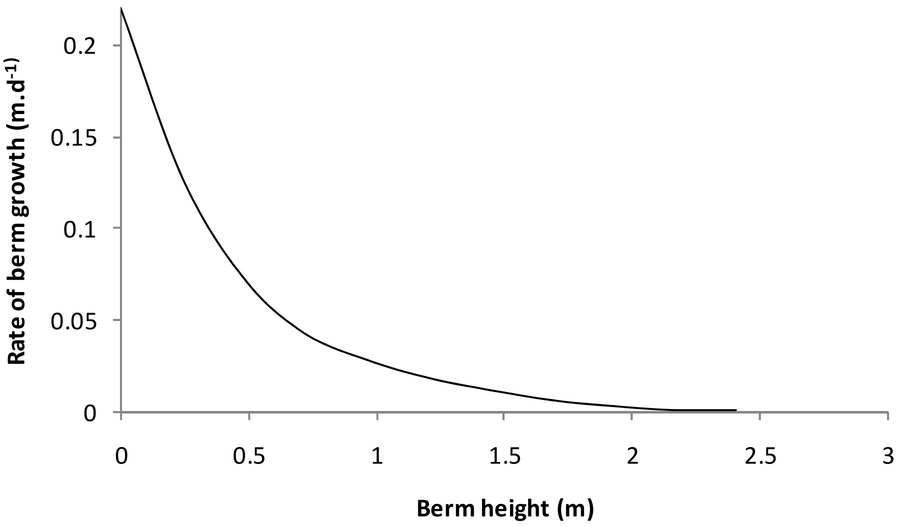

Figure 4.9. Rate of sand build up on the berm at the mouth as a function of berm height

Berm height was allowed to grow up to a maximum height of 2.4 m or until water level in the estuary exceeded the height of the berm (either as a function fresh or seawater inflow), at which point the berm was immediately reduced to a height of 0.5 m, in effect breaching the system. All water in the system was allowed to flow out at this point leaving behind that volume of water corresponding to a water depth of MSL + 0.5 m.

4.2.8 Wave height

Maximum daily wave height data were calculated from hourly wave height data obtained from the CSIR based on a wave-rider buoy off East London.

4.2.9 Salinity

Estimates of daily averaged salinity for the estuary was generated by the model by modeling the amount of salt in the system as a stock which was flowed into the system with seawater (either over the berm as tidal inflows or through overwash at a rate of 35 kg.l^{-1} of influent water) and out again when the mouth breached or as seepage through the berm at a rate corresponding to the prevailing salinity of the system and volume of water lost from the system during the event in question. Salinity at any point in time was calculated as the amount of salt in the system divided by the total volume of water in the system at that time.

4.2.10 Dissolved inorganic nitrogen (DIN)

Estimates of DIN concentration in the estuary were generated by modeling the amount of DIN in the system as a stock which was flowed into the system with seawater (at a rate of 214 µg.l^{-1} of influent water) and river inflow (at a rate of 2 500 µg.l^{-1} of influent water). Estimates of the concentration of DIN in seawater and freshwater were taken from values measured by Taljaard (2006) for the Kleinemonde estuary. DIN was removed from the system by flowing it out during a breaching event at a rate corresponding to the prevailing DIN concentration in the system and volume of water lost from the system during the event in question. Primary producers in the system were also allowed to consume DIN as they grew at a rate corresponding with their protein content (30% for microalgae and 10% for submerged macrophytes) and the nitrogen content of this protein (6.25%). Prevailing DIN concentration in the system at any point in time was calculated as the amount of DIN in the system divided by the total volume of water in the system at that time.

4.3 Microalgae

4.3.1 Sub-model overview

The aim of the microalgae sector was to simulate changes in the biomass of phytoplankton and benthic microalgae in the East Kleinemonde Estuary in response to changes in key physicochemical changes in the system. The microalgae sector was composed of three components – a phytoplankton model and two benthic microalgae models, one for epipellic microalgae (microalgae growing on sediments) and one for epilithic microalgae (microalgae growing on submerged aquatic macrophytes). Each of the models included a stock component (the biomass of algae modeled as chlorophyll a (Chl a) in kg), which was allowed to grow in response to the concentration of dissolved inorganic nitrogen (DIN) in the system and was lost from the system either through consumption (assumed to be mostly by invertebrates) or lost out of the mouth during a breaching event. Key input parameters for these models included DIN concentration, water volume in the estuary, surface area of submerged sand and mud, submerged macrophyte surface area,. Key outputs from this sector included DIN uptake rates for phytoplankton and benthic microalgae, phytoplankton biomass in the system, phytoplankton concentration, benthic microalgae biomass in the system, benthic microalgae biomass per unit area, and DIN production by blue-green algae in the system.

4.3.2 Phytoplankton

4.3.2.1 Phytoplankton biomass

Phytoplankton biomass was expressed either as total biomass of chlorophyll-a in the system in kg or as chlorophyll-a concentration in µg l^{-1}. The starting phytoplankton biomass was set at 3.5

µg l^{-1} corresponding to low biomass levels observed following a period of prolonged closure (Gama 2007).

4.3.2.2 Phytoplankton growth rate

Dissolved inorganic nitrogen concentration is considered to be the primary factor limiting phytoplankton growth in non-turbid estuaries (Adams *et al*. 1999). Growth rates of phytoplankton vary for different communities, with rates ranging from zero up to 3-3.6 doubling per day (Furnas 1990). Thus, we used a standard growth equation of the form:

*Growth = N*r*(1-N/K)*

Where N = the current population size (measured as µg Chla.l^{-1})
r = growth rate, and
K = the carrying capacity of the system.

Growth rate (r) was defined in relation to DIN concentration (Figure 4.10) and ranged from 0 to a maximum of 3.6 doublings per day. Carrying capacity (K) was defined in terms of DIN concentration, with maximum K corresponding to the maximum measured concentration of phytoplankton concentration in the East Kleinemonde Estuary from Goma (2007) multiplied by a factor of 1.25 (i.e. 25 µg Chla.l^{-1},). Maximum K was considered to be achieved at a DIN concentration of 500 µg N.l^{-1} with no further increase in K expected for DIN concentrations above this level.

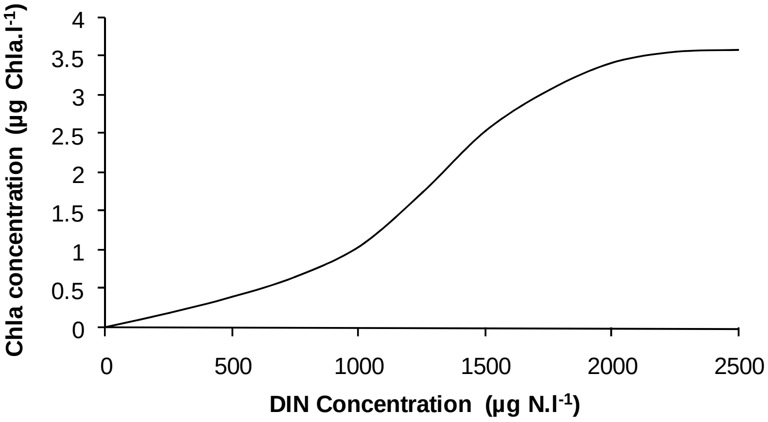

Figure 4.10. Growth rate (r) of phytoplankton defined in terms of dissolved inorganic nitrogen (DIN) concentration in the system.

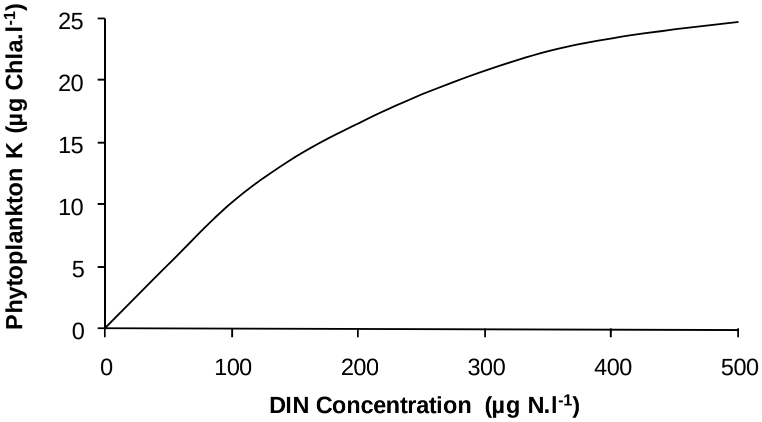

Figure 4.11. Relationship between phytoplankton K and DIN concentration used in the model.

4.3.2.3 Uptake of dissolved inorganic nitrogen (DIN) by phytoplankton

In the model, uptake of DIN by phytoplankton was assumed to occur in proportion to the growth of the phytoplankton stock. However, the phytoplankton stock is measured in terms of its chlorophyll-a biomass rather than actual mass. Thus, in order to calculate the rate of DIN uptake by phytoplankton it was first necessary to express phytoplankton biomass in terms of whole mass rather than chlorophyll-a, and to estimate protein content of the phytoplankton and the nitrogen content of the protein. We assumed that phytoplankton comprises ~30% chlorophyll-a, and ~30% protein and that protein contains ~6.25% N. Thus, the rate of DIN uptake by phytoplankton in the model was calculated as a factor of 6.25% of the growth of phytoplankton per day measured as chlorophyll-a biomass.

4.3.2.4 Loss/consumption of phytoplankton biomass

Phytoplankton biomass was removed from the system either through grazing (assumed to be mostly by phytoplankton-feeding invertebrates but also fish) or in response to a breaching event. In the latter case, the total mass of phytoplankton was lost from the system during a breaching event except for the biomass in the residual volume of water in the estuary. In the former case we used data from Froneman (2004) who estimated who estimated consumption rates of phytoplankton by zooplankton in the nearby Kasouga estuary. Consumption rates reported by Fromeman (2004) were reported to vary seasonally, in response to variations in the phytoplankton biomass (measured as Chl a concentration). We derived the following relationship from his data:

Ingestion rate (ng/zooplankton/d) = 5.567x Chl a concentration (mg.m^{-3}*) - 1.570*

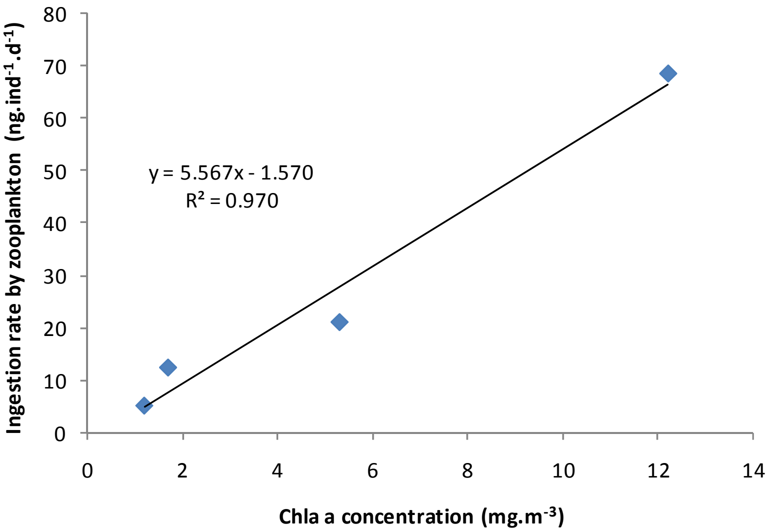

Figure 4.12. Consumption rate of phytoplankton by zooplankton (*P. hessi*) in the Kasouga estuary (data from Froneman 2004).

The correlation coefficient (r^2) for this relationship was 0.97, indicating a very good relationship. We applied this directly in the model, allowing zooplankton to graze on the phytoplankton in proportion to their abundance and in proportion to the density of phytoplankton in the system.

4.3.3 Benthic microalgae

4.3.3.1 Benthic microalgae biomass

Benthic microalgae are unicellular organisms similar to phytoplankton except that they that live attached to sand grains (episammic flora), bottom sediments (epipelic flora), rock (epilithic flora) and other plants below the surface of the water (epiphytic flora) (Adams & Bate 1999). The bulk of the benthic microalgae in the East Kleinemonde Estuary occur on submerged mud and sand (epipelic flora) or on submerged and emergent macrophytes (epiphytic flora). These two stocks were treated separately in the model expressed either as total biomass of chlorophyll-a in the system in kg or as chlorophyll-a concentration in kg.ha^{-1} or µg.m^{-2}. Starting values for benthic microalgae were taken from recent studies on the Kleinemonde system, taken as 140 mg.m^{-2} for submerged and emergent macrophytes (Gordon 2006) and is 242 mg.m^{-2} for mud and sand (Gama 2007). Total biomass at the start of the model simulation period (estuary open) was related to the available habitat area and translated to a total of 7.0 kg for the epipelic microalgae and 42.0 kg for the epiphytic microalgae.

4.3.3.2 Benthic microalgae growth rate

Little information is available on factors limiting benthic microalgae growth in estuaries. It was assumed that this would be the same as for phytoplankton (i.e. DIN) except that it is likely that the epipelic microalgae would be able to access some nitrogen from the sediments and hence maintain modest growth rate even when DIN concentrations declined to very low levels (i.e. <20 ug.l^{-1}). This is in keeping with findings from Adams & Bate (1999) who report that benthic microalgae biomass is generally greater than phytoplankton biomass, up to three times in some cases.

Growth of both microlagae stocks in the model was calculated using a standard growth equation of the form:

$Growth = N*r*(1-N/K)$

Where N = the current population size (measured as µg Chl a.m^{-2})
r = growth rate, and
K = the carrying capacity of the system.

Growth rates of benthic microalgae are reported to be similar to that for phytoplankton (Adams & Bate 1999). Thus, growth rate (r) was defined in relation to DIN concentration (Figure 4.13 and Figure 4.14) where r for the epipelic microalgae maintained a value of 0.5 even when DIN concentration declined to zero owing to reasons outlined above and rose to a maximum of 3.6 doublings per day, while r for epilythic microalgae ranged from 0 to a maximum of 3.6 doublings per day. Carrying capacity (K) was defined in terms of available habitat area (i.e. area of submerged mud and sand and submerged macrophyte area) with maximum allowable biomass of microalgae set at the highest measured concentration in the estuary (250 and 165 mg.m^{-2} for epipelic and epilythic microalgae respectively) multiplied by a factor of 1.25. Limited data are available for the East Kleinemonde Estuary but data for other estuaries in South Africa indicate that maximal concentrations lie in the range of 100-300 mg.m^{-2} (Adams & Bate 1999), which follows closely the values used in this study (313 and 206 respectively).

Rate of uptake of DIN in the system was calculated in the same way as for phytoplankton.

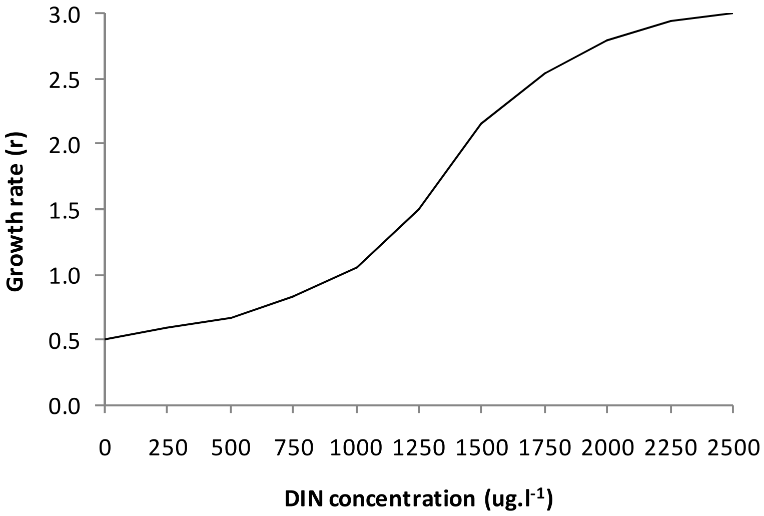

Figure 4.13. Growth rate (r) of epipelic microalgae in the East Kleinemonde Estuary defined in terms of dissolved inorganic nitrogen concentration in the system.

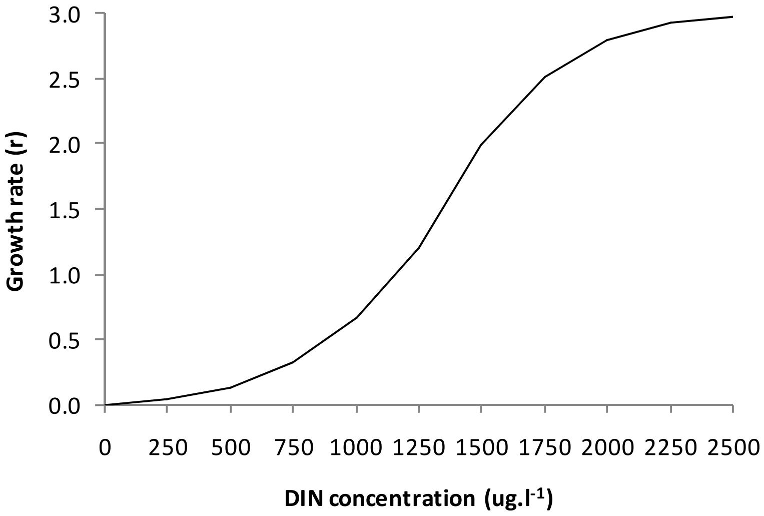

Figure 4.14. Growth rate (r) of epilithic microalgae in the East Kleinemonde Estuary defined in terms of dissolved inorganic nitrogen concentration in the system.

