MODELLING THE PHYSICAL DYNAMICS OF ESTUARIES FOR MANAGEMENT PURPOSES

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ABSTRACT

South African estuaries are characterised by highly variable inflows owing to the semi-arid nature of the land mass which they drain. The interaction of this variability with that of the marine environment (seasonality, high wave events, synoptic effects) gives rise to the distinctive character of South African estuaries. In general, they are small, micro-tidal, bar-built systems with strong flood tidal dominance. Approximately half of the 273 systems along the coast exhibit intermittent closure of the mouth, while a number can become hypersaline during dry periods. In view of the increasing development pressures on the rivers and estuaries of South Africa and their strong dependence on freshwater flow for the maintenance of their character and functioning, and the need for justifiable, scientifically-based decision making regarding the freshwater requirements of estuaries is evident.

This study was initiated to address this issue by first developing a model to simulate the physical dynamics of South African estuaries over time scales from months to years, so enabling prediction of the medium to long term consequences of alterations in the freshwater inflow on the abiotic components of an estuary. Thereafter, the efficacy of management policies involving water releases and mouth breachings could be evaluated in terms of their success in maintaining the character and functioning of an estuary.

A semi-empirical estuarine systems model incorporating seven state variables, namely water volume, salt content, stratification, circulation, tidal flushing, freshwater flushing and the height of the sill at the mouth, was formulated and implemented on two case studies. Estuarine physics concepts were incorporated dynamically in the model in a novel manner. For instance, the bulk densimetric Froude number and the Estuarine Richardson number are used in the simulation of the stratification-circulation states, while the Ackers and White sediment transport formula was modified to yield results which agreed with field observations of the closure and breaching of the mouth of the Great Brak Estuary. Additionally, tidal exchange through the mouth was modelled phenomenologically and successfully calibrated against observations for both case studies. Model results were found to be fairly robust to uncertainties in parameter values. However, most encouraging of all is that behaviour known to occur in shallow estuaries, such as modulation of the mean water level by low frequency forcing and the generation of overtides, was reproduced

by the estuarine systems model although it was not specifically included in the model formulation. The model is thus considered to reliably predict the physical dynamics of South African estuaries over time scales of months to years.

A number of management policies involving freshwater allocations, water releases and breachings of the mouth (where appropriate) were tested on the two case studies, namely the Great Brak Estuary, a small, temporarily open system, and the permanently open Kromme Estuary. The results indicate an increase in marine dominance as freshwater flow to the estuaries decreases. The variability in the estuarine environment declines and the systems become more inert to freshwater flooding and more sensitive to marine forcing. By applying the estuarine systems modelling approach, the performance of different management policies could be evaluated in comparison with reference policies. Accordingly, for both case studies, preferred management policies which utilize the present total annual allocations to the estuaries more beneficially could be indicated. Further management applications included the use of the estuarine systems model in a linked system of abiotic and biotic models to facilitate more comprehensive prediction of the consequences of freshwater abstraction and so more informed assessment of estuarine freshwater requirements. The estuarine systems model results were critical in enabling the prediction of the faunal and floral responses in the intermittently closed Great Brak Estuary as it is presently the only abiotic model capable of simulating the closure and breaching of the estuary mouth over a number of years. It is anticipated that further developments will occur in biological prediction in the near future and that this could require developments or adaptations to the estuarine systems model, particularly when details of the type of information required for biological prediction becomes known. Additionally, the use of the estuarine systems model in a strategic management sense is suggested. It could play a role as a screening tool for regional water resource planning, while the preliminary quantification of the extent of anthropogenic influence in expediting the movement of estuaries towards the later successionary stage of a coastal lagoon is a powerful indication of the level of prediction which could become possible in the future. Thus enhanced management decision making is now possible on a site specific basis and at a more strategic water resources planning level.

PREFACE

The research work described in this thesis was conducted under the supervision of Prof J W Hearne of the Department of Mathematics and Applied Mathematics, University of Natal, Pietermaritzburg, from May 1990 to September 1996, while the author was employed by the CSIR, Stellenbosch.

These studies represent original work by the author and have not otherwise been submitted in any form to another University. Where use has been made of the work of others it is duly acknowledged in the text.

ACKNOWLEDGEMENTS

> Not that I am competent in myself to claim anything for myself, but my competence comes from God. He has made me competent as a minister of a new covenant - not of the letter but of the spirit; for the letter kills, but the spirit gives life.
>
> 2 Corinthians 3: 5,6

I owe a debt of gratitude to Prof Charles Breen and Prof John Hearne for introducing me to the field of environmental management and to Dr John Largier for introducing me to estuarine science - my thanks to each of you. Additionally, members of the Consortium for Estuarine Research and Management are thanked for many stimulating discussions and their enthusiasm and interest in the modelling work and its application to South African estuaries. Prof John Hearne, the supervisor of this thesis, is thanked for his support and patience throughout the study.

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1. INTRODUCTION

This study was motivated by a concern to improve the management of South African estuaries through enhanced understanding of their functioning and dynamics. Fundamental to the study is a modelling approach focussed on the physical environment of estuaries. The reasons for adopting this approach, and the issues leading to the inception of the study, will be explained through consideration of the following questions:

- What is an estuary?
- What are the management issues associated with South African estuaries?
- Why model?
- Why model the physical dynamics?
- What implications does this study hold for estuarine management in South Africa?

In this process the particular objectives of the study and the significance of the work will be addressed.

1.1 What is an Estuary?

1.1.1 Definition of an estuary

The definition of an estuary has long been the subject of debate (Cameron & Pritchard 1963, Lauff 1967, Pritchard 1967, Dyer 1973, Begg 1978, Noble & Hemens 1978, Heydorn 1979, Day 1980, Reddering 1980, Day 1981). Cameron and Pritchard (1963) defined an estuary as:

> "a semi-enclosed coastal body of water which has a free connection to the open sea and within which sea water is measurably diluted with fresh water derived from land drainage".

In contrast to the majority of European and North American estuaries (the subjects of the bulk of the estuarine literature), which generally have perennial inflow and tidal action, two phrases within this definition pose a problem in the context of South African estuarine systems. These are: *a free connection to the open sea* and *diluted with fresh water*. The mouths of many South African 'estuaries' are prone to closure during periods of low river flow, interrupting the connection to the sea. Further, salinities may exceed those of sea water (hypersalinities) in times of drought or the system may become entirely fresh under river flood conditions. Thus under extremely low or high river flow conditions, 'estuaries' may temporarily exhibit characteristics of estuarine lagoons (Gary *et al.* (eds.) 1974, Begg 1978), marine inlets (little or no fresh water inflow) or river mouths.

To clarify the temporal variability in estuarine state which is associated with the estuaries of arid to semi-arid regions (variable inflow) and/or exposed coasts (variable mouth condition), Day (1980) modified the definition of an estuary to read:

> "a partially enclosed coastal body of water which is either periodically or permanently open to the sea and within which there is a measurable variation of salinity due to the mixture of sea water and fresh water derived from land drainage".

Day (1981) also recognised three types of estuaries, namely: normal estuaries where salinity decreases from the mouth to the head, temporarily hypersaline estuaries and temporarily closed or blind estuaries. This definition excludes tidal inlets with no fresh water inflow (eg. Langebaan Lagoon), is restrictive of systems exhibiting predominant freshness or hypersalinity (despite fresh water inflow), but includes systems subject to periodic mouth closure.