DIN production by blue-green algae was estimated as follows. It was estimated that 33% of benthic microalgal biomass was made up of blue-green algae. The rate of N-fixation is 1500 µg N.ha^{-1}.day^{-1} based on an average of three habitats (Talbot 1982). At the Swartkops estuary, blue-green algae produce some1500 µg N.ha^{-1}. The biomass of blue greens in the Swartkops is in the region of 20 µg.chl a.m^{-2}. A reasonable estimate of the rate of production is 25% per day, which would translate to some 0.004 kg.ha^{-1} (Bate pers. comm.). Therefore the rate of production of nitrogen was assumed to be in the order of 0.0375 kg N.kg^{-1} of blue-green production

4.3.3.3 Loss/consumption of benthic microalgae

Benthic microalgae biomass was removed from the system either through grazing (assumed to be mostly by invertebrates and fish) or in response to a breaching event. In the latter case, the total mass of benthic microalgae was lost from the system during a breaching event except for 25 µg.m^{-2} which remains on the residual surface area of mud and 16.9 µg.m^{-2} on the remaining macrophyte surface area. This is based on the lowest measurement on mud (Gama 2007) and macrophytes (Gordon 2006) at the East Kleinemonde Estuary. In the former case we used separate exponential density-dependent mortality rates for epipelic (Figure 4.15) and epilithic (Figure 4.16) microalgae, which follows the assumption that as benthic microalgae concentration in the system increases so will its attendant consumer biomass, collectively as a function of somatic growth, reproduction and as consumers switch to consuming phytoplankton when it becomes more abundant than other food sources.

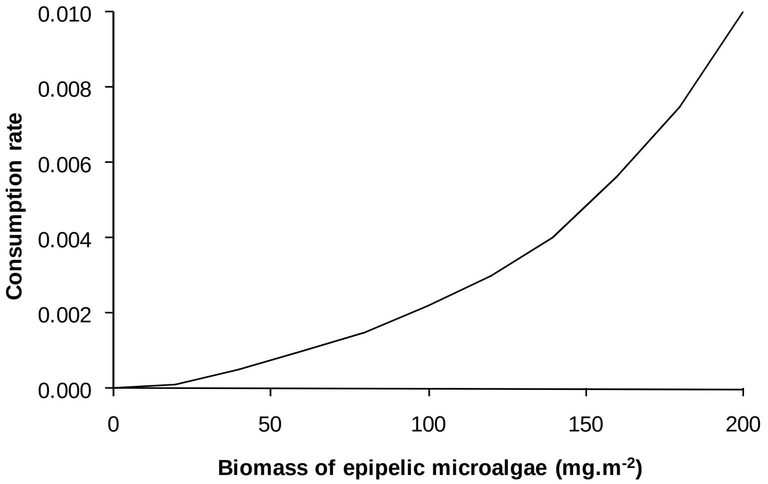

Figure 4.15. Consumption rate of epipelic microalgae applied in the model of the East Kleinemonde Estuary.

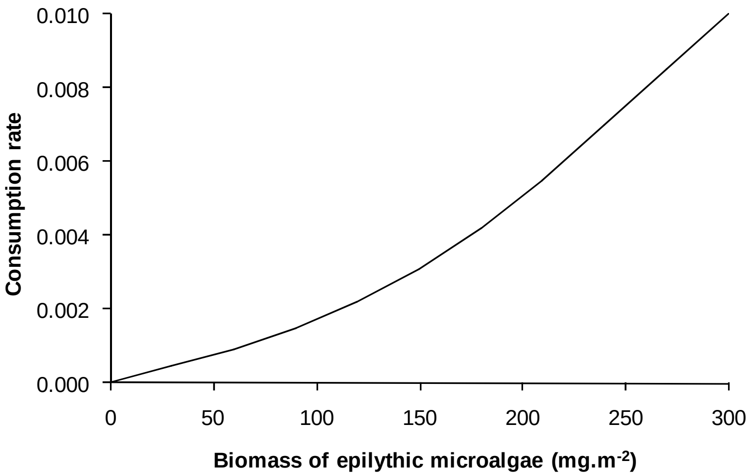

Figure 4.16. Consumption rate of epilythic microalgae applied in the model of the East Kleinemonde Estuary.

4.4 Macrophytes

4.4.1 Sub-model overview

This sub-model determines the changes in the area of submerged macrophytes, reeds and sedges, intertidal saltmarsh and supratidal saltmarsh, and as a residual, the area of mudbanks, sandbanks and open water, in response to four drivers: water level, mouth state, water column salinity and DIN. The vegetation types are quantified in terms of area cover (ha) rather than biomass. The model was based on data collected on the present vegetation cover in the system, coupled with estimated cover under extremes of mouth condition made with the use of bathymetric data in GIS (Table 4.1).

Table 4.1: Area cover of the different habitat types for the present (mapped using GIS) and two extreme scenarios.

Habitat Type	**Cover (ha)**		
	Present	Closed > 3 years (salinity less than 20 psu, maximum water level	Open > 4 months.yr^{-1}
Intertidal saltmarsh	4.04	0	9.46
Supratidal saltmarsh	6.36	5.36	6.36
Reeds and sedges	1.01	0.5	3.01
Mudbanks	9.83	0	3.915
Estuarine water area	12.69	14.07	12.69
Sandbanks	1.78	1.78	0.78
Submerged macrophytes	0.5	14.5	0
Total (ha)	**36.21**	**36.21**	**36.21**

4.4.2 Submerged macrophytes

All the submerged macrophytes occurring in the East Kleinemonde Estuary were combined into a single stock. Although there are differences in growth rate, biomass and rate of dieback the submerged macrophyte model was based on understanding of the dominant species i.e. *Ruppia cirrhosa*. Macroalgae is also included in this stock due to low biomass values recorded in the East Kleinemonde Estuary. Macroalgae include *Cladophora sp.* and *Ulva intestinalis* while submerged macrophytes include *Chara vulgaris, Halophila ovalis* and *Potamogeton pectinatus*.

Much of the biomass of the submerged macrophytes lies below the sediment surface in the form of roots and underground rhizomes. This root stock is less sensitive to exposure than the above ground biomass and continues survive for a considerable time after the above ground biomass has died and decayed and can allow the submerged macrophyte stock in the estuary to resprout rapidly after a period of exposure provided this is not excessive. In order to account for this the above ground macrophyte biomass and subsurface biomass (termed the seed bank) were modelled as separate stocks with the former dying back very rapidly in response to exposure (over a few days) while the latter was allowed to survive for up to 30 days before starting to die back.

The above-ground macrophyte stock was allowed to grow in accordance with the normal growth function:

$$Growth = N*r*(1-N/K)$$

Where the growth rate "r" varied in response to salinity and dissolved inorganic nitrogen (DIN) concentration in accordance with the response factors graphed in Figure 4.17 and Figure 4.18. The salinity growth response curve of *Ruppia cirrhosa* was derived from McMillan & Moseley (1967) and Adams & Bate (1994a, b) as well as from the observed salinity tolerance of the species in the East and West Kleinemonde estuaries (Ridden & Adams 2006). Maximum area attainable by the above ground biomass defined as "K" in the model was set as 1.05 times the area of the seed bank or subsurface biomass of submerged macrophytes in the estuary. Using a value of 1.05 allowed the subsurface areal coverage of submerged macrophytes to expand but at a much slower rate that that at which resprouting could take place, which corresponds with observations made by Ridden and Adams (2006). Maximum potential area that could be colonised by submerged macrophytes was limited by available habitat defined in accordance with the relationship with water area shown in Figure 4.19.

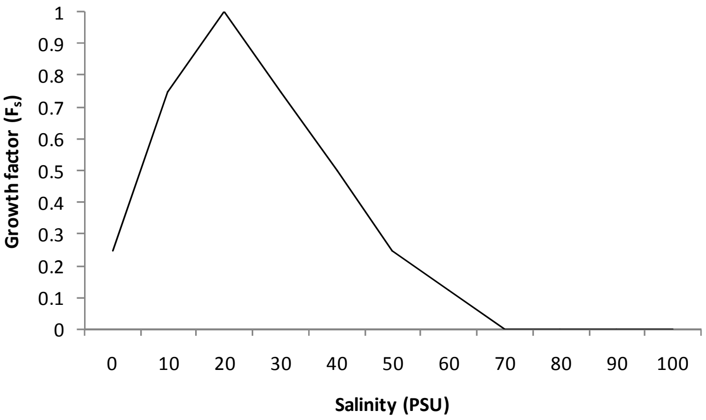

Figure 4.17: Growth response curve of submerged macrophytes to salinity

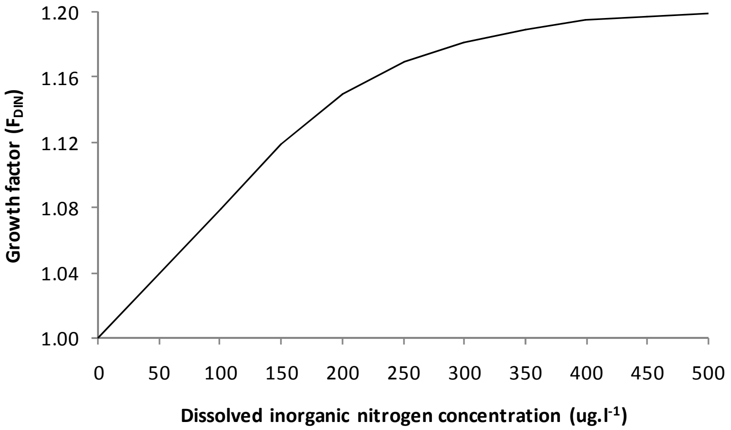

Figure 4.18. Growth response curve of submerged macrophytes to DIN

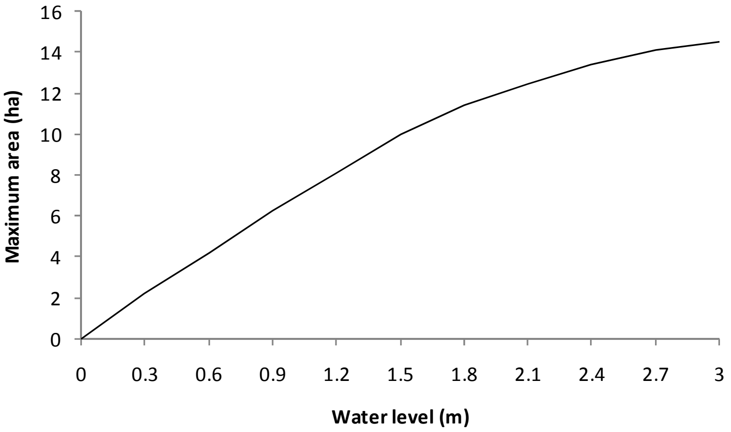

Figure 4.19: The relationship between water level and maximum potential area for submerged macrophytes

The loss of area due to dieback during a breaching event (as a result of desiccation, currents and salinity) was linked to the model as an out flow of the submerged macrophyte stock. Dieback rates for the above ground and subsurface macrophytes stocks differed as indicated above due to differences in their desiccation and hence mortality rates. When the mouth opened the above ground biomass was allowed to dieback at a rate of 50% of the exposed stock per day whereas death of the subsurface biomass (the seedbank) was only initiated after 30 days of exposure at a rate of 2% of the stock per day.

4.4.3 Reeds and sedges

All the reeds and sedges occurring in the East Kleinemonde Estuary were combined into a single stock for the purposes of the model. Although there are differences in growth rate, biomass and rate of dieback we based the reeds and sedges model on the dominant species i.e. *Phragmites australis*. The stock also included members of the Cyperaceae family.

Growth in area was estimated by the population growth equation:

$Growth = N*r*(1-N/K).$

The growth rate of reeds and sedges was estimated by the following equation:

$r*salinity\ factor$

…where r is the estimated growth rate at optimal conditions (0.001 which corresponds to an area expansion of 0.15 $ha^{-1}.yr^{-1}$ (Benfield 1984, Ridden 1999, Ridden & Adams 2006). The growth rate was influenced by salinity using the response curve in Figure 4.20. This was based on the published salinity tolerance ranges of *Phragmites australis* (Lissner & Schierup 1997; Hootmans & Wiegman 1998; Hartzendorf & Rolletchek 2001; Mauchamp & Mesleard 2001). The water level affects the maximum potential area that can be colonised by reeds and sedges (Figure 4.21). The response was assumed to be instantaneous.

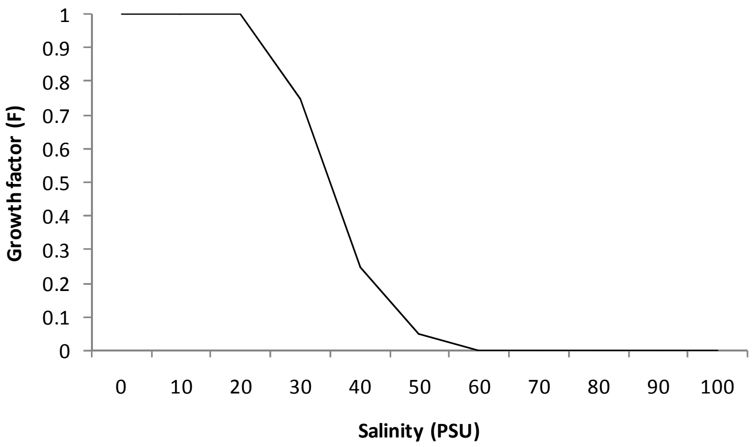

Figure 4.20: Growth response curve of reeds and sedges to salinity

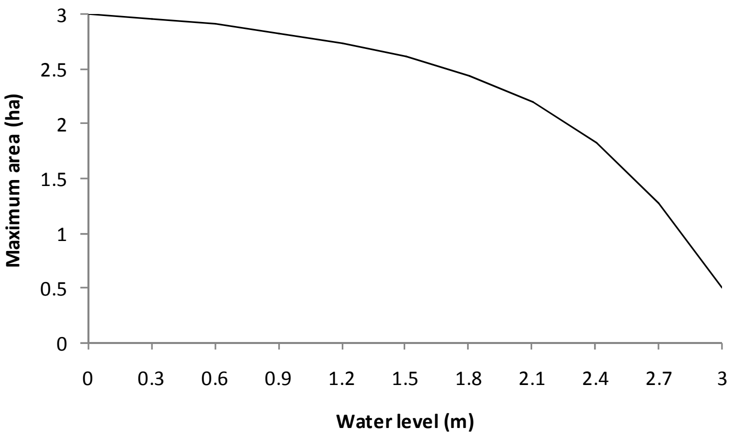

Figure 4.21: The relationship between water level and maximum potential area for reeds and sedges

The loss of area due to dieback was linked to mouth status, with a rate of loss of 0.005 ha.d^{-1} when the mouth is closed, down to a minimum stock of 0.5 ha. Ridden & Adams (2006) observed reed expansion only during the open phase when water levels in the East Kleinemonde Estuary is low.

4.4.4 Intertidal saltmarsh

All the intertidal saltmarsh occurring in the East Kleinemonde Estuary was combined into a single stock for the purposes of the model. Although there are differences in growth rate, biomass and rate of dieback, the intertidal saltmarsh model was based on the dominant species i.e. *Sarcocornia perennis*. Other species included *Salicornia* sp., and *Paspalum* sp.

Using the same growth equation as above, the growth rate of intertidal saltmarsh was estimated by the following equation:

*r*salinity factor*

…where r is the estimated growth rate at optimal conditions (0.1) of intertidal saltmarsh. Germination and establishment of intertidal saltmarsh occurs within 3 days of being exposed (Ridden & Adams 2006) therefore no delay function was required. The salinity response curve of *Sarcocornia perennis* (Figure 4.22) was derived from Adams & Bate (1993). The water level affects the maximum potential area that can be colonised by intertidal saltmarsh (Figure 4.23)

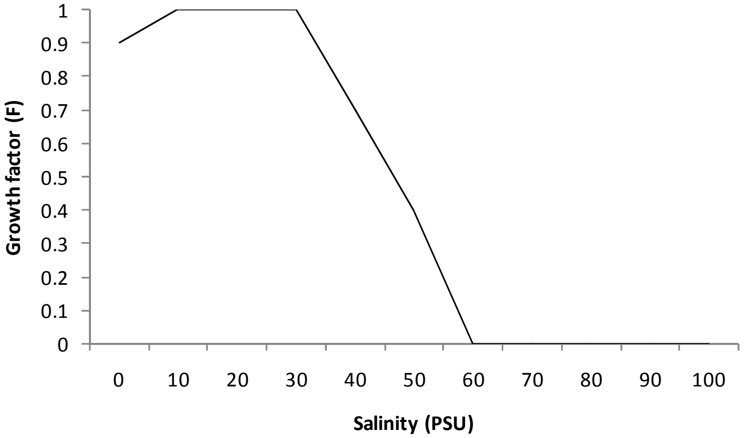

Figure 4.22: Growth response curve of intertidal saltmarsh to salinity

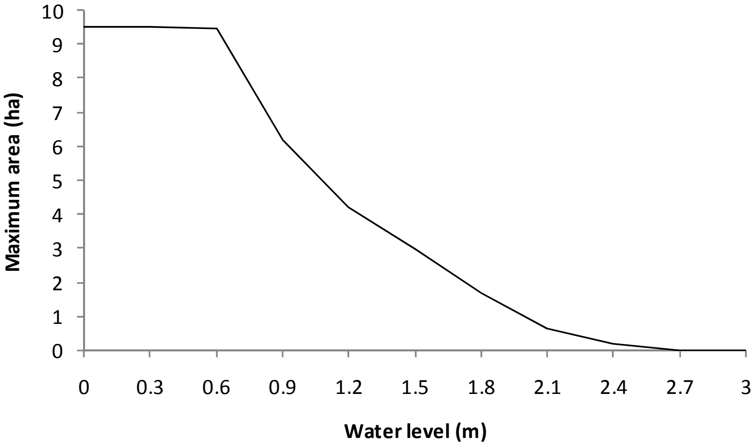

Figure 4.23: The relationship between water level and maximum potential area for intertidal saltmarsh

The loss of area due to dieback was linked to mouth status, in that if the mouth state is overtopping or closed then intertidal saltmarsh will dieback at a rate of 0.05 ha.d^{-1}. Intertidal salt marsh dies back completely after 2 months of inundation (Ridden & Adams 2006).

4.4.5 Supratidal and brackish saltmarsh

All the supratidal and brackish saltmarsh occurring in the East Kleinemonde Estuary were combined into a single stock for the purposes of the model. Although there are differences in growth rate, biomass and rate of dieback the supratidal saltmarsh model was based on the dominant species i.e. *Sarcocornia pillansii*. Other species included *Sporobolus virginicus, Juncus kraussii, Juncus acutus, Triglochin striata, Bassia diffusa, Limonium scabrum* and *Samolus porosus*.

The change in area was estimated using the growth equation. The growth rate was estimated by the following equation:

*r*salinity factor*

…where r is the estimated growth rate at optimal conditions (0.05 (Bornman 2002)). The salinity response curve of *Sarcocornia pillansii* (Figure 4.24) was derived from Bornman (2002). The water level affects the maximum potential area that can be colonised by supratidal and brackish saltmarsh (Figure 4.25; Bornman 2002).

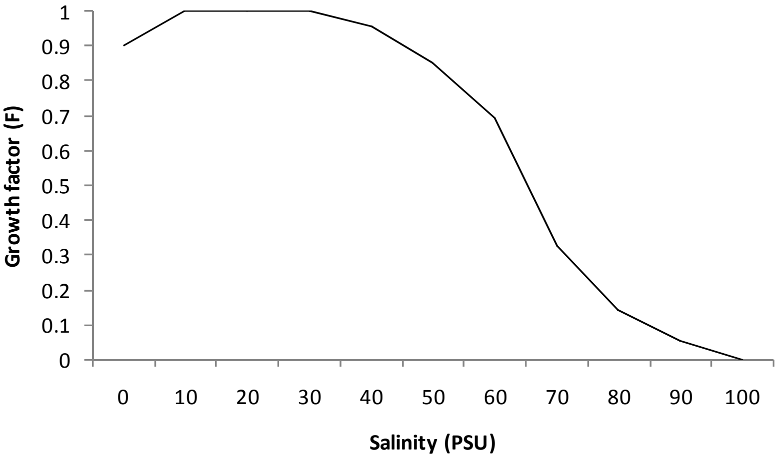

Figure 4.24: Growth response curve of supratidal and brackish saltmarsh to salinity

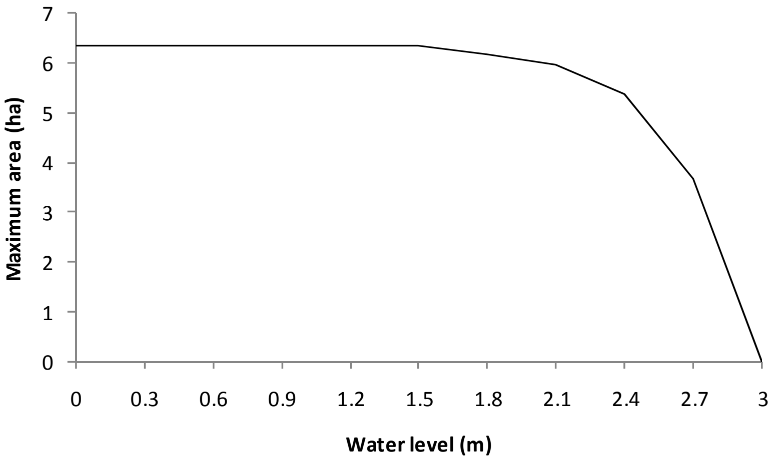

Figure 4.25: The relationship between water level and maximum potential area for supratidal and brackish saltmarsh

The loss of area due to dieback was linked to mouth status, such that if the mouth state is overtopping or closed then the supratidal and brackish saltmarsh will dieback at a rate of 0.001 ha.d^{-1}.

4.4.6 Mudbanks

Water level influences the maximum possible mudbank area (Figure 4.26). Maximum mud area achievable is 9.83 ha under the open mouth phase. During prolonged mouth closure the mudbanks may be colonised by *Phragmites* and intertidal saltmarsh. To estimate the colonisation of mudbanks by reeds the following equation was used:

IF reeds and sedges stock > 1 then (reeds and sedges –1) else 0

….where max mud area is a function of water level (Figure 11) and reed encroachment is estimated above and 0.5 is the fraction of possible area of reed encroachment into mudbanks (other 0.5 into sandbanks).

IF intertidal saltmarsh is > 5 then (intertidal saltmarsh – 5) else 0

….where the intertidal saltmarsh can expand into the mudbanks area by a maximum of 5 ha during the open phase.

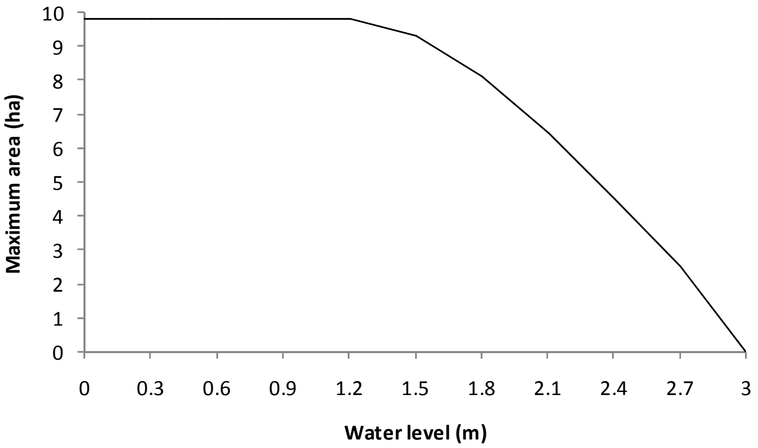

Figure 4.26: The relationship between water level and maximum potential area for mudbank.

The total mudbank area at given time is estimated as:

*Max mud area - intertidal saltmarsh encroachment - reed encroachment*0.5*

….where max mud area is a function of water level (Figure 4.26), intertidal saltmarsh encroachment and reed encroachment is estimated above and 0.5 is the fraction of possible area of reed encroachment into sandbanks (other 0.5 into mudbanks).

4.4.7 Sandbanks

Water level influences the maximum possible sandbank area (Figure 4.27). Maximum sand area achievable is 1.78 ha under the open mouth phase. During prolonged open mouth conditions the sandbanks may be colonised by *Phragmites* to a maximum of 1 ha (restricted by growth rate). The relationship between *Phragmites* and the colonisation of the sandbank (reed encroachment) is given by the equation:
If reed and sedges stock is > 1 then (reed and sedges stock – 1) else 0

Therefore, even under prolonged conditions there will always be 1.78 ha of submerged sandbank area available under open conditions 1 ha of this sandbank area may be colonised by *Phragmites* (if conditions favour the growth of reeds). The estimate total sandbank area at any given time is the following equation was used:

*Max sand area-reed encroachment*0.5*

….where max sand area is a function of water level (Figure 4.27) and reed encroachment is estimated above and 0.5 is the fraction of possible area of reed encroachment into sandbanks (other 0.5 into mudbanks).

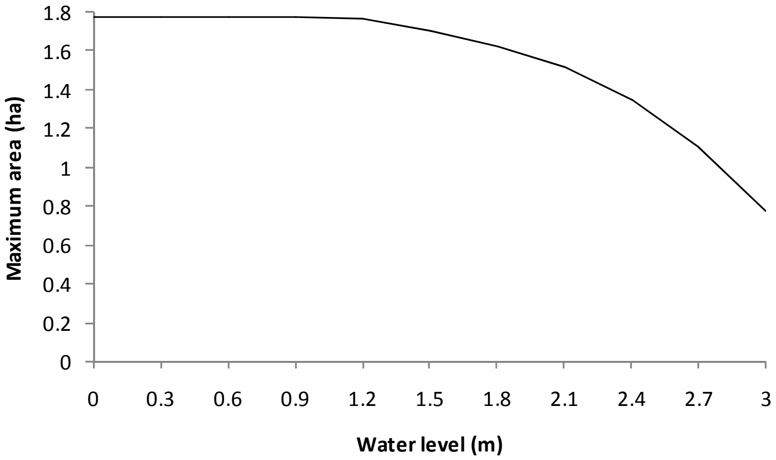

Figure 4.27: The relationship between water level and maximum potential area for sandbanks

4.4.8 Estuarine open water area

The total open water area (only mudbanks and sandbanks below i.e. no submerged macrophytes or reeds) is influenced by the area covered by intertidal saltmarsh, reeds and sedges, submerged macrophytes, supratidal and brackish saltmarsh, sandbank area and mudbank area. Therefore the total open water area is estimated by the following equation:

36.21 – (intertidal saltmarsh + reeds and sedges + submerged macrophytes + supratidal and brackish saltmarsh + sandbank area + mudbank area)

…where 36.21 ha is the total area of the estuary.

4.5 Invertebrates

4.5.1 Overview

The invertebrate sector of the model comprised the zooplankton and the benthic macrofauna. Data for this section were drawn primarily from recent studies on the East Kleinemonde (Vorwerk & Froneman 2007, Wooldridge 2007), but also from older studies (Wooldridge) and the literature. Abundance of invertebrates is generally measured in terms of numbers, and reliable estimates of biomass are difficult to come by. Errors in the Vorwerk & Froneman (2007) report meant that the estimates made here may need revision. In general the invertebrate sector concentrated on the dominant species, ignoring a few minor species that make up an insignificant portion of numbers and biomass. Hyperbenthos was not modeled as a separate group, but was assumed to be 'captured' within the zooplankton and benthic macrofauna.

4.5.2 Zooplankton

Zooplankton in the East Kleinemonde Estuary comprise a suite of at least 43 species but are dominated by Nauplii larvae, *Acartia longipatella,* Mussel larvae, Zoeae, *Harpacticoidia* sp., *Limacina* sp., and *Pseudodiaptomus hessei (*Vorwerk & Froneman 2007). These were all were modeled as a single stock with inflows in the form of the standard growth equation and outflows from the stock comprising mortality due to starvation and losses during breaching. Abundance of zooplankton in the East Kleinemonde system are reported to vary very widely, ranging from 44 to 44 000 individuals per m^3 or 17 to 58 mg DW m^{-3} (Vorwerk & Froneman 2007).

The modeled stock of zooplankton was allowed to grow in accordance with the function

*Growth = N*r*(1-N/K)*

Where the growth rate "r" was dependant on phytoplankton concentration in the estuary in accordance with the relationship depicted in Figure 4.28, and was constrained by the carrying capacity "K" also defined in terms of phytoplankton concentration (Figure 4.29).

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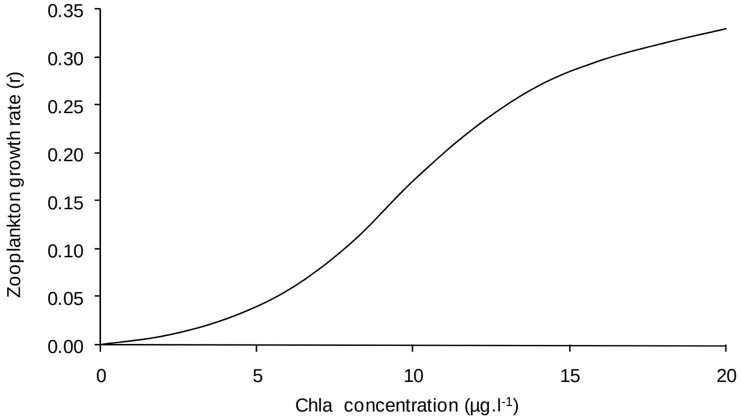

Figure 4.28. Zooplankton growth rate (r) as a function of chlorophyll a concentration.

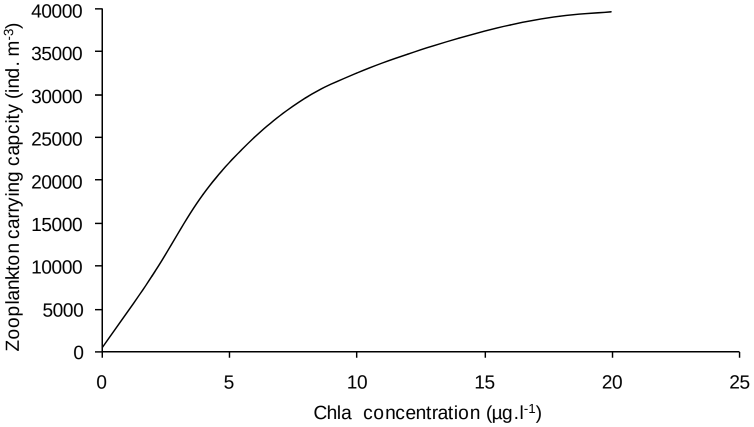

Figure 4.29. Zooplankton carrying capacity defined in terms of chlorophyll a concentration

A starvation function was introduced to reduce zooplankton populations back to the carrying capacity of the population whenever this was exceeded due to consumption of phytoplankton or through reductions in phytoplankton abundance for any other reason (e.g. reduction in DIN concentration). Loss during breaching was effected by flowing out zooplankton in accordance with the volume of water lost during the breach multiplied by the concentration of zooplankton in the system at the time.

4.5.3 Benthic macrofauna

The benthic macrofauna were modelled as trophic groups or subgroups that were assumed to behave in a relatively similar fashion: *Hymenosoma* crabs, *Callianassa* sandprawns, cumaceans, bivalves, peracarid crustaceans, predatory isopods, *Desdemona* polychaetes, *Prionospio* polychaetes, and predatory polychaetes. Abundance is expressed in both numbers and biomass. The model was based on measured densities in prolonged closed and open conditions in sand and in mud. For each substrate, average measured densities between the two extreme conditions were interpolated using a sigmoid-shaped function. Since there is no growth function, the abundance of these groups responds instantly to changes in habitats. In most cases, their response is expected to be fast. Nevertheless, there are interesting responses of some invertebrate species such as *Caliannassa* to change in conditions which should be incorporated in future versions of the model.

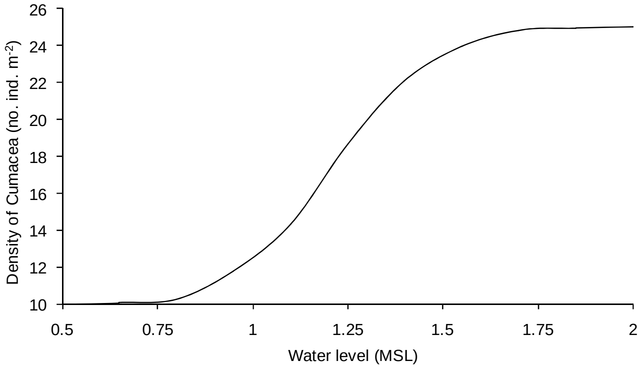

Figure 4.30. The relationship between water level and density of Cumacea

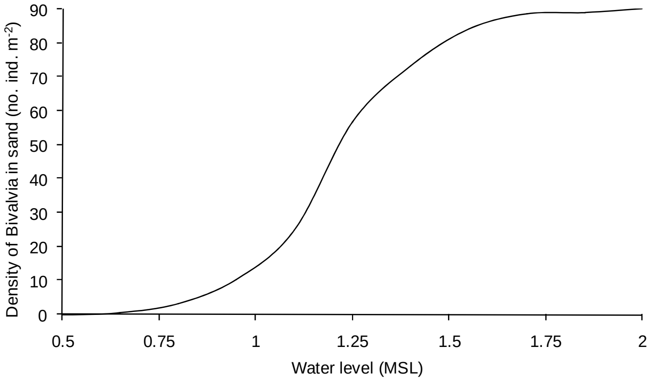

Figure 4.31. The relationship between water level and density of bivalves in sand

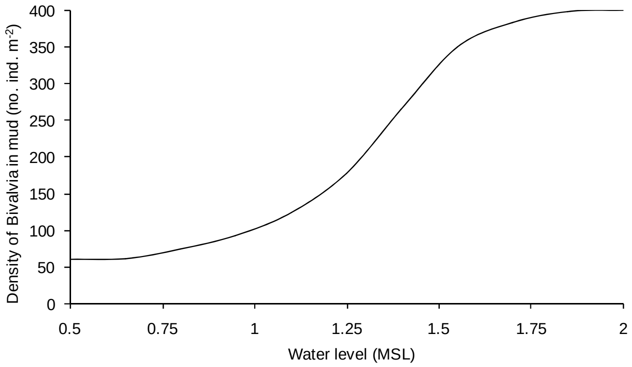

Figure 4.32 The relationship between water level and density of bivalves in mud

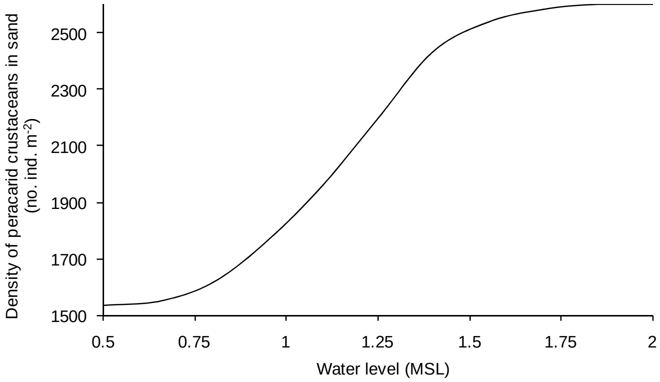

Figure 4.33. The relationship between water level and density of peracarid crustaceans in sand.

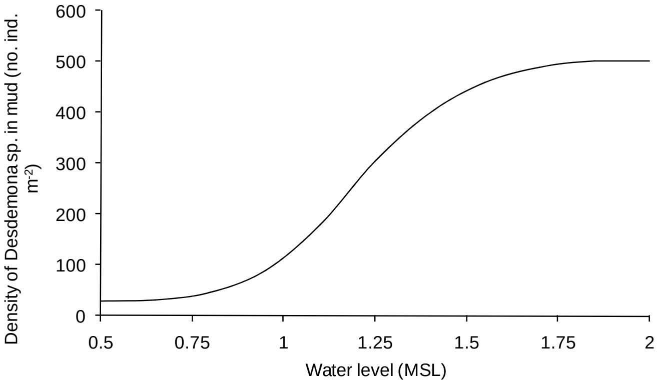

Figure 4.34. The relationship between water level and density of *Desdamona* sp. in mud.