Whitfield (1992) has expanded on the work of Day (1981) and recognises five estuarine types in southern Africa, namely: permanently open estuaries, temporarily open estuaries, estuarine lakes, estuarine bays and river mouths. The Whitfield characterisation, based on estuarine configuration and the associated flow and salinity regimes, excludes tidal inlets with no fresh water inflow, but is not limiting of periodic mouth closure, hypersalinity or predominant freshness.

It is evident that variability in inflow and mouth conditions make a comprehensive definition of an estuary problematic. In this study, the least restrictive use of the term estuary is adopted, namely the definition by Day (1980) as expanded in the characterisation by Whitfield (1992).

1.1.2 **The dynamic character of an estuary**

The location of estuaries at the interface between river and sea leads to variability in the physical environment. On the one hand, oceanic and climatic forcing in the form of tidal variations, wave events, storm surges, offshore currents, coastal upwelling and seasonal and synoptic weather systems influence the water levels, circulation patterns, temperature and salinity structure, nutrient and sediment supply and the state of the mouth of an estuary. On the other hand, variations in insolation, precipitation and drainage from a catchment affect the temperature, volume and nutrient and sediment supply of the riverine input to an estuary. The state of an estuary at any particular time is a composite response to the present and previous physical forcing.

The temporal scale of these influences can vary from hours to decades, while the spatial variation can extend over tens of kilometres within South African estuaries.

The relatively sheltered and shallow waters of estuaries, which receive nutrients and sediment from both land and sea, provide habitats for an array of plant and animal life. Estuarine biota are adapted to cope with variations in salinity, flow velocity, water level and wave and tidal action, and the dynamic abiotic environment of estuaries is associated with highly productive ecosystems (Odum 1971, Clark 1974, Branch & Branch 1981, Knox 1986). Many of the invertebrate and fish fauna occupying an estuary for all or part of their life cycles have developed mechanisms which allow them to use the natural variation in an estuary to aid feeding and/or migration (Day 1981, Whitfield & Kok 1992). Some plankton species, in turn, are dependent on the opposing current patterns associated with ebb and flow and/or surface to bottom stratification in an estuary to position themselves optimally for light penetration and nutrient and food supply (Grindley 1972, 1981, Wooldridge & Erasmus 1980, Hilmer & Bate 1991), while plants colonising the intertidal areas are dependent on tidal variation for regular inundation and exposure (Clark 1974, Day 1981, Adams & Bate 1994). Thus, in its natural state an estuary supports a balanced network of biotic inter-relationships largely based on the interplay between tidal exchange and river flow.

The range of habitats provided by estuaries differs with location within an estuary and from one system to the next, owing to local and regional variation in the physical environment. For example, an area at the head of an estuary is likely to experience less tidal variation than at the mouth, while in the regional context the tidal range may differ along a coast. Thus the variation in physical environment within an estuary and from one estuary to the next results in the unique character of individual estuarine ecosystems.

1.2 What are the Management Issues Associated with South African Estuaries?

It is within the context of increasing regional and local development that decisions regarding the management of South African estuaries need to be made. Although the particular character of individual estuaries necessitates site specific management, the presence of 273 estuaries (Whitfield 1995) along the approximately 2950 kilometre southern African coastline from Namibia to Mozambique, precludes this as an undertaking on economic and logistical grounds. Thus a broader perspective of the issues associated with the management of South African estuaries is required to facilitate the cost-effective development of management strategies on a national and/or regional scale, rather than solely on a local scale.

1.2.1 Distinctive features of South African estuaries

The variability in freshwater flows, the high incidence of mouth closure and the occurrence of hypersalinities have already been mentioned as characteristic features of South African estuaries (Section 1.1.1). Other features generally associated with South African estuaries are:

- the majority are small, microtidal systems with tidal prisms of 10^6 m^3 or less (Reddering & Rust, 1990),
- most estuaries occupy drowned river valleys and do not have the extensive tidal flats usually associated with estuaries that have developed on coastal plains (Reddering & Rust, 1990),
- flood tidally dominated systems are common (Reddering & Rust, 1990),
- the density structure of the water column generally is responsive to short term changing patterns of river and tidal inflow (Allanson 1992), and
- the balance between freshwater and sea water inputs is a determining factor in the ecology of an estuary, i.e. the ecosystem reflects a secondary response to the hydrodynamic and geomorphological character of the estuary (Jezewski & Roberts 1986).

These features contribute to the distinct, yet diverse nature of the estuaries of South Africa compared with the large, deep systems characteristic of the coasts of North America and Europe. Similar small systems are found in the semi-arid areas of southern Australia, California, Mexico, Indonesia and Malaysia (Day 1981, Dr J L Largier *pers. comm.*, Mr P Kerssens *pers. comm.*). However, it is the combination of the particular development pressures on South African estuaries and their distinctive nature which defines the critical issues in estuarine management.

1.2.2 **Local, regional and national development pressures on South African estuaries**

An estuarine ecosystem may be subjected to stress owing to local development. Man has traditionally utilized an estuary as a resource; initially for food, later as a site for settlement, industrial development, waste disposal and recreation. As usage of a system increases, the accompanying impact on the natural functioning of the system generally increases. The loss of mudflats and intertidal areas due to settlement, constrainment of flow due to bridge construction, dredging activities and manipulation of the mouth to suit recreational or commercial purposes eg. commercial fishing, are examples of local development actions which may cause the character of an estuary to change (Begg 1978, Council for the Environment 1991).

However, the functioning of an estuary also may be threatened by more distant (regional or national) development. As an estuary is the recipient of fresh water, nutrients and sediments due to riverine inflow, developments in the catchment resulting in a reduced or altered pattern of fresh water inflow could potentially have severe consequences for a downstream estuary (Whitfield & Bruton 1989, Council for the Environment 1991). In semi-arid southern Africa, where fresh water is a limited resource under increasing demand for domestic, industrial and agricultural usage (DWA 1986), this is a real danger. In fact, investigations of the water supply potential of various southern African catchments are currently underway for the purposes of planning dam constructions to meet anticipated future water demands (DWAF 1992, 1993a, 1993b, 1996).

Presently, implementation of the integrated environmental management procedure (DEA 1992), which facilitates the involvement of all interested and affected parties as well as incorporating specialist environmental inputs, allows informed decision-making on local developments to occur. This is possible primarily because the effects of actions such as discharge of effluent to an estuary, dredging and bridge or marina construction amongst others, are known or may be predicted accurately using hydrodynamic, transport-dispersion and water quality models (CSIR 1984, Najarian & Harleman 1989, Chris Mulder Assoc. Inc. 1991, CSIR 1993a). However, the consequences of regional/national development actions such as dam constructions and inter-basin transfer schemes generally cannot be predicted with accuracy as yet (Whitfield & Bruton 1989, Prof B Davies *pers. comm.*). This is due partly to the fact that limited monitoring of the effects of such developments on the downstream estuaries has occurred in South Africa and that where monitoring is occurring, the developments took place too recently for the full consequences to be evident (CSIR 1990a, 1990b, 1992a, 1994b). It is also ascribable to the inadequacy of standard predictive models in this regard. The critical element in the prediction of the consequences of freshwater deprivation or addition is the long term nature of the changes occasioned in the estuary. Generally standard hydrodynamic, transport-dispersion and water quality models are not designed to yield long term predictions, but are limited to simulation time frames of days to months (Knoester *et al.* 1991). Additionally existing ecosystem models (Baird & Ulanowicz 1993, Adams & Bate 1994) generally operate at spatial and temporal scales disparate from models capable of predicting physico-chemical changes in an estuary. The urgency of the societal demand for freshwater and the inadequacy of existing models in predicting the consequences of alterations in freshwater flows to downstream estuaries, caused the assessment of the freshwater requirements of estuaries to be identified as a management issue deserving of immediate research attention in 1992 (CERM 1992). Thus the development of techniques to reliably predict the effects of alterations in freshwater flow to an estuary in the medium to long term (months to years) is a critical management requirement.