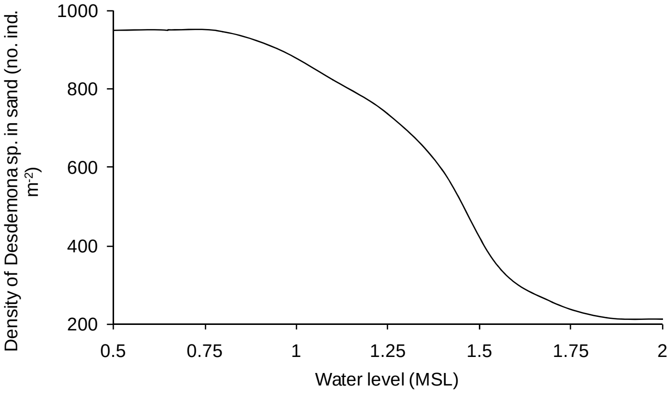

Figure 4.35. The relationship between water level and density of *Desdamona* sp. in sand.

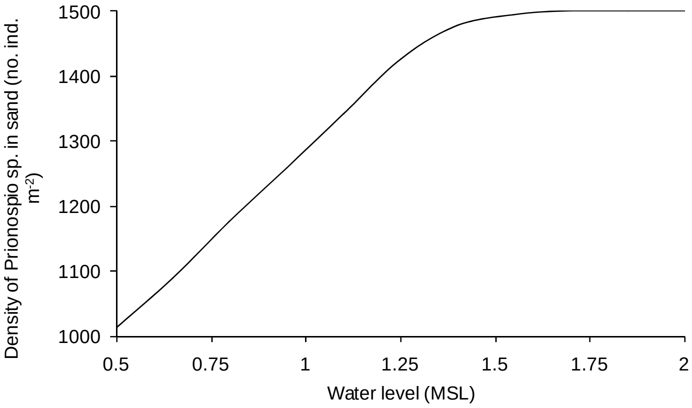

Figure 4.36. The relationship between water level and density of *Prionospio* in sand.

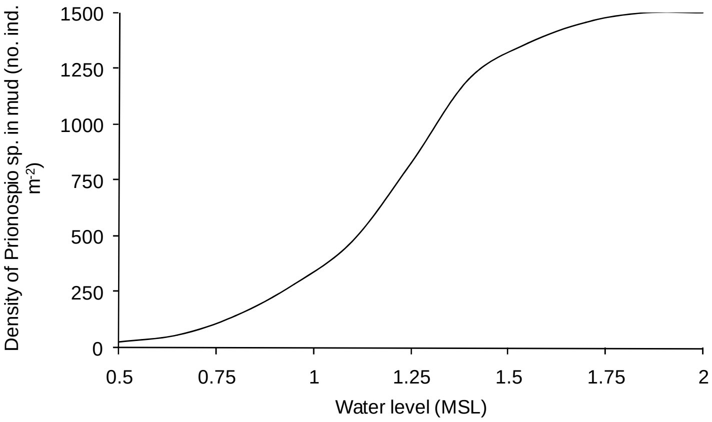

Figure 4.37. The relationship between water level and density of *Prionospio* in mud.

4.6 Fish

4.6.1 Background

The fishes associated with the East Kleinemonde Estuary have been subject to considerable research attention over the last decade. Published findings of this research effort include descriptions of the fish community (Cowley and Whitfield 2001a, Vorwerk *et al.* 2003, James *et al.* 2008), quantification of fish population sizes (Cowley and Whitfield 2001b), fish biomass and production estimates (Cowley and Whitfield 2002), composition of larvae in the marine environment adjacent to the mouth of the estuary (Cowley *et al.* 2001), composition and abundance of larval fishes in the estuary (Strydom *et al*. 2003) and the recruitment strategies by marine-spawning species (Bell *et al*. 2001). More recently, as part of the WRC Project k5/1581: Freshwater requirements of Cape intermittently open estuaries; additional research included the following has been undertaken:

- An investigation of the long-term inter-annual changes in fish community structure, with emphasis on estuarine mouth conditions. This study made use of a long-term dataset of seine and gillnet catches that was initiated in 1993 (Cowley and Whitfield *unpublished data*) and was recently published as a PhD thesis (James 2006, James et al. 2008).
- James *et al.* (2007) conducted an assessment of the year-to-year recruitment success of *Rhabdosargus holubi* and *Lithognathus lithognathus*. This study also made use of the long-term dataset as well as records of daily mouth state and linked them to the recruitment of distinct year-class cohorts between 1994 and 2006.
- A study on the temporal abundance and recruitment dynamics of the dominant freshwater species *Oreochromis mossambicus*. A fry seine net was used to sample the littoral zone fish community at 24 sites on a bi-monthly basis from March 2005 to September 2006. The data were published as two independent BSc (Hons) projects (Shanyengange 2005 and Ellender 2006).
- An investigation of the larval and small juvenile fishes, with particular reference to the influence of freshwater input and estuary mouth state (Muller 2006, Strydom *unpublished data*).

4.6.2 Summary of key findings

Analysis of the long-term datasets of fish catches and estuary mouth state revealed that the timing of mouth opening had a significant effect on species composition in the East Kleinemonde Estuary with MDS grouping years into two distinct groups (Figure 4.38). More species were recorded during years that succeeded spring (September to November) mouth opening events than in years following no mouth opening events during this period. Mean annual catch per unit effort (CPUE) for species that recruit predominantly in spring were higher in ‘spring opening’ years. Species that are known to recruit during both overwash and open mouth conditions were consistently recorded each year, irrespective of a ‘spring opening’.

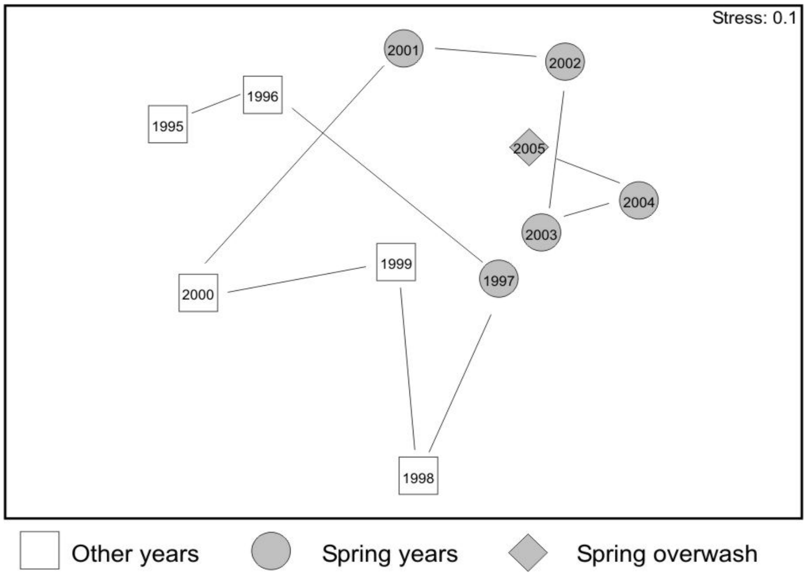

Figure 4.38. Ordination plot showing annual seine net fish CPUE data from the East Kleinemonde Estuary between 1995 and 2005 (after James 2006).

Individual fish populations in the estuary were dynamic as the abundance (CPUE) of all species varied markedly between years (Table 4.2). Quantitative population size estimates, using mark-recapture methods, also revealed a high degree of inter-annual variability (Cowley and Whitfield 2001b).

Table 4.2. Annual CPUE for species caught in large seine nets in the East Kleinemonde Estuary between 1995 and 2005 (after James 2006).

Species	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**
Rhabdosargus holubi	53.3	25.4	97	135.6	88.6	40.2	21.3	22.8	264.4	54.1	30.2
Lithognathus lithognathus	1.5	0.9	7.7	2.8	2	1	0.4	0.3	1.8	2.1	3.1
Lichia amia	0.5	0	0	0	0	0	0	0	0	0	0.3
Myxus capensis	0.2	2.6	16.3	0.3	8.9	34.8	2.7	25.2	19.5	82.6	18.8
Liza dumerilii	0.2	0.8	4.6	13.3	0.5	0.9	1.8	1.7	1.6	6.6	3.9
Liza richardsonii	0.6	4.1	5.1	24.4	3.2	0	1.1	1.8	5.8	7	3.3
Liza tricuspidens	0	0	0.3	0	0	0	0.4	0.7	0.8	0.8	1
Mugil cephalus	0.1	0.2	0.8	0	0.1	0.2	0.1	6.9	1.1	0.9	0.2
Argyrosomus japonicus	0	0	0	0	0	0	0	0	0	0.1	0.1
Oreochromis mossambicus	0	3.7	2.8	1.4	2.8	0.1	0.2	0.5	8.4	0	3.4
Monodactylus falciformis	1.2	0.4	24.3	9.6	13.2	7.3	0.8	0.6	2.2	4.1	2.7
Pomadasys commersonni	0	0	0.1	0.4	0.1	0	0.3	0.2	0.3	0.3	0.1

Rhabdosargus holubi was the dominant species in large seine net catches throughout the study period and comprised more than 50% of the catch in all years except 2000, 2002, 2004 and 2005. In years when *R. holubi* comprised <50% of the catch *Myxus capensis* comprised a large proportion of the catch (Table 8.1). Also of importance in large seine catches were *Monodactylus falciformis*, comprising between 0.7% (2003) and 15% (1997) of the catch and *Lithognathus lithognathus* comprising between 0.4% (2002) and 4.8% (1997) of the catch (James 2006). A coefficient of concordance was used as an index of community stability. The rankings of the top ten species between 1995 and 2005 indicated a 41% (0.41) concordance in the estuary over time; where a value of 1 (100%) would indicate no change in species rank over the study period.

The timing of mouth opening events and life-history pattern (reproductive seasonality) together influenced species composition and abundance. Species with medium to extended breeding seasons that recruit during overwash and open mouth conditions or breed in the estuary dominated catches numerically (Table 4.3). In contrast, species with restricted spawning seasons, most of which only recruit into estuaries during open mouth conditions, were found in low numbers within the estuary.

Table 4.3. Reproductive seasonality and recruitment behaviour of the 10 most abundant fish species recorded in the East Kleinemonde Estuary (after Cowley and Whitfield 2001b, James 2006).

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

4.6.3 Sub-model overview

The fish sub-model simulates changes in the abundance of key species in relation to mouth breaching events, recruitment (estuary access opportunities), reproductive seasonality, growth and natural mortality. Marine-spawning species included Cape Stumpnose *Rhabdosargus holubi,* White steenbras *Lithognathus lithognathus,* mullet species and Kob *Argyrosomus japonicas.* Estuary-spawning species included Estuarine roundherring *Gilchristella aestuaria,* Estuarine pipefish *Syngnathus watermeyeri,* and gobies (Gobidae).

A generic model was developed for selected marine-spawning and a second for estuary-spawning species, and adapted as required for each of the modelled species. The model for marine-spawning species included a stock component which grew as a result of recruitment (immigration) and diminished by way of losses as a consequence of mouth breaching (i.e.

emigration) or natural mortality (Z) a function of either natural (N) or fishing (F) losses, accounted for by the inclusion of input parameters based on values obtained from the literature. Estimates for recruitment were based on density estimates obtained from Cowley *et al*. (2001), Kemp and Froneman (2004) and Muller (2006). These values were multiplied by a concentration factor linked to a cueing response elicited by inputs of freshwater into the marine environment. A concentration factor of 10 was used, based on evidence provided by Whitfield (1989) and Harris and Cyrus (1999) who showed that post-flexion larval concentrations were approximately 10-fold higher during the open mouth phase (i.e. following a breaching event). The number of recruits entering the estuary was estimated using the concentrated larval densities multiplied by the modelled volume of water that enters the estuary during the mouth open phase. Overwash conditions were also considered for species that are known to recruit during these periods (see Bell *et al*. 2001, Cowley *et al*. 2001, Kemp and Froneman 2004). Estimates of densities of newly recruited 0+ fish within the estuary were obtained from Ellender (2006) and Muller (2006). In addition, density estimates as well as population size estimates of the temporarily resident fish populations in the estuary were obtained from Cowley and Whitfield (2001b).

The model for estuarine-spawning species adopted a different approach. Here the stock was modelled using a standard population growth equation:

$$G = N * r * (1 - N/K)$$

where: G = population growth, N = estimated population size, r = population growth rate (dependent on food abundance e.g. zooplankton stock) and K = carrying capacity (max recorded density multiplied by a factor of 1.25 and multiplied by estuary surface area). Where relevant, losses associated with fish being stranded in exposed macrophyte beds following a breaching event (Cowley *personal observations*) or of fish being flushed out of the system, were also built into the various species models.

A brief description of the rationale and source of each the input and output parameters are provided in the species profiles that follow. Models were not developed for all species. Instead, selected species were chosen on basis of numerical dominance, gravimetrical abundance (biomass), reproductive guild (marine- versus estuarine- spawning species), recruitment strategy (overwash versus open mouth recruitment), trophic guild, fishery importance and conservation significance.

4.6.4 Cape Stumpnose *Rhabdosargus holubi*

Rhabdosargus holubi is endemic, marine-spawning, estuarine dependent species (category IIa) that can tolerate salinity ranges of 0.7 - 70 ‰ and temperatures ranging between 10 and 30 ºC. The diet of postlarval *R. holubi* (approximately 15 - 30 mm SL) consists mainly of copepods, whereas juveniles (> 30 mm SL) feed mainly on filamentous algae, aquatic macrophytes and epibenthic invertebrates (Whitfield 1984, 1985). Blaber (1973) found that juvenile *R. holubi* in the West Kleinemonde estuary feed mainly on aquatic vegetation that is not digested, whereas the assimilated portion of their diet comprises epiphytic diatoms. Blaber (1974) revealed that *R. holubi* has a winter/spring peak spawning period with a spring/summer influx of 0+ individuals to the West Kleinemonde estuary. The abundance of post-flexion larvae and early juveniles (length range: 9 - 21 mm) peaks in early spring (August and September) but are recorded throughout the year in the surf zone adjacent to the mouth of the East Kleinemonde Estuary (Cowley *et al.* 2001). Consequently, this species' serial spawning habits allow for high recruitment success, particularly considering that mouth opening events are not strictly seasonal, but occur sporadically throughout the year. Furthermore, *R. holubi* also recruit successfully during overwash conditions (Cowley *et al*. 2001).

R. holubi juveniles (<60 mm SL) were recorded consistently in summer and winter samples between 1995 and 2005. Recruitment by this species appeared to be uninterrupted and was not determined by seasonality of mouth opening events. This species dominated the marine-spawning component of the fish community and comprised 63% (range = 34 - 92%) of the catch composition sampled with the large mesh seine net. During the period 1993-1996 (representative of a submerged macrophyte dominated phase) the marine-spawning fishes associated with the estuary were overwhelmingly dominated by *Rhabdosargus holubi*, which:

- comprised 77.6% of the ichthyoplankton catch composition in the surf zone adjacent to the mouth of the estuary (Cowley *et al*. 2001),
- comprised 75.3% of the estuarine catch composition sampled with the large mesh seine net (Cowley and Whitfield 2001a),
- accounted for approximately 55%, 75% and 80% of the total population size of marine-spawning species above a certain minimum size during three independent mark-recapture experiments (Cowley and Whitfield 2001b), and
- accounted for 74% of the total secondary production by the entire estuarine fish community (Cowley and Whitfield 2002).

More recently, Ellender (2006) revealed that *R. holubi* was the third most abundant species in shallow littoral of the East Kleinemonde with a mean density of 1.365 m^{-2}.

For the purposes of the model, *R. holubi* was allowed to recruit throughout the year, with post-flexion larvae of 10 to 15 mm SL entering the estuary during breaching and overwash events. The number of recruits for each estuary access opportunity (EAO) was based on surf zone larval density estimates provided by Cowley *et al*. (2001b) and multiplied by a factor of 10 to account for concentrations during recruitment pulses (after Whitfield 1989). Recruitment continued with every EAO until concentrations of 17 individuals per m^2 of submerged macrophyte habitat together with 0.5 individuals per m^2 of non-vegetated habitat were obtained. This led to an average number of recruits of about 1.3 per m^2 in shallow water area (under 30cm) over the whole period, similar to the average density found on average by Ellender

(2006). Losses were ascribed to natural mortality at a rate of 10% per month and a 20% loss during a breaching event. Thirteen percent of the recruit population were estimated to reach a modal size of 60 mm and were recruited into the juvenile stock. Mortality of juveniles was set at 0.5% per month and an additional 80% were lost during each quarterly breaching event.

4.6.5 White steenbras Lithognathus lithognathus

Lithognathus lithognathus is an endemic, estuarine dependent marine-spawning species. This species has a relatively constant and limited breeding season, with spawning taking place in the nearshore environment off the Eastern Cape coast primarily between June and August (Mehl 1973), while juveniles recruit into estuaries at a size of about 50 mm TL (Bennett 1993). Long-term records of daily mouth conditions were linked to the recruitment of distinct year class cohorts in the East Kleinemonde Estuary (James *et al.* in prep). Significant recruitment into the estuary only occurred in years when the mouth opened between late August and January. Recruitment and CPUE of small and larger juvenile *L. lithognathus* increased from 2002 onwards when the mouth opened consistently during spring, thus allowing 0+ juvenile fish to enter the estuary on an annual basis (James *et al*. in prep). Analysis of nine distinct recruitment cohorts between December 1994 and July 2006 showed that size at recruitment ranged from 15 to 45mm SL. Size at departure ranged from 210 to 340 mm (maximum modal sizes recorded in the estuary). Residency periods of these cohorts ranged from 28 to 48 months (James *et al*. in prep). Recruitment and departure only occurred during opening events, not during over-wash. There were also years of little or no recruitment when the mouth failed to open during spring (1997 and 1999) or possibly as a consequence of recruitment failure from the marine environment. Recruitment failure into TOCEs may have major implications for this important fishery species. Year-class strengths of fishery recruits may decrease in the years following closed mouth conditions in spring, thus adding to the pressure exerted on the existing stock, which is already considered to be collapsed.