While other local, regional and national development issues such as effluent disposal, agricultural development in the catchment and increased recreational use also will affect estuaries, this study aims to enhance prediction of the consequences of alterations in freshwater flow on downstream estuaries and hence to improve the design of management strategies to alleviate deleterious effects. The development of improved capabilities in these areas will assist generally in planning of regional and national water resource developments.

1.3 **Why Model?**

Clearly, prediction of the consequences of alterations in freshwater flow to estuaries over the medium to long term is a complex task. One needs to consider the primary effects on the hydrodynamics and geomorphology and the secondary effects on the chemistry and ecology of the estuary as well as interactions between the abiotic and biotic components. The only way in which "fluctuating variables can be satisfactorily integrated to facilitate the solution of practical management problems is by the use of mathematical modelling techniques as tools of integration" (Knoester *et al.* 1991). A modelling approach, therefore, is essential.

1.4 Why Model the Physical Dynamics?

Because the physical dynamics of an estuary define the environment to which the ecosystem adapts, improved management of estuaries requires improved prediction of the physical environmental response to alterations in freshwater flows. Reliable prediction of the abiotic environment is the first building block in a full systems modelling approach involving linked physical and ecosystem models as advocated by Knoester *et al.* (1991).

A further reason for focussing attention on the development of a modelling capability for predicting the physical dynamics of South African estuaries over the medium to long term is the fact that the management actions commonly employed to mitigate the effects of developments are controlled water releases from upstream impoundments and mechanical breaching or skimming (maintenance of the sand bar at a constant level) of the estuary mouths (CSIR 1990a & b, 1992b), that is, actions that relate directly to the physical environment of estuaries. It is also noteworthy that these management actions focus on the *fresh water* inflow to the estuary and the *connection to the open sea*, distinctive features of South African estuaries.

1.4.1 Objectives of the study

Consequently, this study involving modelling of the physical dynamics of estuaries was initiated with the following specific aims:

(i) to develop a model which adequately represents the physical dynamics of South African estuaries over time scales from months to years and so enables prediction of the medium to long term consequences of alterations in the fresh water inflow on the physical functioning and character of an estuary, and

(ii) to enable evaluation of the efficacy of management policies involving water releases and mouth breachings in the maintenance of estuarine character and functioning,

and the more general aim of enhancing the assessment of the freshwater requirements of estuaries and so assisting in planning of regional and national water resource developments.

1.5 Implications of the Study

At the inception of the study, it was evident that a fresh view was required of the small, bar-built, highly dynamic estuaries characteristic of the South African coast and that the aspects of the physical dynamics relevant for inclusion in the model would need to be selected thoughtfully. The requirements that the modelling study contribute substantially to the management of estuaries, also placed bounds on the level of the theoretical investigation which could be undertaken. Consequently, a direct outcome of the study is the provision of a practically applicable, scientifically-sound model relevant to estuarine management - a substantial advance in the state of estuarine prediction in South Africa.

However, the implications of the study extend further than the formulation and implementation of the estuarine systems model on South African estuaries. The understanding of the physical dynamics of small, bar-built estuaries on high wave energy coasts has increased immeasurably through this study, primarily because this is the first simulation tool focussing purely on their medium to long term behaviour. This is demonstrated by the fact that:

- behaviour known to occur in shallow estuaries, such as modulation of the mean water level by low frequency forcing and the generation of overtides (Aubrey & Speer 1985, Brundrit *et al.* 1988, MacKay & Schumann 1991), was simulated by the estuarine systems model although it was not included specifically in the model formulation, and
- the study was able to supply quantitative information to the debate on the extent to which anthropogenic influence is expediting the natural successionary process of South African estuaries towards coastal lagoons.

2. THE STUDY METHOD AND MODELLING TECHNIQUE

Fischer (1979) maintains that "the most important step in the art of modeling is selection of the right model". Accordingly, the first task of the study involved selecting a modelling technique. This was undertaken by determining selection criteria, critically examining existing modelling techniques for applicability and finally selecting the most appropriate technique.

2.1 Considerations in the Selection of a Modelling Approach

The temporal and spatial scales of the issues to be addressed by the model are central to the selection of an appropriate modelling technique (Smith 1986, Hauhs 1990). The issues to be addressed in this case include:

(i) ***Physical Dynamics*** - The time scales over which input parameters vary range from hours (eg. freshette, tidal action) to seasons (eg. alterations in sea state or fresh water inflow with season), while the time scale of the estuarine response can range from hours to years. The spatial extent of these influences is limited to the estuary and adjacent floodplain. Estimations of circulation and mixing within the estuary are required (eg. degree of stratification). The effects of changes in climate or sea level on the physical dynamics of estuaries, which extend over time scales of months to decades, are beyond the scope of this study and are not addressed by the model.

(ii) ***Management Actions*** - These commonly involve breaching of the mouth and the release of water from upstream impoundments. Mouth breaching usually occurs over a period of a few hours and activities are confined to the mouth area. Water releases, on the other hand, can occur over a few days at high flow rates or continually at low flow rates. The temporal scale, therefore, ranges from days to years, while on the spatial scale some knowledge of the longitudinal and vertical extent of influence of the fresh water inflow to the estuary is required.

(iii) ***Consequences of Management Actions*** - These range over temporal scales from hours (eg. draining of an estuary subsequent to the breaching of the mouth) to decades (eg. cumulative effect of reduced fresh water input). The spatial scale is limited to the size of the estuary, with particular emphasis on the mouth region and the estimation of general circulation and mixing conditions within the estuary. The potential to quantitatively distinguish between the consequences of different management actions, for instance on the condition of the mouth, is required.

It is evident from the considerations enumerated above that the modelling technique selected must be time-dependent. Further, the capability to undertake medium to long term simulations (months to years) is a stringent requirement. High spatial resolution appears not to be as strict a requirement, but some indication of processes in the mouth region and the influence of fresh/saline water on circulation and mixing is desirable. Quantitative comparison of the efficacy of different management strategies is a requirement.

Further issues to be considered in the selection of a modelling technique are the data requirements, the availability of these data, the accuracy and computational effort of model applications and the management orientation of the modelling technique. These will be discussed further in relation to the specific techniques described subsequently.

2.2 Existing Modelling Techniques

The criterion of time-dependence, which has arisen from the specification of the issues that the model must address, eliminates from consideration static or time-independent techniques. Similarly, the dearth of data of the medium to long term consequences of alterations in freshwater flow to estuaries means that expected system states cannot be associated with variations in riverine flow, wave heights and tidal flux on a generic basis with any certainty. This limits the applicability of a purely statistical treatment of the problem to those estuaries for which sufficient data is available. A deterministic modelling approach is thus selected in preference to a stochastic approach.

Techniques that represent the system response as a set of discrete states and qualitatively predict the variation with time (Starfield & Bleloch 1986) are a possibility. However, key physical processes such as tidal flux, the exchange of salinity between the sea and the estuary and erosion/deposition of sediment in the mouth area, could not be modelled continuously or accurately using this approach. The temporal and spatial resolution of the technique is insufficient for accurate prediction of the estuarine response to different management strategies eg. the state of the estuary mouth and stratification in the estuary, thus limiting the degree to which quantitative comparison of the efficacy of the different management policies could occur. Therefore, predictive, rule-based modelling such as that undertaken for the St Lucia system (Starfield *et al.* 1989, Carter *et al.* 1996) or for estuarine macrophytes (Adams & Bate 1994) is not suitable, despite the technique's excellent orientation towards management. This leaves mechanistic modelling techniques such as standard hydrodynamic, transport-dispersion and morphological modelling and systems modelling for consideration (Figure 2.1).