In the model, recruits were only allowed to enter the estuary during mouth opening events between August and January, with no recruitment taking place during overwash events. Recruits entered the estuary within the size ranges depicted in Figure 4.40 (after James *et al*. 2008) and remained in the estuary for at least the first year of life, growing at a rate of 10mm per month. It was estimated that approximately 1 000 individuals above 120mm are in the estuary at any-one-time, with a maximum carrying capacity of 8 000 fish. It was assumed that 65% of fish older than one year leave with each quarterly breaching event. Maximum recruitment per year was modelled as a function of the number of days the estuary was open per year in the period August to January with numbers declining exponentially as number of days open per year increased (Figure 4.39). Loses due to natural mortality were set at 10% per month for the first year and 1% per month for subsequent years.

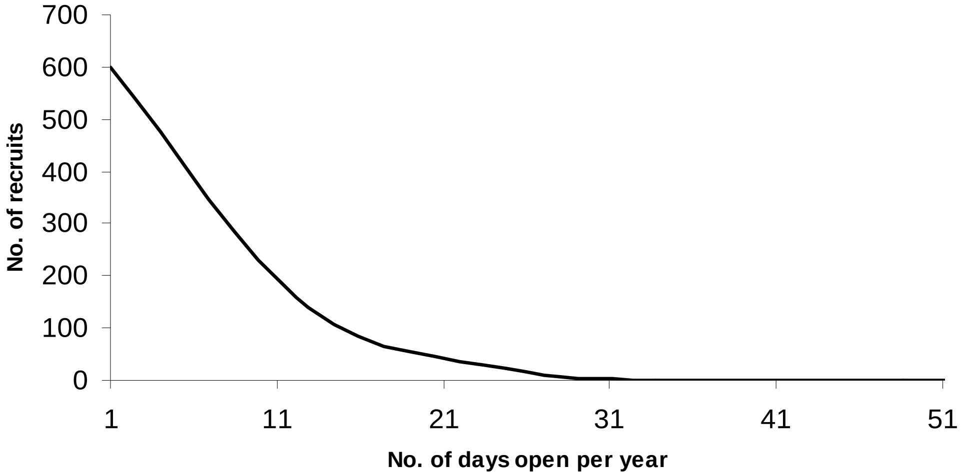

Figure 4.39. Relationship between the number of *L. lithognathus* recruits entering the East Kleinemonde Estuary and number of days that the mouth remains open in that year in the period Jan-Aug.

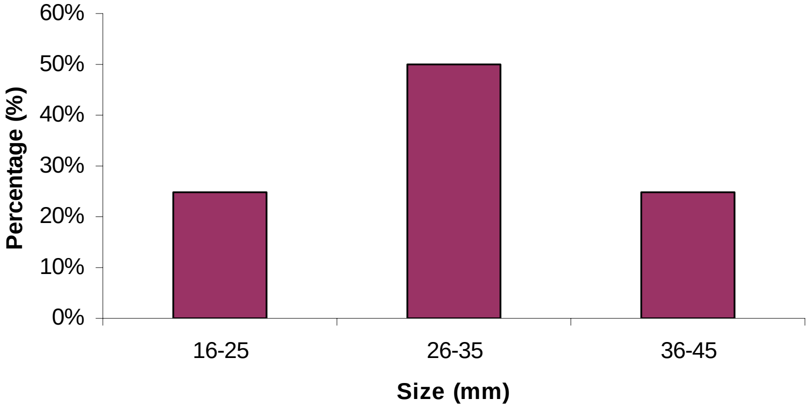

Figure 4.40. Modal sizes of *L. lithoganthus* recruits entering the East Kleinemonde Estuary (after James *et al*. 2008).

4.6.6 Mullet (Mugilidae)

Mullet are widely distributed along the South African coastline and most species are dependent on estuaries as nursery grounds. All species spawn in the marine environment and recruit into estuaries mostly as post-flexion larvae. Mugilids feed predominantly on the lower levels of the trophic pyramid, focusing on macrophytic plant material, filamentous algae, epiphytic diatoms, forminiferans, copepods and macruran larvae (Whitfield 1998). The dominant mugilids in the East Kleinemonde Estuary include *Myxus capensis, Liza richardsonii, Liza dumerilli*, *Liza tricuspidens* and *Mugil cephalus* (Cowley and Whitfield 2001a). The inter-annual abundance of all species was variable over the period 1995 to 2005 (Table 4.2). Cowley and Whitfield (2001b) revealed that mugilids had a collective population size of approximately 14 000 individuals with a total biomass of 5.9 g.m^{-2} during a macrophyte dominated phase between 1993 and 1996. Recruitment occurs predominantly in spring and summer months and the recruitment period varies from short (1-3 months) to extended (>7 months) depending on the species (Table 4.3). Mugilids recruit into the estuary at lengths between 9 and 15 mm SL during both mouth opening and overwash events. Based on the size composition of fishes in the estuary, mugilids have residency periods of one to three years. Mean density of early recruits in the shallow littoral zone of the East Kleinemonde Estuary was 0.943 m^{-2} (Ellender 2006). Average densities of post-flexion Mugilidae in the surf zone adjacent to the closed mouth of the estuary were recorded as 0.4 per 100 m^{3} (Cowley et al. 2001).

For the model simulations, it is assumed that following a breaching event post-flexion recruits (9-15 mm BL) would concentrate by 10 fold off the mouth of the estuary (after Whitfield 1998) and would enter the estuary at a concentration of 0.004 ind. m^{-3} multiplied by the modeled volume of water that entered the estuary. The stock at the start of model simulation period was set at 15 000 post-flexion larvae, 800 1+ year old fish and 250 2+ aged fish. Losses associated with breaching events were set at 90% of 2+ aged fish per quarterly breaching event. Monthly mortality was assumed to be 10%, 5% and 2% for 0+, 1+ and 2+ individuals, respectively. Survival was numerated as follows: 20% of new recruits (post-flexion larvae) became 1+ year olds and 50% of remaining fish entered subsequent year classes (adults).

4.6.7 Kob *Argyrosomus japonicus*

Argyrosomus japonicus is an estuary-dependent picivorous predator that spends at least its first year of life in estuaries. Sexual maturity is reached at approximately 100 cm at an age of 6 years, spawning taking place close inshore, nocturnally from October to January in the Eastern Cape. Juveniles (20-30mm) recruit into estuaries from October to February. These recruits remain in the upper, fresher and more turbid parts of estuaries only migrating to the middle reaches once they reach 150mm. Growth is rapid and *A. japonicus* reach 350-400mm in their first year. Diet of new recruits (<50mm) consists mostly of copepods and mysids whereas fish greater than 170mm are primarily piscivorous feeding on small fish such as *G. aestuaria* and juvenile Mugilidae. As they grow prey also includes larger fish species such as *R. holubi*.

Recruitment into the East Kleinemonde is usually low with occasional good years, which coincided with major breaching and prolonged opening events during summer months. Furthermore, *A. japonicus* recruitment into the East Kleinemonde and similar systems is probably enhanced by the increase in turbidity during phases of macrophyte senescence (e.g. 2003 – 2007). It is hypothesized that in most years the nearby permanently open Great Fish Estuary probably absorbs the bulk of the kob spawned locally. Surplus recruits (< 50 mm) that “miss” the Great Fish probably drift southwards usually suffering high mortalities but occasionally recruit into IOEs, provided the mouths of these system are open. However, slightly contradictory to the above hypothesis is that recent size frequency data from East Kleinemonde suggests that recruits comprise mostly >200 mm long fish and probably only recruit after having spent their initial juvenile stage in other more accessible systems such as the Great Fish (Table 4.4).

Table 4.4. Number caught, minimum, maximum and mean sizes of kob *A. japonicus* caught in the East Kleinemonde Estuary using gill nets between 1998 and 2007 (after James *et al*. 2008).

Year	No. caught	Min (mm SL)	Max (mm SL)	Mean (mm SL)
1998	9	332	472	378
2002	1			380
2003	18	155	481	343
2004	24	229	598	339
2005	23	290	520	413
2006	23	251	440	338
2007	9	232	401	291

For the model simulations, recruitment of *A. japonicus* was assumed to be very low, set at 500 individuals per million m^3 of influent seawater. Losses during the closed phase were assumed to be mostly a function of fishing mortality (F), estimated to be approximately 30% per annum (3% per month) (P. Cowley pers. obs.). The majority of the surviving fish (95%) were allowed to emigrate during the first breaching events of each quarter.

4.6.8 Estuarine roundherring *Gilchristella aestuaria*

Gilchristella aestuaria is one of the few members of the Clupeidae (sardines etc) to be found in freshwater. It is a small species seldom growing to more than 90 mm in length. *G.aestuaria* is a southern African endemic found in estuaries from Kosi Bay in the east to the Gariep (Orange) on the west coast (Whitfield 1998). It is predominately an estuarine species preferring the river-estuarine interface (REI) zone of 0-10 ppt but is capable of breeding in freshwater and occurs in the lower reaches of many rivers and coastal lakes. Being an abundant shoaling species it is an important food source to piscivorous fish and birds

Gilchristella aestuaria breeds throughout the year with peaks in the spring and summer months (Whitfield 1998). The eggs are 1 mm in diameter and buoyant, usually found suspended within the water column. In POEs, the larvae avoid ebb-tide surface currents to maintain position in the middle and upper reaches (Melville-Smith *et al.* 1981). *G.aestuaria* reaches maturity at 7 months of age at a length of 28 mm.

Gilchristella aestuaria was the most abundant species recorded in the estuary and comprised between 33% of the catch in 1996 to 94% of the catch in 2005 (James 2006). Cowley and Whitfield (2001a) also found that mouth state is important for the reproductive success of estuarine-spawning species in the East Kleinemonde Estuary. Reproductive activity is halted during low level water conditions following a mouth opening event while closed mouth conditions result in more stable physical conditions, elevated water levels and habitat inundation resulting in an increase in breeding success. However, analysis of the long-term dataset suggested that inter-annual changes in the abundance of certain estuarine-spawning species were related to rainfall and thus river pulses. A peak in CPUE of 1 662 fish per haul was recorded in 2005. This coincided with the highest annual rainfall recorded over the 11-year study period.

Whitfield (2005) reported that river pulses are particularly important in influencing the abundance of estuarine resident species. The filter-feeding *G. aestuaria* forms an important link in the food chain in South African estuaries as it is preyed upon by various predatory fish (Whitfield and Blaber 1978). Martin *et al.* (1992) recorded an order of magnitude increase in the abundance of *G. aestuaria* in the St Lucia Estuary following minor flooding that led to the mouth of the estuary opening. This was attributed to the phytoplankton bloom and increased zooplankton stocks associated with the flooding. However, Strydom *et al*. (2002) recorded an absence of young *G. aestuaria* in the water column of the Great Fish Estuary during periods of high flow and attributed this to large numbers of eggs and larvae being swept out of the estuary under these conditions.

Mortality in the East Kleinemonde Estuary occurs largely due to year round bird predation and stranding in exposed macrophyte beds during major breaching events. Recruitment is generally poor if the mouth remains open for an extended period during summer. Growth is mostly likely governed by whether they are feeding on zooplankton or phytoplankton and or whether they are filter or selectively feeding. Filter feeding is probably confined to high densities of zoo- & phytoplankton.

For the model simulations population growth, carrying capacity and mortality for *G. aestuaria* were all considered to be a function of zooplankton biomass in the estuary, read off the graphical functions in Figure 4.41., Figure 4.42 and Figure 4.43.

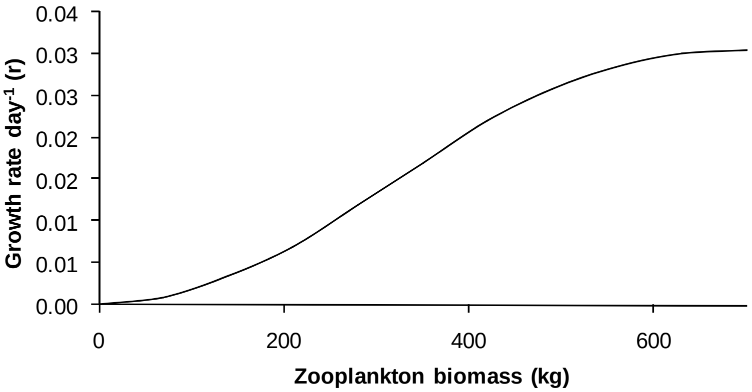

Figure 4.41. Relationship between *G.* aestuaria population growth in the East Kleinmonde estuary and zooplankton biomass.

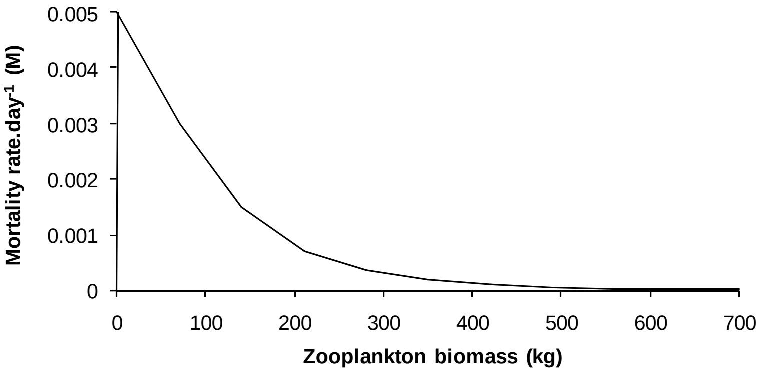

Figure 4.42. Relationship between mortality rate of *G.* aestuaria in the East Kleinemonde Estuary and zooplankton biomass.

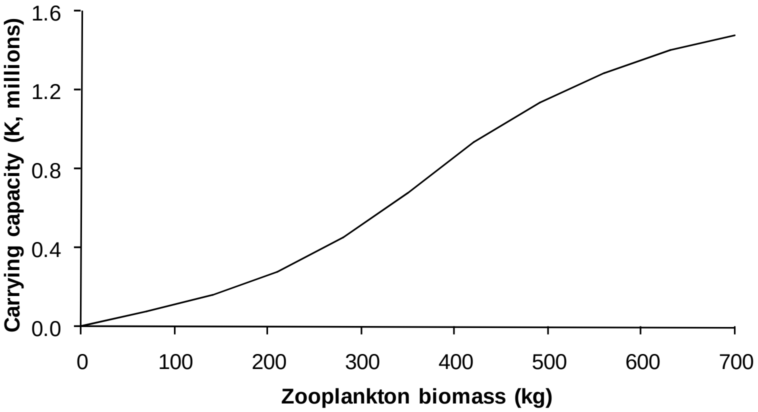

Figure 4.43. Relationship between carrying capacity (K) for *G. aestuaria* in the East Kleinemonde Estuary and zooplankton biomass.

4.6.9 Estuarine pipefish *Syngnathus watermeyeri*

This species is a critically endangered (IUCN listed), range restricted estuarine-spawning species. There is some evidence of spring spawning (Whitfield 1998). Presence in the East Kleinemonde Estuary is thought to be totally dependent of submerged macrophytes. Previously only known from Kariega, Bushmans and Kasouga estuaries and thought to have left these estuaries due freshwater abstraction (hypersaline conditions) leading to a collapse in food resources (zooplankton stock). They were recorded in the East Kleinemonde from 1995 to 2002, when macrophytes beds (both *Ruppia* and *Potamogeton*) were present. Subsequent to May 2003 flood no estuarine pipefish have been recorded. Cowley & Whitfield (2001) estimated a population size of about 4000 individuals with a total biomass of about 2 kg.

In the model simulations, growth and mortality rates were assumed to be linked to zooplankton abundance in accordance with the graphical functions displayed in Figure 4.44 and Figure 4.45, while carrying capacity was assumed to be linked to macrophytes abundance on accordance with the graphic function displayed in Figure 4.46.

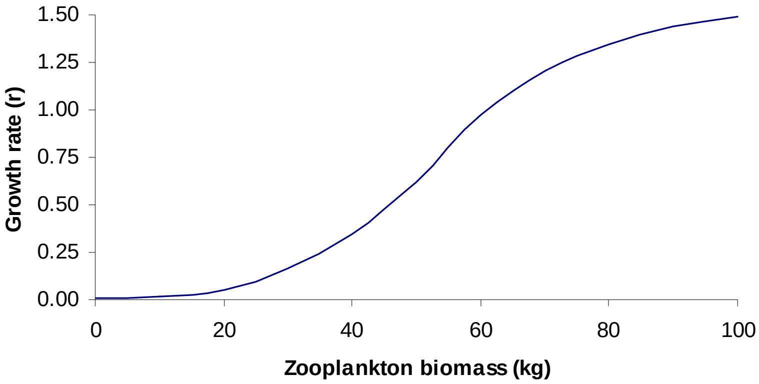

Figure 4.44. Relationship between population growth rate (r) for *S. watermeyeri* in the East Kleinemonde Estuary and zooplankton biomass

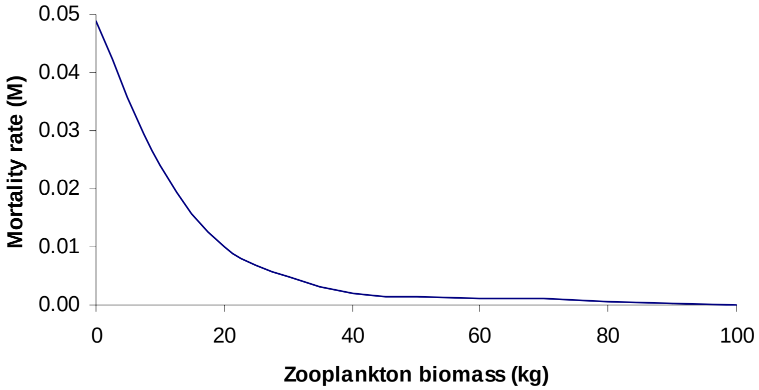

Figure 4.45. Relationship between mortality rate (M) for *S. watermeyeri* in the East Kleinemonde Estuary and zooplankton biomass

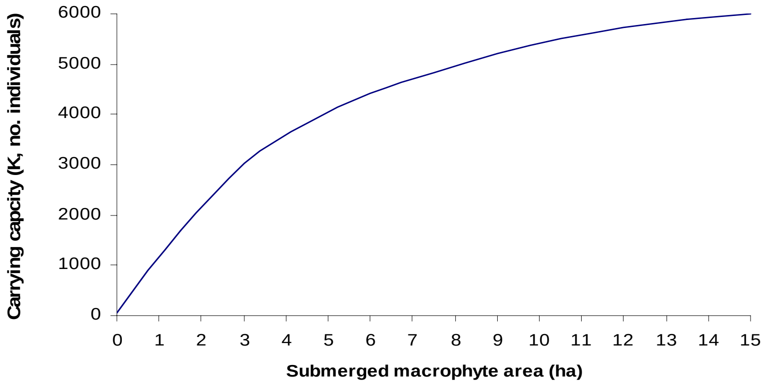

Figure 4.46. Relationship between carrying capacity (K) for *S. watermeyeri* in the East Kleinemonde Estuary and submerged macrophytes area.

4.7 Birds

4.7.1 Background

Little information exists on the avifauna of intermittently open estuaries (IOEs). Several bird counts have included permanently-open estuaries as well as IOEs (Ryan & Cooper 1985, Ryan *et al.* 1988, Turpie 2004, Underhill & Cooper 1984) but the only detailed studies on estuarine birds in IOEs focused solely on the piscivorous component of the bird population (Blaber 1973, Cowley 1998, Terörde 2005). Estuarine birds were counted regularly at the East Kleinemonde

Estuary from March 2005 to November 2006, in a total of 72 counts. Bird abundance on the estuary over the modelling period ranged from 21 to 144 individuals with a mean of 63.8.

Some 45 species of waterbirds have been recorded at the estuary. Piscivorous birds made up the dominant component of the avifauna in terms of abundance (66.4% of all recorded individuals) and diversity (22 species). The bird species were be divided into groups according to their feeding group, migration pattern and habitat(Table 4.5). Species with only one or two records (African Finfoot, Cape Shoveller) and species not dependant on aquatic habitat were disregarded (Hamerkop, Sacred Ibis, Egyptian Goose).

Table 4.5. Bird groups used in the model and the species included in each group

Group	**Species**
Resident diving piscivores	African Fish Eagle, Giant Kingfisher, Half-collared Kingfisher, Malachite Kingfisher, Pied Kingfisher
Migratory diving piscivores	Caspian Tern, Common Tern, Little Tern
Pursuit swimming piscivores	African Darter, Cape Cormorant, Little Grebe, Reed Cormorant, White-breasted Cormorant
Wading piscivores	African Spoonbill, Black Stork, Goliath Heron, Great Egret, Grey Heron, Little Egret, Purple Heron
Resident (non-migratory) invertebrate feeding waders	African Black Oystercatcher, Black-winged Stilt, Kittlitz’s Plover, Three-banded Plover, Water Thick-knee
Migratory invertebrate feeding waders	Common Greenshank, Common Ringed Plover, Common Sandpiper, Common Whimbrel, Curlew Sandpiper, Grey Plover, Little Stint, Marsh Sandpiper, Ruff, Ruddy Turnstone, Sanderling, Wood Sandpiper
Herbivorous waterfowl (feeding on aquatic vegetation)	Red-billed Teal, South African Shelduck, Yellow-billed Duck

4.7.2 Sub-model overview

The bird submodel simulates changes in abundance of estuarine bird species in relation to the main driving factors that influence population dynamics. These were identified as season, water level and mouth status.

4.7.3 Resident diving piscivores

This group is made up of kingfishers and the African Fish Eagle which hunt by diving into the water from a perch or the air. It was found that a relationship between their abundance and water levels in the estuary exists. Abundance was highest at water levels of 1 – 2 m above mean sea level, but lower at lower and higher water levels. During open mouth phases the mean number of individuals recorded was 11.4.

In the model the number of diving piscivores present at the estuary was described as a function of water levels during closed mouth phases (Figure 4.47). The number of individuals present during open phases was set to their mean recorded number of 11.4.

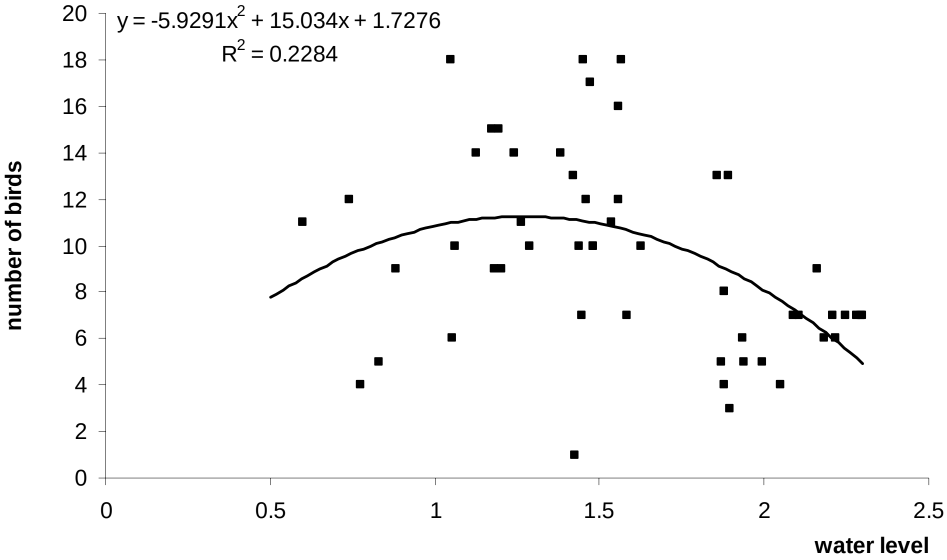

Figure 4.47. Resident piscivorous diving birds recorded at the East Kleinemonde Estuary at different water levels between March 2005 and November 2006 during closed mouth phases.

4.7.4 Migratory diving piscivores

Migratory terns occurred in low numbers at the estuary expect immediately after a breaching or overtopping event when recruiting fish may attract them in high numbers for a short period. They did not show a relationship with changes in water levels. Their abundance was therefore set as the mean recorded population size for summer (1.43 individuals) and for winter (0.41 individuals), but a multiplier was applied which was related to the quantity of recruiting fish.

4.7.5 Pursuit swimming piscivores

This group responded primarily to mouth status and changes in water levels. The mean number of birds recorded during open mouth phases was 20.4, while a mean of 11.8 was recorded for closed mouth phases. The number of cormorants and darters recorded during closed mouth periods increased with increasing water levels. These birds feed by pursuing and catching their prey under water and therefore respond positively to higher water levels which increases the area in which they are able to hunt.

In the model the population size of cormorants was made a function of water level and mouth state. The expected population size during an open mouth state was set to the mean recorded number of 20.4. The population size during closed mouth phases was made a function of the water level as shown in Figure 4.48. Summer and winter values for closed mouth state were lumped together as they showed a similar trend.

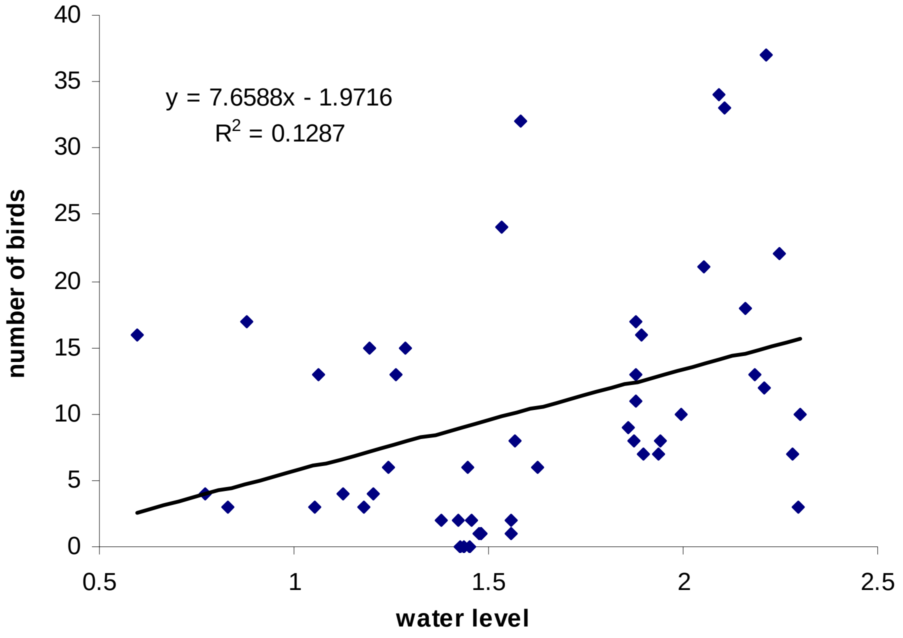

Figure 4.48. Pursuit swimming piscivorous birds recorded at different water levels (meters above mean sea level) at the East Kleinemonde Estuary during closed mouth phases from March 2005 to November 2006.

Little Grebe occurred sporadically and were only recorded on eleven occasions at the East Kleinemonde Estuary. While this species feeds primarily on fish and aquatic invertebrates it is believed to be associated with reed beds and submerged macrophytes (Cowley, pers. obs.) They have occurred regularly and in higher numbers at this estuary 10 years ago when there were more submerged macrophytes present in the system (P. Cowley, pers. comm.). When present their numbers were higher with higher water levels, but due to their low occurrence and abundance a clear trend can not be proven. They were only recorded once during an open mouth phase and this was one day after a breaching event.

In the model Little Grebe were separated from the other diving piscivores. Their abundance was made a function of submerged macrophyte availability and mouth state. They are expected to increase with increased submerged macrophyte availability during closed mouth phases. Their abundance values were set to the recorded mean during open and closed mouth phases.

4.7.6 Wading piscivores

Water level was identified as the primary driving factor in the abundance of wading piscivorous birds. These birds feed in shallow water of approximately less than 30cm depth. Their numbers decreased with increasing water level (Figure 4.49), as less shallow water was available for feeding. The function in Figure 4.49 was used in the model.

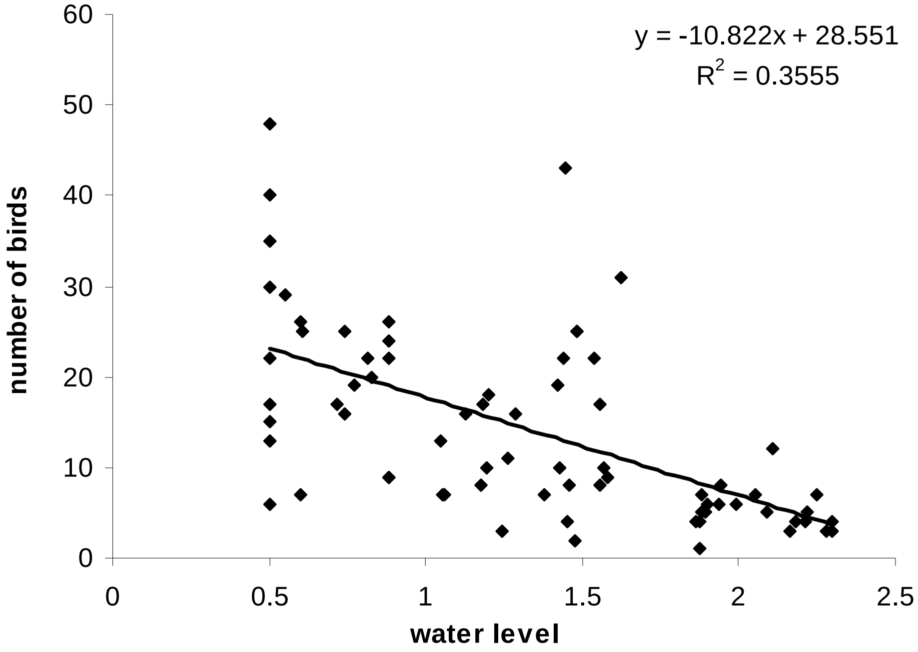

Figure 4.49. Wading piscivorous bird numbers recorded at different water levels (meters above mean sea level) at the East Kleinemonde Estuary from March 2005 to November 2005.

4.7.7 Non-migratory waders

Wader species that do not show a seasonal migration out of South Africa were more abundant in winter (Mean: 11.0) than in summer (Mean 5.1). They also showed a response to changes in water level, with a decrease in abundance with decreasing water levels. Their abundance was therefore modelled as a function of water levels and season (Figures 9.4 & 9.5).

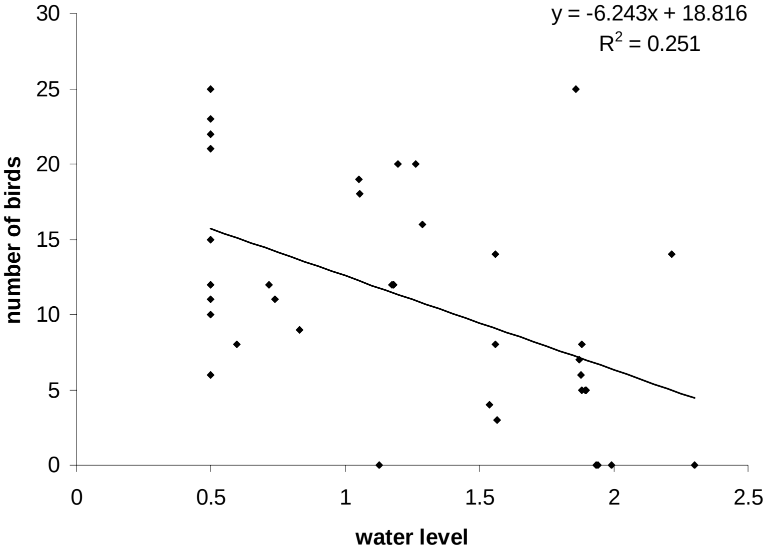

Figure 4.50. Non-migrating waders recorded in the winter months (April – August) of 2005 and 2006 at the East Kleinemonde Estuary at different water levels.

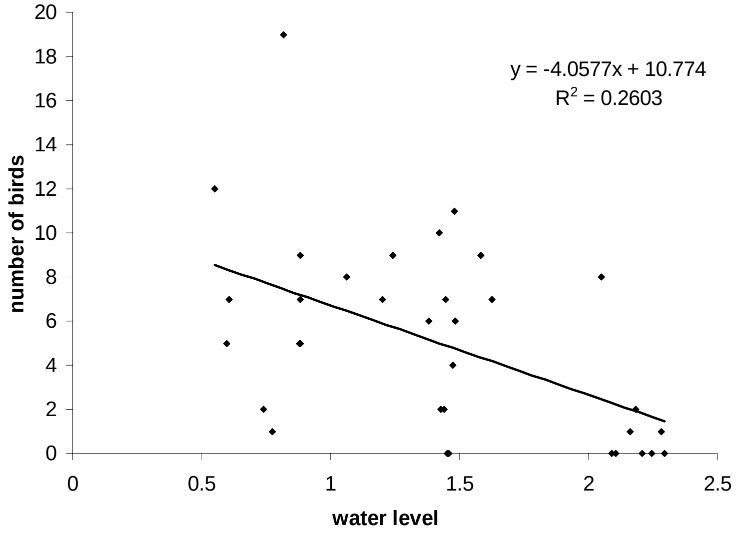

Figure 4.51. Non-migrating waders recorded in the summer months (September – March) of 2005 and 2006 at the East Kleinemonde Estuary at different water levels.

4.7.8 Migratory waders

This group of birds consists of Palearctic breeding migrants that occur in South Africa from September to April. These species mostly occur in intertidal habitat and on exposed mudbanks (Hockey *et al.* 2005) and do not frequent intermittently open estuaries regularly or in great numbers (Perissinotto *et al.* 2004). At the East Kleinemonde Estuary migratory waders were recorded in low numbers throughout the summer months of 2005 and 2006, with occasional small flocks of Curlew Sandpiper and Greenshanks occurring. The maximum number of recorded birds was 50. The mean number of birds recorded during the summer months (September – March) was 8.7. There were two isolated records of a Greenshank and a Common Sandpiper in the winter months. Due to the low abundance and occurrence of this group at the estuary no relationships between wader numbers and water levels could be determined.

The sector for this group therefore predicts migratory wader abundance according to a seasonal curve with a maximum of the mean number of birds recorded in the summer months and a minimum of zero in the winter months.

4.8 Management

The model contains a management submodel which allows the user to decide on whether artificial breaching will take place, and the rules for breaching. This submodel will be linked up during the final phase of the project.

4.9 RDM (estuary health) indicators

The RDM Indicators sub-model produces average values over the total run period for parameters that are needed to calculate the scores used to indicate estuary health under the Resource Directed Measures (RDM) methodology.

4.10 Total economic value

The Total Economic Value sub-model (Figure 4.52) produces an estimate of the total Net Present Value (NPV) of the ecosystem services provided by the estuary in 2007 rands. The NPV is calculated as the sum of the discounted values over the total run period of the model. The discount rate can be varied on the Interface layer of the model. The total NPV is the sum of property sector and tourism sector turnover ascribed to the estuary, as well as nursery value and existence value. The values and relationships are based on an economic study of the estuary (Turpie et al. 2007). For recreational and aesthetic value, utility is calculated as a function of fish abundance, water quality, water level and vegetation. Water quality (from a human use perspective) was not estimated in the model nor studied during the research programme. Thus water quality was assumed to begin deteriorating if the estuary had been closed for long periods. Nursery value was calculated as the value of fish exported from the estuary in the fish model. Existence value was based on the average population size of Estuarine Pipefish, and hence its probability of extirpation.