2.2.1 Standard hydrodynamic, transport-dispersion and morphological modelling techniques

One-dimensional hydrodynamic models have been implemented on estuarine systems since Preissmann (1961) and Leendertse (1967) put the numerical treatment of the shallow water or Saint-Venant equations on a sound footing. This modelling technique has been applied to South African estuaries from the late 1970's onwards (Hutchison 1976, Hutchison & Midgley 1978, Huizinga 1985a, 1985b, CSIR 1987, 1990a, 1990c, 1993b, 1994c, Slinger 1996). The partial differential equations comprising these models are based upon the principles of conservation of mass and momentum (Preismann 1961, Leendertse 1967, Abbott 1979, Huizinga 1985a, DHI 1992). Solution techniques can vary; amongst the most common are finite difference, finite element and boundary element methods (Abbott 1989). However, the following aspects are common to all such models:

- the estuary is divided into segments along its length with facilities to accommodate channel branching in some cases,
- data on the water level variation or flow at each boundary (usually the head and mouth of an estuary) are required to drive the models,
- the models yield water levels and depth-averaged current velocities for each segment and volume fluxes between segments,
- data on the water level variation and current velocities within the estuary are required for model calibrations, and
- the simulation time step required is of the order of a minute to ensure numerical stability and accuracy of the output.

Thus detailed results of variations in currents and water levels under different fresh water inflows and tidal conditions may be obtained. However, the length of time over which such scenarios usually are simulated is less than a year (Fischer 1979), generally ranging from weeks to months. While simulations can be conducted over longer time periods, the computational effort and manipulation of data files that is required, is considerable (Mr P Huizinga *pers. comm.*, CSIR 1993b). If such simulations are undertaken, constant fluvial and tidal inputs are usually assumed (Fischer 1979). Standard one-dimensional hydrodynamic models, therefore, are oriented to solving management problems requiring accurate, short term simulations.

Two and three dimensional hydrodynamic modelling systems have been formulated and applied to estuaries, bays and coastal seas (Dronkers 1964, Blumberg & Oey 1985, Langerak & Leendertse 1986, Falconer et al. 1989, Delft Hydraulics 1994a, 1994b). These address the inadequacies of one-dimensional hydrodynamic modelling by enabling the simulation of multi-layer dynamics and the consideration of lateral processes and effects. The simulation time step required by two and three dimensional models to ensure numerical stability and accuracy is usually of the order of a minute, but can be considerably less. Intensive computational effort is required to run these models, limiting the capability of multi-dimensional modelling techniques in medium to long term prediction. The data requirements of such complex models are extensive (Elliott 1977, Rasmussen 1989, Delft Hydraulics 1994a) and these data do not exist for most South African estuaries (Whitfield 1995). Further, in narrow, shallow estuaries, such as commonly occur in South Africa (Section 1.2.1), the effects of variations in currents with depth and lateral position are likely to be less significant than in deep, wide systems (Smith 1986). Therefore, where vertical stratification is limited, one dimensional hydrodynamic modelling (including branching) may be considered adequate for the treatment of practical problems in South African estuaries (Fischer 1979, Mr P Huizinga *pers. comm.*).

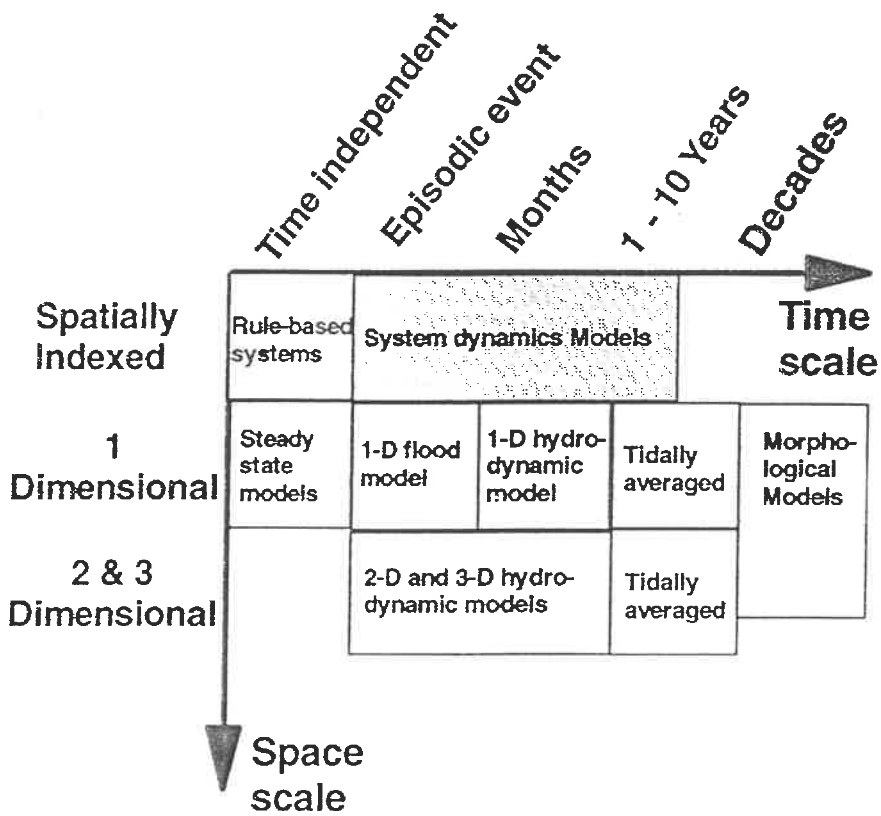

Fig 2.1 The temporal and spatial resolutions commonly associated with the different modelling techniques

Transport-dispersion models may be coupled to one, two and three dimensional hydrodynamic models (Huizinga 1985a, Nihoul & Jamart 1987, Falconer *et al.* 1989, Knoester et al. 1991, DHI 1992, Delft Hydraulics 1994a). These simulate the transport and dispersion of conservative pollutants and constituents such as salinity in response to the current regime. Transport-dispersion modules linked to one dimensional hydrodynamic models have been applied to certain South African estuaries to investigate longitudinal salinity distributions (Huizinga 1985b, Huizinga & Haw 1986, CSIR 1993b, 1994c). Model results are in the form of depth-averaged salinities. As with one-dimensional hydrodynamic modelling, the limitations of this approach lie in:

- the time period of the simulation, which generally is considerably less than one year unless steady inflow conditions are applied (Mr P Huizinga *pers. comm.*),
- the computational effort required for simulations, and
- the lack of vertical and lateral resolution.

However, no applications of complex two and three dimensional transport-dispersion models have been undertaken on South African estuaries as yet. It is doubtful whether the water level, current velocity and salinity data required for model calibration and verification are available at the necessary level of detail and the cost of their acquisition is prohibitive.

Water quality models to simulate the non-conservative behaviour of dissolved nutrients, oxygen, bacteria and pollutants in estuaries have been linked to coupled transport-dispersion and one, two or three dimensional hydrodynamic models (Falconer *et al.* 1989, DHI 1992, Delft Hydraulics 1994a). Such models have been applied to European, North American, South-east Asian and South American estuaries (Malmgren-Hansen et al. 1985, Bramley & Edwards 1989, Hooper *et al.* 1989, DeGroodt & de Jonge 1990, Delft Hydraulics 1994b). However, a lack of appropriate data on dissolved nutrient distributions, dissolved oxygen dynamics, bacteria and pollutant dispersion in South African estuaries has limited the use of these models to the implementation of a one dimensional water quality model on one estuary, namely the Great Berg Estuary (Slinger & Taljaard 1996). Thus, although water quality problems are known to exist in the Mgeni, Hartenbos and Swartkops Estuaries (CSIR 1993c, Bruwer & Hanekom 1989, MacKay 1994), no one, two or three dimensional water quality models have been calibrated on these South African estuaries to date.