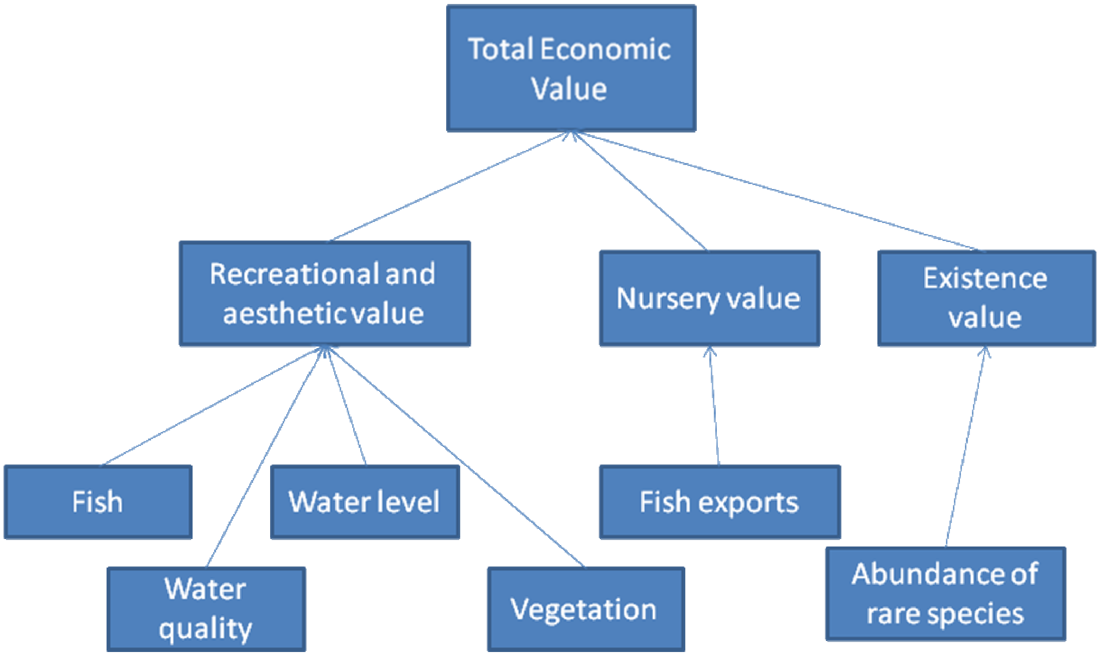

Figure 4.52. Schematic diagram of the Total Economic Value submodel

5 MODEL RESULTS 1992-2000

5.1 Physical dynamics and water quality

Model simulation results from the Physical Dynamics sub-model for the period 1 January 1992 – 31 December 2000 are presented in Figure 5.1 and Figure 5.2 together with daily observations on mouth state over the same period. The level of congruence between the observations of mouth condition and the model simulation is very good. Nearly all the actual opening events appear on or close to their actual date of occurrence in the simulation results, with only a small number of opening events being missed (e.g. three opening events in the period December 1993 to March 1994) and few additional unrecorded events being simulated (e.g. two opening events in the period October 1999 to March 2000). Days during which overwash was reported were also closely matched with marine inflow events in the model (Figure 5.1 and Figure 5.2). These incidences generally corresponded with periods of high waves and or elevated sea levels (i.e. spring tides).

Table 5.1. Comparison between model results and actual observations of mouth state in each in the period Mar 1993-December 2000. Data shown are frequency of occurrence (in days) in each of three mouth states.

	Observed	**Modelled**
Closed	2321	2219
Overwash	467	514
Open	73	129

Comparisons between the frequency of occurrence of different mouth states in the observed data and model results also show a high level of congruence between the modelled and observational data. Over the 93 month period for which observation data were available, number of days for which the mouth was open in the model was only marginally fewer than observational data reported, days for which overwash was reported were slightly lower than in the model simulations , while days open were also higher in the model simulations than in the observational data.

Observational records are not available for any of the other modelled parameters. Results for the remaining parameters are thus not shown for the full eight year period but rather just for a limited period covering the full range of conditions experienced within the system over the eight-year model period. The period selected for this purpose covers the time from 1 September 1996 – 31 January 1999. Model simulation results for the two water quality variables included in the model - salinity and DIN – are shown in Figure 3.1Figure 5.3 together with simulated changes in water level in the estuary, and freshwater and marine inflows, for the period under consideration.

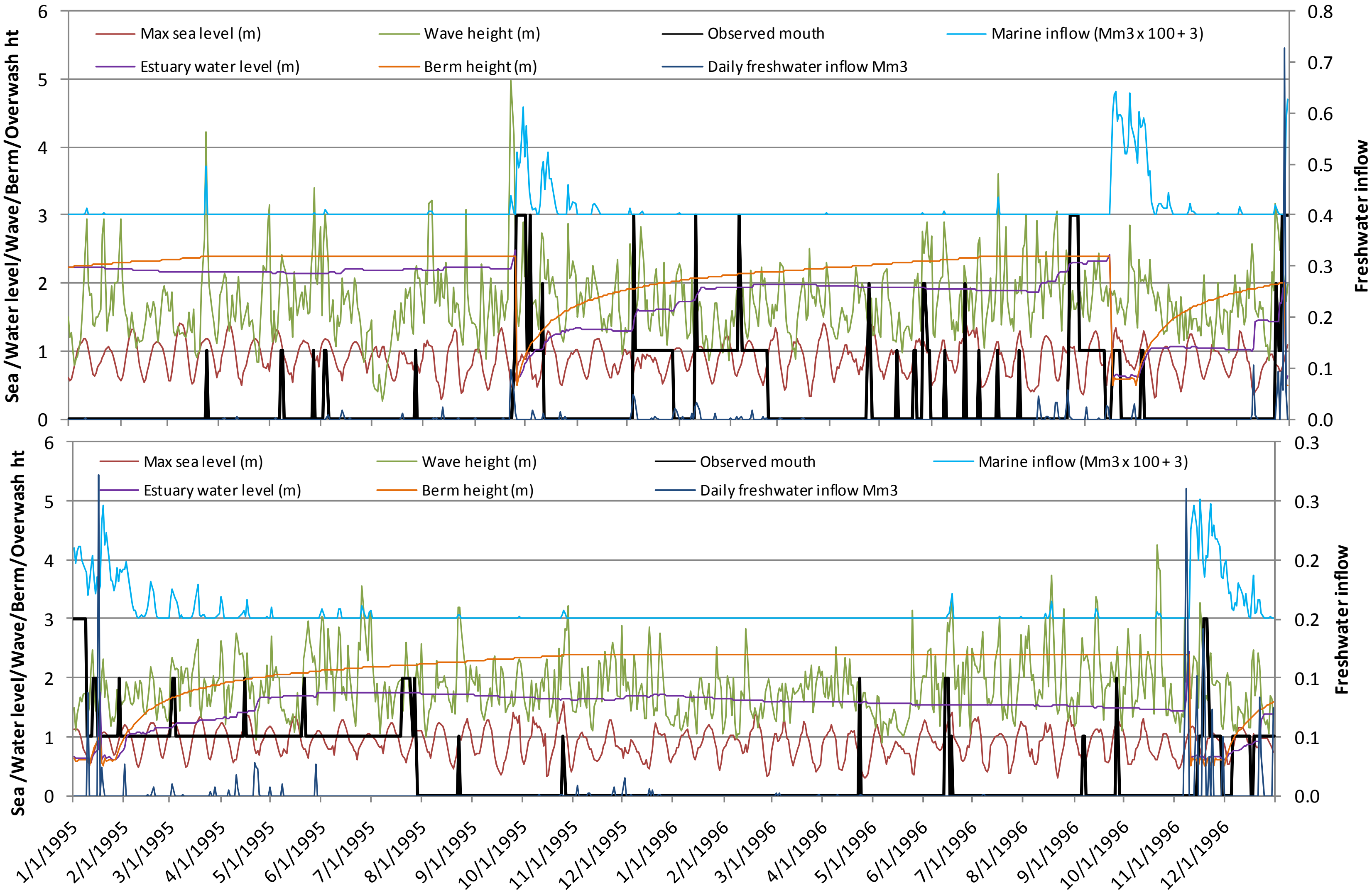

Figure 5.1 Results from the East Kleinemonde estuary physical model compared with observed mouth condition (Jan 1994-Dec 1996). Note that freshwater inflow is plotted on a separate axis from the other variables and overwash height is exaggerated 100-fold and elevated by +4m for improved clarity. Note also that a simultaneous drop in berm height and water level denotes a breaching event in the model data. Data for observed mouth condition are coded in the following manner: 0=closed, 1 = small overwash, 2 = large overwash, and 3 = open.

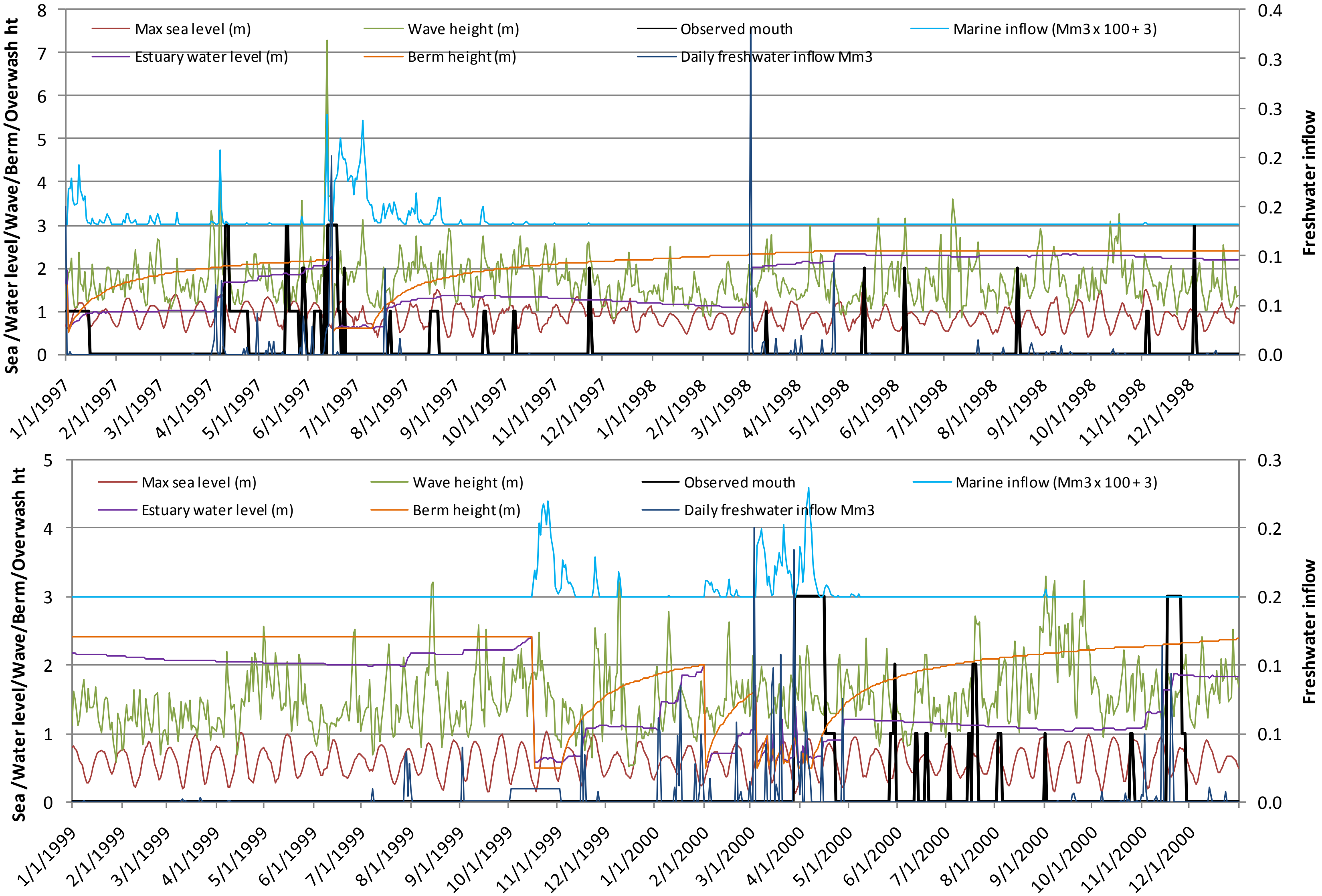

Figure 5.2 Results from the East Kleinemonde estuary physical model compared with observed mouth condition (Jan 1997-Dec 2000). Note that freshwater inflow is plotted on a separate axis from the other variables and overwash height is exaggerated 100-fold and elevated by +4m for improved clarity. Recorded mouth condition is coded as: 0=closed, 1 = small overwash, 2 = large overwash, and 3 = open.

The beginning of the period in question corresponds to the end of a prolonged period for which the mouth was closed (22 months – Feb 1995 to Nov 1996) and for which very little overwash was evident. Salinity is relatively high at this time (25-27 PSU) and gradually rising due to evaporative water loss from the estuary (Figure 5.3).

Salinity drops abruptly with the introduction of a pulse of freshwater into the estuary on around the 9th of November 1996 but recovers rapidly when the mouth opens and seawater flows into the estuary. This is followed by a period when salinity in the estuary oscillates up and down with alternating pulses of freshwater and marine overwash before the mouth is breached again on around 4 January 1997. Salinity variations during the remainder of the simulation period shown in Figure 5.3 continue in a similarly variable manner, increasing as a result of evaporation and with inflows of sea water, and decreasing when freshwater pulses enter the estuary and raise water levels in the system.

Average salinity across the entire simulation period was 17.7 PSU, which corresponds well with observation made by Bernard McKenzie (P. Cowley, unpublished data) for the period 7 March 2005 to 4 October 2006 (average of 55 observations = 22.7 PSU).

Dissolved Inorganic Nitrogen (DIN) concentrations in the estuary during the simulation period shown in Figure 5.3 are mostly low (<200 µg.l-1) for the first half of the period except for a single spike in DIN associated with a large pulse of freshwater that entered the estuary on around 9 November 1996. DIN levels were restored to the modest levels observed prior to this time due to a breaching event that followed, which allowed these nitrogen rich water to flow out and nutrient poor seawater to replace it. DIN levels spiked again later in the simulation period (around 1 March 1998) as another large pulse of freshwater entered the system. DIN levels in the estuary remained high following this breach sustained by continuous small inputs of nitrogen rich freshwater.

Data on DIN concentrations in the Kleinemonde estuary are available for only two days and are of little help for validating the model results.

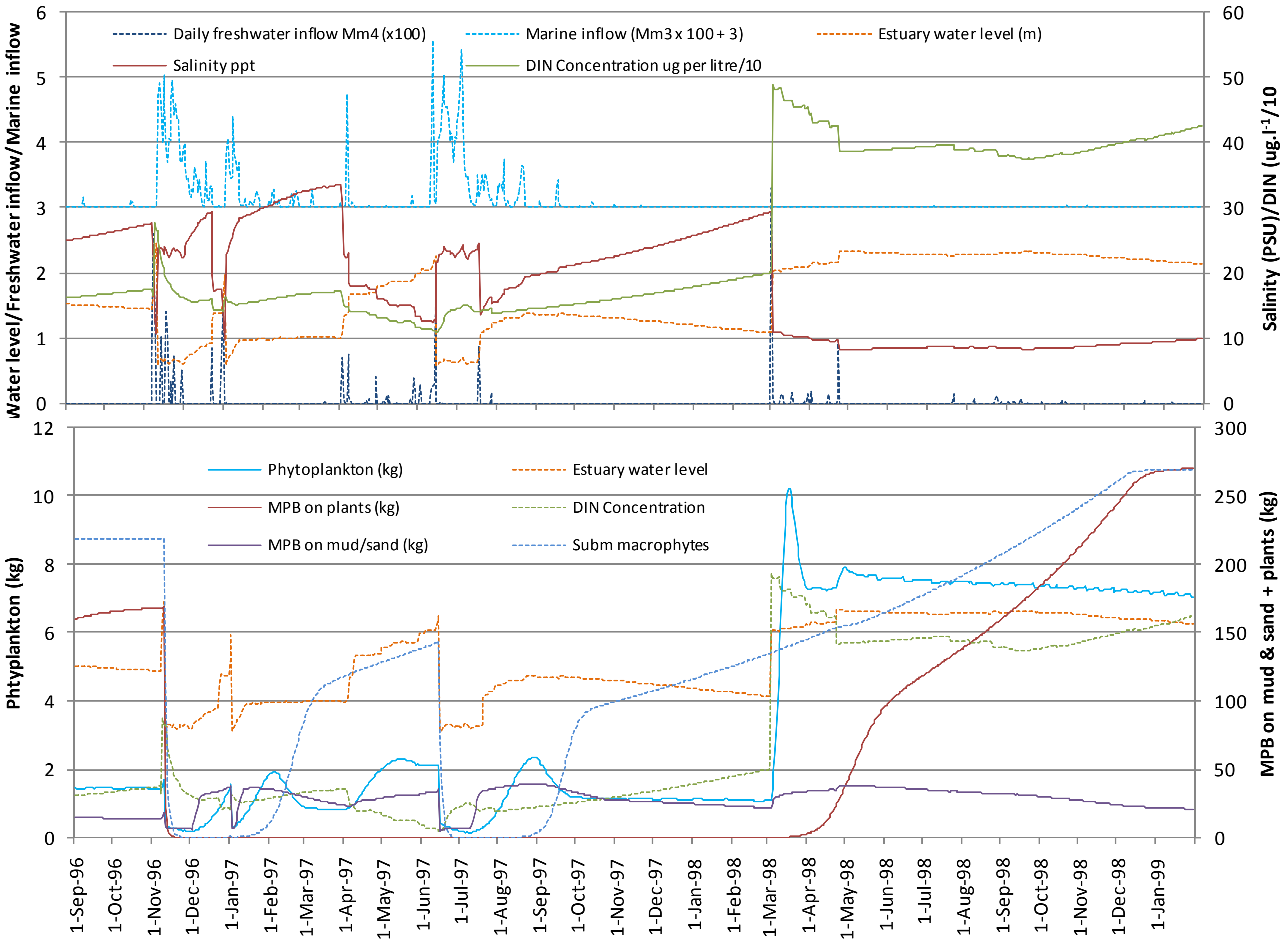

Figure 5.3. Model simulations of salinity and dissolved inorganic nitrogen (DIN) concentration (top) and phytoplankton, microphytobenthos (MPB) on sand mud and on plants biomass (bottom) in the East Kleinemonde estuary in relation to water level, freshwater inflow and marine inflow (top) and water level, DIN concentration and submerged macrophyte cover (bottom) over the period 1 September 1996 – 31 January 1999. Only trends in the latter variable are shown rather than absolute numbers.

5.2 Microalgae

Microalgae in the model were represented by three separate stocks – phytoplankton, microphytobenthos (MPB) on sand and mud and MPB on plants (primarily submerged aquatic macrophytes. These stocks are plotted against their key drivers, DIN, water level and submerged aquatic macrophyte cover for the period 1 September 1996 to 31 January 1999 in Figure 5.3 As might be expected, simulated changes in phytoplankton biomass closely tracked changes in DIN concentration and water level, increasing as DIN concentrations increased and as water level (and hence water volume) rose. MPB on sand and mud showed a similar response (Figure 5.3), closely tracking changes in DIN concentration and water level (a proxy for available habitat for this group). Biomass of MPB on mud and sand remained low (<50 kg) throughout the period in question. MPB on plants exhibited a more complex response in the simulation period shown in Figure 5.3. Biomass was relatively high at the start of the simulation period (160-169 kg), corresponding to a high submerged macrophyte biomass, but remained at very low levels for a long time following the first breach limited mainly by low levels of DIN in the system. Stocks of MPB on plants recovered again in March 1998 corresponding with a dramatic increase in DIN concentrations in the system. Biomass peaked at 270 kg at the end of the end of the period of observation.

5.3 Macrophytes

Macrophyes in the East Kleinemonde estuary were represented by four separate stocks in the model – submerged aquatic macrophytes, reeds and sedges, intertidal saltmarsh and supra-tidal saltmarsh. Variations in the aerial cover by these groups over the period 1 September 1996 to 31 January 1999 is shown in Figure 5.4. Submerged aquatic macrophyte cover tracked variations in water level very closely, dying back during breaching events, but recovering rapidly thereafter to a level corresponding to the extent of the seedbank that remained alive following the breaching event, thereafter increasing at a more modest rate until the next breaching event or until the maximum aerial extent was attained.

Intertidal saltmarsh also tracked variations in water level very closely, but in an inverse manner to the submerged aquatic macrophytes. Intertidal saltmarsh tends to flourish when the estuary mouth is open and salinity levels are high, and occupies mush of the same habitat area as the latter group.

Supratidal saltmarsh and reeds and sedges showed little variation during the observation period (or indeed during the entire simulation period) only dying back slightly at very high water levels and recovering when water levels declined again.
Invertebrates were represented by a large number of different groups in the model, of which only three (zooplankton, *Callianassa* and *Hymenosoma*) are presented separately here, the rest being grouped as "small invertebrates". Drivers and hence responses of taxa in the latter groups were all similar and did not warrant being presented individually.

Variations in zooplankton biomass over the period September 1996 to January 1999 closely tracked its main driver, phytoplankton biomass, with peaks in biomass lagging behind one another latter by as much as three weeks as might be expected under a classic predator-prey response model (Figure 5.4).

Variations in the biomass of *Hymenosoma* and the other small benthic invertebrates all tracked water level closely increasing and decreasing in concordance with one another (Figure 5.4).

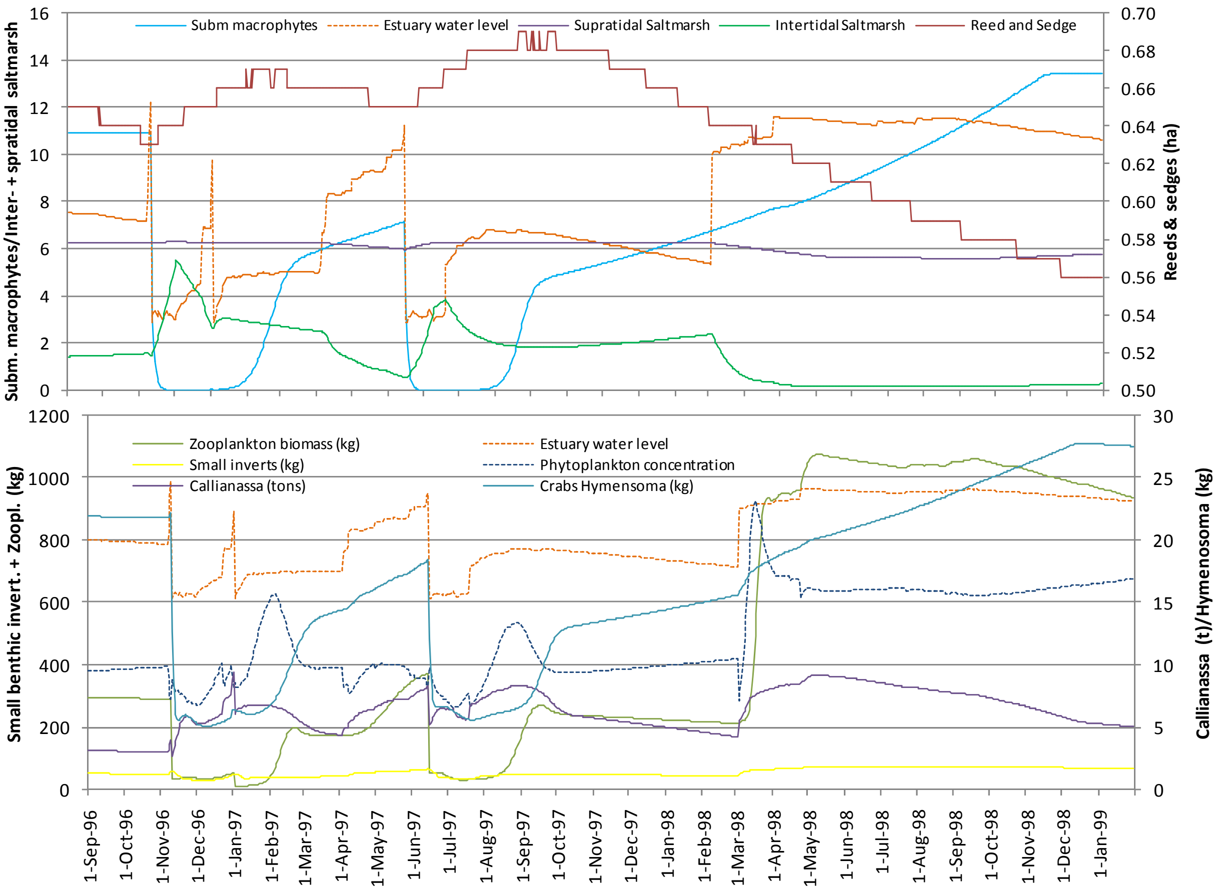

Figure 5.4. Model simulations of macrophyte cover (top) and invertebrate populations (bottom) on the East Kleinemonde estuary in relation to water level and phytoplankton concentration (bottom only) over the period 1 September 1996 – 31 January 1999. Only trends in the latter variables are shown rather than absolute numbers.Invertebrates

5.4 Fish

Fish were represented in the model by the seven most common species/groups recorded in the estuary, four of these being marine migrant species and three being estuarine resident (breeding) species. Marine migrants include Cape stumpnose *Rhabdosargus holubi*, white steenbras *Lithognathus lithognathus*, kob *Argyrosumus inodorus* and various species of mullet. Estuarine residents include estuarine round herring Gilchristella aestuaria, estuarine pipefish Syngnathus watermeyerii, and and various species of goby. Variations in the abundance of the marine migrant species were plotted in relation to water level, marine inflow and submerged macrophyte abundance, while the estuarine residents were plotted in relation to water level and submerged macrophyte abundance only (Figure 5.5). Cape stumpnose and mullet are able to recruit during overwash and mouth open events and show clear responses to both types of events, with the inflow of recruits bolstering population numbers at these times. White steenbras and kob by contrast are able to recruit during mouth open events only and show a much reduced variations in abundance, especially during the latter part of the simulation period when the mouth remains closed for a prolonged period. Of further interest is the fact that both white steenbras and kob can be seen recruiting into the estuary during mouth open events during the early part of the simulation period (Nov-Feb) while only kob shows a recruitment response when the mouth opens later in the year when there are no steenbras recruits are available to enter the estuary.

Variations in abundance of round herring closely followed trends in mouth state and zooplankton biomass; gobies simply tracked changes in water level (linked to habitat availability for this species), and pipefish tended to track variations in macrophyte cover.

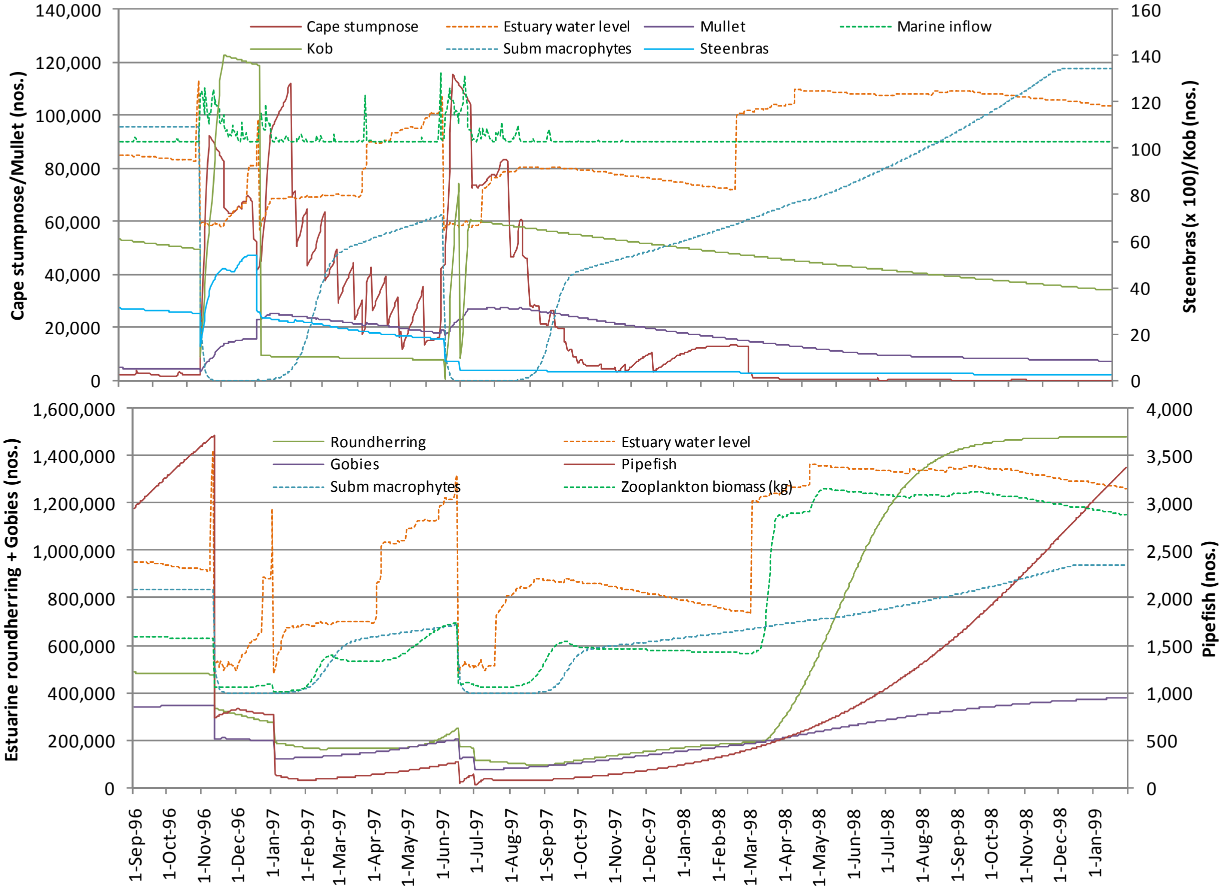

Figure 5.5. Model simulations of estuarine resident fish (top) and marine migrant fish (bottom) populations in the East Kleinemonde estuary in relation to water level, marine inflow, zooplankton biomass and submerged macrophyte cover over the period 1 September 1996 – 31 January 1999. Only trends in the latter variable are shown rather than absolute numbers.

5.5 Birds

The four piscivorous groups of birds - migratory terns, cormorants, grebes and resident diving piscivores (kingfishers, fish eagles and gulls) have been plotted in relation to water level, marine inflow and submerged aquatic macrophytes (Figure 5.6). Cormorants and migratory terns closely follow abundance of newly recruited fish in the estuary, peaking during mouth open and overwash events. The resident diving piscivores tend to track water level in the estuary, while the grebes follow submerged macrophyte cover (Figure 5.6). Waders increase in abundance during the open mouth periods. Herbivorous water, as their name implies, feed on aquatic macrophytes and track the abundance of this resource in the model simulations (Figure 5.6).

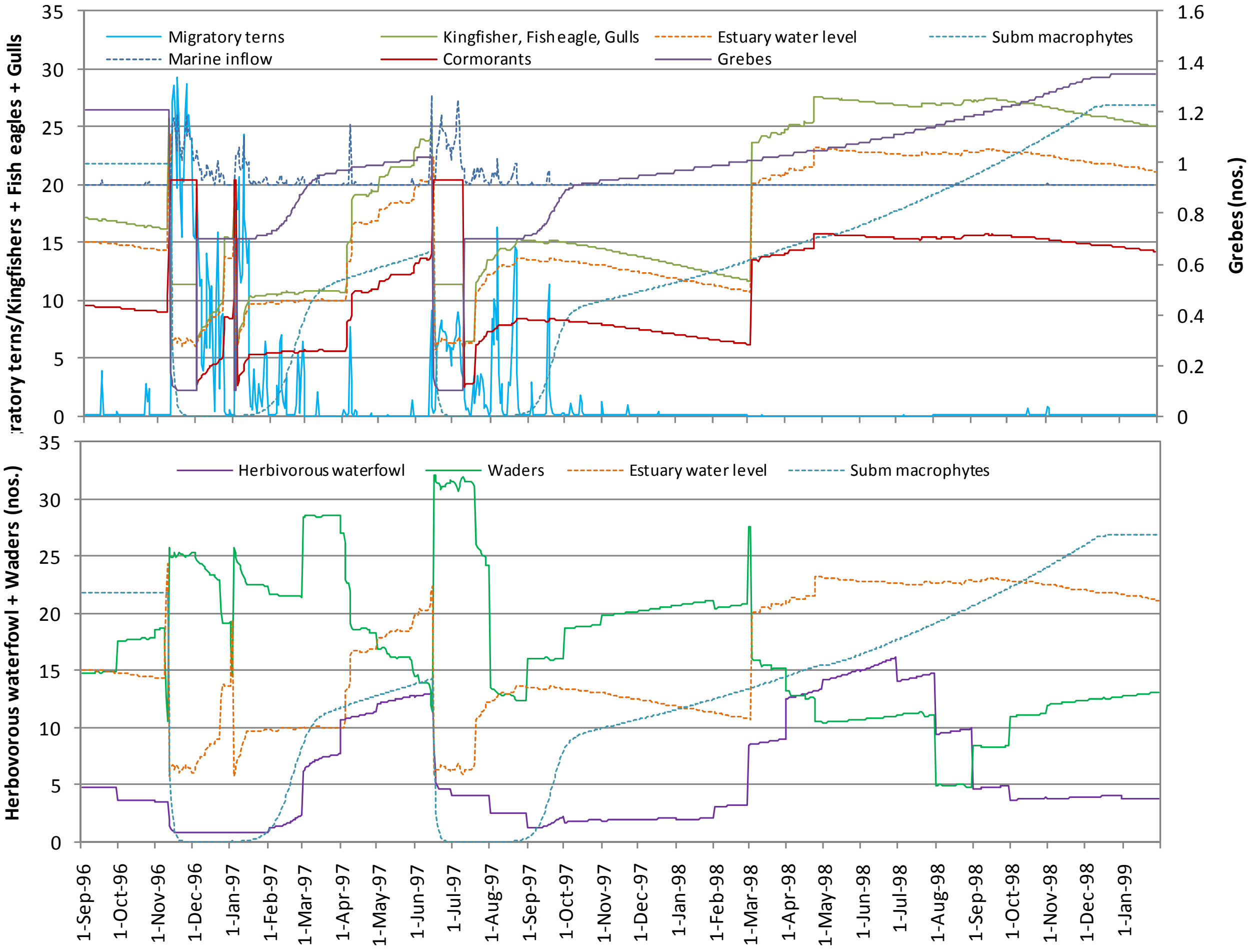

Figure 5.6. Model simulations of piscivourous bird (top) and other bird group (bottom) populations on the East Kleinemonde estuary in relation to water level and submerged macrophyte cover over the period 1 September 1996 – 31 January 1999. Only trends in the latter variable are shown rather than absolute numbers.

5.6 RDM outputs

The results of the health scores used in RDM are summarised in Table 5.2 (physical) and Table 5.3 (biotic). There was a fair degree of congruence. Much of the difference between the modelled score and the expert score can be attributed to the anthropogenic influence. For example the relatively low score for bird abundance was attributed entirely to anthropogenic disturbance. These data will be analysed in more detail in the next phase of the study.

Table 5.2. Physical scores generated by the model for present state and compared with the scores generated by expert opinion for the RDM study and the confidence levels associated with those scores.

Physical parameters	**Modelled score**	**Expert score**	**Confidence**
Mean duration of closure (days/y)	100	90	L
Salinity gradient (mean salinity)	88	90	L
Water quality - N/P (DIN)	84	70	L
Intertidal area	98	85	M
Proportion sandy	Not modelled	75	M
Subtidal channel morphology	Not modelled	90	M
Infrastructure, human disturbance	Not modelled	80	M

Table 5.3. Biotic scores generated by the model for present state and compared with the scores generated by expert opinion for the RDM study and the confidence levels associated with those scores.

Biotic parameter		**Modelled score**	**Expert score**	**Confidence**
Phytoplankton	Abundance	82.9	95	M
	Community comp	Not modelled	95	M
Benthic microalgae	Abundance	63.7	90	M
	Community comp	Not modelled	90	M
Macrophytes	Abundance	96.4	85	M
	Community comp	95%	95	M
Zooplankton	Abundance	90.6	95	M
	Community comp	Not modelled	95	M
Benthic inverts	Abundance	92.7	95	M
	Community comp	98%	95	M
Fish	Abundance	77.1	90	M-H
	Community comp	93%	90	M-H
Birds	Abundance	97.8	85	M
	Community comp	95%	95	M

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