Formulae, determining whether erosion, deposition or transport of sediment will occur also have been coupled to hydrodynamic models (DHI 1992). However, the accuracy and reliability of the predictions of sediment transport and the associated bed and channel form changes is not well established (De Vriend 1993). This is true particularly for dynamically active areas such as the mouth of an estuary (Mr P Huizinga *pers. comm.*). As theoretical understanding develops in this field, the accuracy and reliability of the formulae will improve. However, it is common practice at present to apply more than one sediment transport formula to a particular problem, to comparatively assess the predictions and even to average the most realistic results. This was the approach adopted in an investigation of sedimentation in the Kromme Estuary (CSIR 1991a) and in an assessment of the effect of the width of the bar at the mouth of the Wilderness Estuary on flood levels (CSIR 1994a). Such applications of sediment transport formulae provide an indication of areas of sediment accretion or erosion, but cannot simulate medium to long term morphological changes because consideration of the feedback between changing channel and bed forms is omitted (DHI 1992). Modelling of medium to long term morphological change is an active area of research, requiring the development of time stepping mechanisms allied to the time scale of morphological changes (De Vriend 1993). At this stage, tidal-averaging has proved to be the most useful technique for extending the time horizon of hydrodynamic and transport-dispersion model applications.

The tidal-averaging technique involves averaging of the output of hydrodynamic and transport-dispersion models (running at time steps of the order of a minute or less) over a tidal cycle and the use of these results or comparable field measurements as input to tidally-averaged hydrodynamic and transport-dispersion models, enables the simulation time step to be lengthened to the order of a tidal cycle. Medium to long term simulations (a year and longer) may then be undertaken (Fischer 1979). One, two and three dimensional hydrodynamic models may be driven in tidally-averaged mode (Fischer 1979, Webb & Singleton 1989, Knoester *et al.* 1991, Delft Hydraulics 1994a). Realistic results are obtained in situations where dramatic changes do not occur at irregular intervals or within a tidal cycle. Where river discharge is highly variable and infrequent periods of high discharge account for the majority of the flow, or sedimentation may be on such a scale as to close an estuary mouth within one tidal cycle (so altering the physical forcing experienced by the system), tidal averaging techniques are either not applicable (NERC 1983 p10), or require extensive data regarding the variability itself (Fischer 1979). Because river inflow to the majority of South African estuaries is highly variable and 48% of estuaries experience mouth closure (often within a tidal cycle) at some time (Begg 1978, Mr C Gaigher *pers. comm.*, Whitfield 1995), tidal averaging is not a viable modelling option for medium to long term simulation at present. If extensive information on concurrent fluvial inflows, mouth conditions and water levels and flows within an estuary were to become available in future, this technique could yield useful results.

Numerous numerical methods are employed in the modelling of sediment transport, salinity structure and the hydrodynamics of estuaries. Consideration of the details of different methods is beyond the scope of this brief review, which has concentrated on standard modelling techniques and their applicability to the problem at hand. For instance, explicit mention of the rival merits of Eulerian and Lagrangian formulations has been omitted as has mention of the use of random walk theory in the simulation of diffusion and dispersion. Additionally, semi-empirical morphological modelling techniques such as that employed by van Dongeren & de Vriend (1994), have not been discussed as these are more closely related to systems modelling techniques than standard morphological modelling. However, critical issues such as temporal and spatial resolution (Figure 2.1), the capability for medium to long term prediction, data requirements and the accuracy and computational effort involved in standard hydrodynamic, transport-dispersion and morphological modelling techniques have been addressed.

2.2.2 **Systems Modelling**

The system dynamics methodology was developed by Forrester (1961, 1969) as a result of his conviction that "the human mind is excellent as a recorder of decisions, reasons (policy), motivations and structural relationships, but is not good as a dynamic simulator of the behavioural implications of a complex policy structure" (Forrester 1980). Thus the technique was developed with the solution of management problems in mind and since the 1960's many system dynamics models have been applied to problems in the fields of ecology, social geography and management (Hearne 1987). System dynamics models can usually be described by a system of non-linear, first-order differential equations of the form:

$$\frac{dx_j}{dt} = F_j(\mathbf{x}, \mathbf{p}, t) \quad j = 1, 2, ..., n$$

where $\mathbf{x} = (x_1, x_2, ..., x_n)^T$ and $\mathbf{p} = (p_1, p_2, ..., p_m)^T$ are the state variables and the parameters of the system, respectively. Examples of system models include the Lotka-Volterra equations (at a theoretical level) and simulation models such as those of Forrester (1961, 1969, 1971), Odum (1971), Kremer & Nixon (1978) and Starfield & Bleloch (1986).

This "method of analysing problems in which time is an important factor and which involve a study of how a system can be defended against or made to benefit from the shocks which fall upon it from the outside world" (Coyle 1977) has the following attributes:

(i) A strong management orientation, although by focussing on mechanisms internal to the system excessive bias towards particular management objectives is avoided,

(ii) Moderate data requirements; often information on processes affecting measurable quantities is required rather than extensive information on the quantities themselves,

(iii) Simulation time steps are usually relatively large (of the order of hours to days), but depend on the time scales of the internal dynamics of the system being modelled,

(iv) The facility of incorporation of equations that are non-linear in variables and/or parameters (Sommer 1984),

(v) The flexibility to incorporate discontinuities (Wolstenholme 1982), and

(vi) The ability to represent natural adaptation processes by information feedback (Wolstenholme 1982).

Unlike standard hydrodynamic, transport-dispersion and morphological modelling, the system dynamics approach is of limited use in short term, high precision forecasting (Randers 1980), but comes into its own in the prediction of medium to long term dynamics and trends (Figure 2.1). A failing of the method lies in the high degree of detail which often is incorporated. A critical and selective approach to the modelling procedure is a prerequisite for successful application of this modelling technique (Prof A Starfield *pers. comm.*). Additionally, the spatial dimension is ignored in traditional system dynamics methodology. However, system models with some spatial differentiation have been formulated and successfully applied (Slinger 1989, Hearne & Quinn 1994, Busse *et al.* 1995). Thus, although system dynamics has not previously been applied to the modelling of hydrodynamic processes, this technique is a possible option for simulating the physical dynamics of estuaries over the medium to long term.

2.3 **The Modelling Technique Selected**

The modelling technique ideal for the study application may be described as follows:

- mechanistic, time-dependent, capable of medium to long term prediction, possessing moderate spatial definition, moderate data requirements, using readily available data, producing quantitative results, needing a fair (not high) degree of computational effort and oriented towards management.

From the preceding discussion and the summary of attributes of existing modelling techniques presented in Table 2.3, it is evident that no technique fulfils all requirements for application to the present study.

Despite its excellent management orientation, rule-based predictive modelling lacks the required temporal and spatial resolution. Standard hydrodynamic, transport-dispersion and sediment modelling, in turn, is primarily concerned with temporally and spatially accurate, short term problem solving. Although techniques such as tidal-averaging have been employed to extend the range of applicability of such models, particularly in the area of morphological modelling, the extensive data requirements, high computational intensity and limited time horizon constrain the applicability of such methods. The system dynamics methodology is transitional between these modelling techniques, lacking spatial definition but oriented towards medium to long term management. While both rule-based predictive techniques and the standard hydrodynamic approach could be used to address different aspects of the problem at hand, the system dynamics method was selected as the most appropriate technique.

Thus, in the absence of a proven technique for simulating the physical dynamics of South African estuaries over the medium to long term for management purposes, the adoption of an approach more commonly applied in the fields of ecology and geography was deemed necessary.

Table 2.3 Attributes of different modelling techniques

MODEL ATTRIBUTES	RULE-BASED PREDICTIVE MODELLING	STANDARD HYDRODYNAMIC, TRANSPORT-DISPERSION & MORPHOLOGICAL MODELLING				SYSTEM DYNAMICS MODELLING
		INTRA-TIDAL		TIDALLY-AVERAGED		
		1-D	2-D & 3-D	1-D	2-D & 3-D	
MECHANISTIC	No	Yes		Yes		Yes
TIME-DEPENDENT	Yes, but not continuously	Yes, dynamic		Yes, dynamic but cannot resolve intra-tidal dynamics		Yes, dynamic
PREDICTIVE CAPABILITY	medium to long term	short term		medium to long term		medium to long term
SPATIAL DIFFERENTIATION	Perhaps, but regions rather than accurate positions	Yes, cross-sectionally averaged	Yes, 2-D models are usually vertically averaged	Yes, but tidally and cross-sectionally averaged	Yes, but tidally averaged	Unlikely, but regions may be specified
DATA REQUIREMENTS	Moderate, usually uses aggregate data	Reasonably high	High	Reasonably high	High	Moderate data requirements. Data on linking processes often required.
		Accurate survey data, water levels, salinities and sediment concentrations are required				
DATA AVAILABILITY	Readily available	Available for specific estuaries, costly to acquire	Not available, cost of acquisition prohibitive	Not generally available, costly to acquire	Not available, Cost of acquisition prohibitive	Generally available. Data on linking processes sometimes not available
ACCURACY	Not accurate, answers are generally qualitative	Spatially and temporally accurate	Spatially and temporally accurate, but verification difficult	Accurate, but verification difficult	Accurate, but verification extremely difficult	Quantitatively accurate in regard to medium to long term trends, not accurate in the short term.
COMPUTATIONAL EFFORT	Low	Reasonable	High	Reasonable	High	Reasonable to low
MANAGEMENT ORIENTATION	Yes, excellent	Problem oriented rather than management oriented.				Yes, excellent

2.4 The Study Method

The selection of the system dynamics methodology as the most appropriate modelling technique is only the first step in the process of developing a model of the physical dynamics of an estuary which has the capability to address the medium to long term consequences of alterations in freshwater flow. The next logical step involves developing the model. Thereafter, application and testing of the model is necessary to establish confidence in the results and an understanding of the limitations of the model. Then a framework for application of the model can be developed and its utility in estuarine management addressed. The study method therefore comprised the following stages:

(i) Identification of the objectives of the study
(ii) Selection of the modelling technique
(iii) Formulation of the estuarine systems model
(iv) Application and testing of the model
(v) Use of the model for present day estuarine management, and
(vi) Identification of the implications of the model for estuarine management in the future.

2.4.1 Structure of the thesis

The formulation of the estuarine systems model is undertaken in Chapter 3. In Chapter 4, the model is implemented on two case studies and the sensitivity of results to uncertainty in parameter values is tested. The generic applicability and validity of the model formulation is further examined by exploring the relationship between concepts in the estuarine physics literature and the behaviour simulated by the model. In Chapter 5, the management use of the model is addressed first by demonstrating its application in the evaluation of different management strategies for the selected case studies (that is, its site specific application), next by indicating its use as a predictive tool in a linked system of models (that is, its integrative application) and finally by discussing its potential use as a screening tool in water resource development planning (that is, its strategic application). The findings, limitations and insights arising from the study are discussed in the concluding chapter (Chapter 6) and further research needs are briefly identified.

3. MODEL DEVELOPMENT

Two major aspects must be addressed in the subsequent development of the estuarine systems model, namely:

- simulation of the physical dynamics of estuaries, and
- evaluation of the efficacy of different estuarine management strategies.

The model, therefore, comprises two components: a physical dynamics component and a management evaluation component. The formulation of the former component will be developed first as "the determination of the mechanical variables" is regarded as an essential initial step in any effort to model a marine system (Nihoul 1975).

3.1 Physical Dynamics Component

3.1.1 Selection of the state variables

The first task in the development of the physical dynamics component of the model is the selection of state variables, that is, the measurable quantities of the system which will provide an adequate description of the estuary and its behavioural response to the type of external forcing to be considered.

In this case, the state variables were chosen on the basis of their success in satisfying the following broad questions:

- If this state variable were omitted, would our understanding of the physical character of the estuary be significantly impaired?
- Does prediction of the chemical and ecological character of the estuary depend on knowledge of this quantity? and
- Does this variable undergo a significant alteration in state in response to management actions such as water releases and mouth breachings?

The water volume and mouth condition are fundamental to a description of the geomorphology and hydrology of an estuary. Quantities such as the salt content, stratification-circulation regime (eg. partially mixed, highly stratified) and the frequency and extent of flushing (eg. stagnant, flushed with freshwater) are fundamental to a description of the hydrodynamic environment of an estuary and underpin prediction of the associated water quality and ecosystem responses (eg. high potential for de-oxygenation of the bottom water, phytoplankton blooms likely to occur, physical environment suitable for estuarine macrophyte growth, ichthyofaunal and invertebrate recruitment possible). All of these measurable quantities are modified significantly by an influx of freshwater and by opening of the estuary mouth. Thus the state variables deemed essential in modelling the physical dynamics of an estuary for management purposes in the medium to long term, are:

- the **water volume**, from which a representative water level is derived,
- the **salt content** of the water within the estuary,
- an index of **stratification,**
- an index of **circulation,**
- the **fresh water flushing** of the water body,
- the **tidal flushing** of the water body, and
- the **sill height** at the mouth, from which the extent of mouth closure/openess is derived.

Thus the selection of the state variables represents a compromise between the faithful portrayal of the complexity inherent in the hydrodynamic and geomorphological character of an estuary and a pragmatic consideration of the relevance of these aspects to an evaluation of management policy. For instance, a choice to model the axial variation in water levels within an estuary in detail would enhance the capability for prediction of the axial extent of saline intrusion and the response of the estuarine macrophyte community, which is sensitive to inundation and exposure times. However, it would impose constraints on the maximum allowable simulation time step and so limit the utility of the model in medium to long term management applications. An alternative choice of modelling the variation in water level over the estuarine basin (the approach chosen in this study) restricts the applicability of the model, because it may not then adequately represent the physical dynamics of long estuaries with strong axial variations in water levels and salinities and prediction of the response of the estuarine flora and fauna is less accurate. However, its capability to model the medium to long term response of the estuary to different management strategies is enhanced.

3.1.2 Features of the model formulation

The relationships between the state variables and their rates of change are described by a series of ordinary differential equations. Where uncertainty exists over the most appropriate formulation eg. in the selection of the sediment transport formula, alternatives are described and the reasons for the eventual selection are given. Where model assumptions limit the range of applicability of the model, these are highlighted.

Features to note in the subsequent model formulation include:

- the phenomenological basis of the model formulation,

- the formulation of the tidal flux rate, which is critical in the determination of salt and sediment fluxes,
- the semi-empirical approach to modelling the intra-tidal effects in the salt export and import rates,
- the formulation of the state variables for stratification, circulation and fresh water and tidal flushing, which incorporate estuarine physics concepts dynamically and provide some indication of the spatial structure within the water body,
- the use of the sill height at the mouth in the indication of mouth state.

These features/concepts are fundamental to the development of the capability to simulate the physical dynamics of estuaries over the medium to long term.

3.2 Water Volume Sector

The volume of water in an estuary is influenced by:

- the inflow of fresh water at the head of an estuary, due to run-off from the catchment and releases from impoundments,
- precipitation and evaporation, which act over the entire surface extent of the estuary,
- the groundwater inflow to the estuary or loss from the estuary, particularly due to seepage through the sand bar at the mouth, and
- tidal exchange through the mouth of the estuary.

By the principle of the conservation of mass, these inflows and outflows determine the rate of change of estuarine water volume according to the differential equation:

$$\frac{dx_1}{dt} = \Sigma x_{1j} \qquad j = 1,4$$

where x_1 = water volume (m^3) and x_{1j} j=1 to 4 are the rates listed in Table 3.2. Certain of the rates incorporate both exogenous variables and endogenously derived components in their formulation, while others are solely exogenously specified.

Table 3.2 Rates influencing the water volume

RATE	SYMBOL	EFFECT	UNITS	EXOGENOUS
Fresh Water Inflow	x_{11}	+	$m^3.yr^{-1}$	yes
Precipitation/Evaporation	x_{12}	+ / -	$m^3.yr^{-1}$	component
Groundwater Flow	x_{13}	+ / -	$m^3.yr^{-1}$	yes
Tidal Flow	x_{14}	+ / -	$m^3.yr^{-1}$	component

3.2.1 Fresh water inflow

Fresh water entering the head of an estuary may derive from two sources, namely as run-off from the catchment or releases from an upstream impoundment. The run-off response to precipitation differs from one rainfall event to the next and from one catchment to another according to such factors as rainfall intensity, duration and distribution, antecedent rainfall and soil moisture, soil type, geology, slope, catchment area and land-use amongst others (Wisler & Brater 1949, DWA 1986). Additionally, the proportion of a water release or run-off event which attains the estuary is dependent upon the antecedent flow conditions and storage capacity of the river channel itself (Linsley *et al.* 1949). Abstraction from, or addition of water to, the river and/or estuary for agricultural, industrial or domestic use may complicate the picture further. Because the focus of this study is on the dynamic response of estuaries to physical forcing, it was not deemed necessary to incorporate these factors explicitly in the model. Instead, the practical approach of modelling all contributions to fresh water flow as one rate entering the head of the estuary was adopted. This is in accordance with the strategy adopted by South African scientists in the prescription of the freshwater requirements of the Great Berg Estuary, where the required flow pattern at the head of the estuary was specified, owing to doubt about the level of abstraction of water from the river channel downstream of a dam by riparian land owners (DWAF 1993a).

Therefore, the total fresh water inflow rate is specified as an exogenous function:

$$x_{11} = IF(t)$$

where x_{11} = fresh water inflow rate ($m^3.yr^{-1}$)
$IF(t)$ = inflow function ($m^3.yr^{-1}$)

3.2.2 Precipitation/evaporation

The input to the estuarine water volume from direct precipitation is given by the product of the surface area, which is determined as a function of water volume, and the rainfall function. The rainfall function is exogenously specified and incorporates effects such as the variation in rainfall with season, drought or storm. Similarly, the rate at which water is lost from the estuary by evaporation is proportional to the surface area and is determined as the product of the surface area and the externally specified evaporation function. This function varies according to season. Owing to the common dependence on surface area of both the evaporative loss and the contribution from precipitation, the nett effect of precipitation and evaporation is calculated as:

$$x_{12} = [RF(t) - EF(t)].SA(x_1)$$

where x_{12} = precipitation/evaporation ($m^3.yr^{-1}$)
$RF(t)$ = rainfall function ($m.yr^{-1}$)
$EF(t)$ = evaporation function ($m.yr^{-1}$)
SA = surface area (m^2)
x_1 = water volume (m^3)

3.2.3 **Groundwater flow**

The characterisation of sub-surface flow requires detailed knowledge of the geohydrology and pedology of the area concerned (Kovacs 1981), in this case the area adjacent to an estuary. For instance, information on the hydraulic conductivity, the depth and breadth of sand and the relative difference in height between the sea and the estuary is required for estimation of the seepage of water through the sand bar at the mouth of an estuary (Kovacs 1981: p28-37, Mr P Huizinga *pers. comm.*). For the majority of South African estuarine systems the information required for accurate modelling of the processes involved in sub-surface flow, is lacking. However, the contribution of groundwater to the water volume of an estuary may be significant during droughts, while seepage can result in considerable volume loss when the mouth is closed (CSIR 1990a). Therefore, although explicit modelling of these processes is beyond the scope of this study, the nett influence of groundwater flow is included in the form of an exogenously specified groundwater function:

$$x_{13} = GF(t)$$

where x_{13} = groundwater flow ($m^3.yr^{-1}$)
$GF(t)$ = groundwater function ($m^3.yr^{-1}$)

3.2.4 **Tidal flow**

The tidal flow may be either an input or a loss to the system. According to Ippen and Harleman (1966), the direct generation of tidal flows by moon and sun is negligible in estuaries, but rise and fall of the sea tide causes an exchange of water through the mouth of the estuary. On the flood tide a mass of sea water enters an estuary and is stored temporarily before draining seaward on the ebb tide. To determine the tidal flux, it is necessary first to know the tidal sea level and then to consider the factors influencing tidal exchange.

Modelling of the tidal water level

The South African coastline is microtidal with a relatively symmetrical, predominantly semi-diurnal tidal signal varying in amplitude from 1,8 m and 1,5 m at spring tide to 0,5 m and 0,6 m at neap tide at Richards Bay and Simon's Bay, respectively (Rosenthal & Grant 1989). Various methods of tidal prediction exist. These range from highly accurate 114-constituent harmonic analyses (Zetler 1982) to simple cosine/sine series. Routine prediction of the tidal level is undertaken by the S A Navy Hydrographic office using a 51-constituent harmonic analysis routine (Shipley 1980). However, Rosenthal and Grant (1989) developed a simpler method, which uses only ten pre-calculated constants. Although less accurate, this method yields results in good agreement with the more detailed method. For practical application the less complex methods, namely the simple cosine/sine series or the method of Rosenthal and Grant, are commonly preferred. The expected variations in tidal sea level are incorporated in the model through an exogenous tide function, the form of which is selected according to the desired accuracy of the tidal prediction.

However, longer period and random fluctuations in sea level occur, which are not predicted by the methods described above (Rosenthal & Grant 1989). Storms, mid-latitude cyclones, high winds and waves are known to influence coastal sea levels (Elliot & Wang 1978, Dyer 1979, Smith 1985) and weather events have been linked to the generation of coastally trapped waves which propagate eastwards along the South African coast, affecting coastal sea levels (Schumann & Brink 1990, Jury *et al.* 1990). Accordingly, observed sea levels differ from those predicted. This effect is modelled by the inclusion of a further exogenous function known as the meteorological function, which modifies the tide function. The tidal water level is then calculated as the product of the tide function and the meteorological function:

$$TWL = TF(t).MF(t)$$

where TWL = tidal water level (m)
TF(t) = tide function (m)
MF(t) = meteorological function (unitless)

Factors influencing tidal exchange

The difference between the tidal water level (a sea-imposed condition) and the water level within an estuary provides the hydraulic head forcing barotropic tidal exchange through the mouth. For a given head difference, however, the volume of water which enters and/or leaves an estuary is known to vary according to:

- the morphology of the estuary basin,
- the effect of bottom friction,
- the gravitational circulation in the estuary, and
- the configuration of the estuary mouth (eg. closed, open, shallow, narrow).

The fact that basin morphology influences the hydrodynamic behaviour of estuaries was acknowledged in the earliest estuarine investigations and has been confirmed by many studies since then (Dronkers 1964, Ippen & Harleman 1966, Dyer 1973, Jay 1991). However, most studies were concerned with the influence of estuarine geometry on the amplitude of the tidal wave as it progresses up the estuary. Thus attention has focussed on features such as the length of the channel relative to the wavelength of the tidal wave, the convergence or divergence of the channel and its average depth. In contrast, the influence of particular topographic elements or basin shapes on volume fluxes has received limited attention. However, some site specific observations and analytical studies do exist. Bowden & Gilligan (1971) found that the shape of the estuary affected tidal volume flux in the Mersey Estuary, while Aubrey and Speer (1985) and Speer and Aubrey (1985) established that the presence of broad tidal flats can result in ebb dominance of an estuary rather than the flood dominance generally associated with narrow systems. Recently, Jay (1991) affirmed that topographic convergence/divergence is a significant factor in the determination of the behaviour (to lowest order) of the tidal long-wave in shallow estuaries, but demonstrated that certain (smooth) variations in channel topography can lead to substantial deviations from classical predictions of tidal transport. Thus the response of an estuary to a given head difference is non-uniform, varying according to the length and shape of the particular estuary. The effects of estuarine basin morphology on tidal flux are included in the model by the introduction of a length factor over which the hydraulic head is assumed to act and by the inclusion of a storage capacity function of water level, which accounts for the influence of estuarine shape.

Jay (1991) reaffirms the classical finding that friction is important in all estuarine systems and dominant in shallow estuaries (Green 1837, Dronkers 1964, Ippen & Harleman 1966). As many South African estuaries are small, narrow and have shallow mouths with well developed flood tidal deltas (Reddering & Rust 1990), bottom friction effects are significant (Aubrey & Speer 1985, Speer & Aubrey 1985). Distortion in the tidal fluxes results and shorter duration, more intense flood tidal flows and longer duration, less intense ebb tidal flows are common features (MacKay & Schumann 1991, CSIR 1990b & c). The effect of bottom friction in promoting the asymmetry between the ebb and flood tidal velocities is modelled phenomenologically rather than causally by the inclusion of the velocity function.

Gravitational circulation in estuaries, more properly termed baroclinic circulation (Fischer *et al.* 1979), arises from horizontal density differences between river water and sea water along the length of the estuary and from buoyancy differences between these water types (Dyer 1973, Fischer *et al.* 1979). Thus gravitational circulation can occur in the absence of barotropic forcing, but more commonly it is the combined effect of both barotropic and baroclinic forcing mechanisms that is responsible for tidal exchange (Dyer 1973, Officer 1976). Baroclinic flows are typically of much lower frequency and propagate more slowly than barotropic flows (Apel 1987). On the time scale of the tidal cycle, therefore, the purely density-driven circulation generally does not noticeably affect the magnitude of the nett volume flowing through the mouth of an estuary in response to the barotropic forcing experienced along the South African coast. However, gravitational circulation does play a significant role in the transport of salt (Section 3.3) and sediments (Section 3.6) and the establishment of characteristic estuarine circulations (Section 3.4) eg. highly stratified, partially mixed, vertically mixed (Hansen & Rattray 1966, Fischer 1972, Dyer 1973, Fischer *et al.* 1979, Dyer 1986, Largier 1986, Jay & Smith 1988, Largier & Slinger 1991). Consequently, a circulation index, representing the fractional contribution of density-induced flow to the mean tidal flow, is developed in Section 3.4 and the effect of gravitational circulation on the tidal flow is included in the model through a function of this circulation index. This function, termed the gravitational circulation function, reflects the weak contribution of density-driven flows to the overall tidal flux in vertically mixed systems and the stronger contribution in highly stratified systems.

The effects of basin morphology, bottom friction and density-driven circulation on tidal flux are not of such severity as to prevent tidal exchange. In contrast, the mouth of an estuary may be closed or so constricted that tidal exchange is curtailed. The influence of mouth configuration on tidal exchange is modelled by first considering the exchange which would occur in the absence of this constraining influence (the target tidal flux) and then determining the magnitude of the reduction owing to mouth constriction. Taking into account the effects of basin morphology, bottom friction and gravitational circulation, the target tidal flux is:

$$\begin{aligned} TTFLUX &= (HH / L).SCF(WL).VF(TWL,WL).GCF(x_4) \\ HH &= TWL - WL \\ WL &= WLF(x_1) \end{aligned}$$

where TTFLUX= target tidal flux ($m^3.yr^{-1}$)
HH = hydraulic head (m)
L = length (m)
SCF = storage capacity function ($m^3.yr^{-1}$)
WL = water level (m)
VF = velocity function (unitless)
TWL = tidal water level (m)
GCF = gravitational circulation function (unitless)

x_4 = circulation index (unitless)
WLF = water level function (m)
x_1 = water volume (m^3)

In a given time period the target tidal flux may not be able to enter or exit the estuary owing to constriction or closure of the mouth. Officer (1976) demonstrated that for steady one layer flow (given energy) a shallowing or constriction in a channel exercised a limiting effect on the flow velocity/volume exchange with the maximum flow condition proving to be the critical flow condition. In expanding on the work of Stommel & Farmer (1953), Farmer and Armi (1986) and Armi and Farmer (1986) established that the shallowing (decrease in depth) and contraction (decrease in width) of an estuary mouth can exercise internal hydraulic control on two layer exchange flow through the mouth. In contrast to Stommel and Farmer (1953) who concluded that a limiting exchange condition (known as overmixing) associated with the presence of stratification occurs, the later work indicated that although volume discharge is constrained a state of overmixing does not exist. Largier and Slinger (1992) demonstrated that internal hydraulic control of two layer exchange flow occurs in the Palmiet Estuary, South Africa, and that features associated with such control (eg. tidal intrusion fronts) occur in other South African estuaries. The constraining effect of the mouth cross-section (shallowing, contraction) on tidal volume exchange is modelled to ensure the applicability of the model to the many shallow-mouthed estuaries in southern Africa.

The mouth is modelled as a rectangular channel connected to the estuary basin at the sill height (Fig 3.2a). The cross-sectional area over which flow through the mouth may occur, is calculated as the product of the width and the effective depth of flow. The effective depth of flow at the mouth, in turn, is calculated as the difference between the greater of the tidal or estuary water levels and the height of the sill (x_7). Assuming a characteristic width for the estuary mouth, the cross-sectional area is given by:

$$
\begin{aligned}
MCS &= w.h \\
h &= TWL - x_7 \ \text{ if } TWL > WL \text{ and } TWL > x_7 \\
&= WL - x_7 \ \text{ if } WL \geq TWL \text{ and } WL > x_7 \\
&= 0 \text{ otherwise}
\end{aligned}
$$

where MCS = mouth cross-sectional area (m^2)
w = width of the estuary mouth (m)
h = effective depth of flow (m)
TWL = tidal water level (m)
WL = estuary water level (m)
x_7 = sill height (m)

When the height of the sill exceeds the water levels of both the sea and the estuary, the effective depth of flow is zero, the mouth is closed (MCS = 0) and consequently the tidal flux is zero. To establish the extent to which the target tidal flux (TTFLUX) is constrained by shallowing or constriction at the mouth (rather than complete closure), it is necessary to determine the critical flow volume.

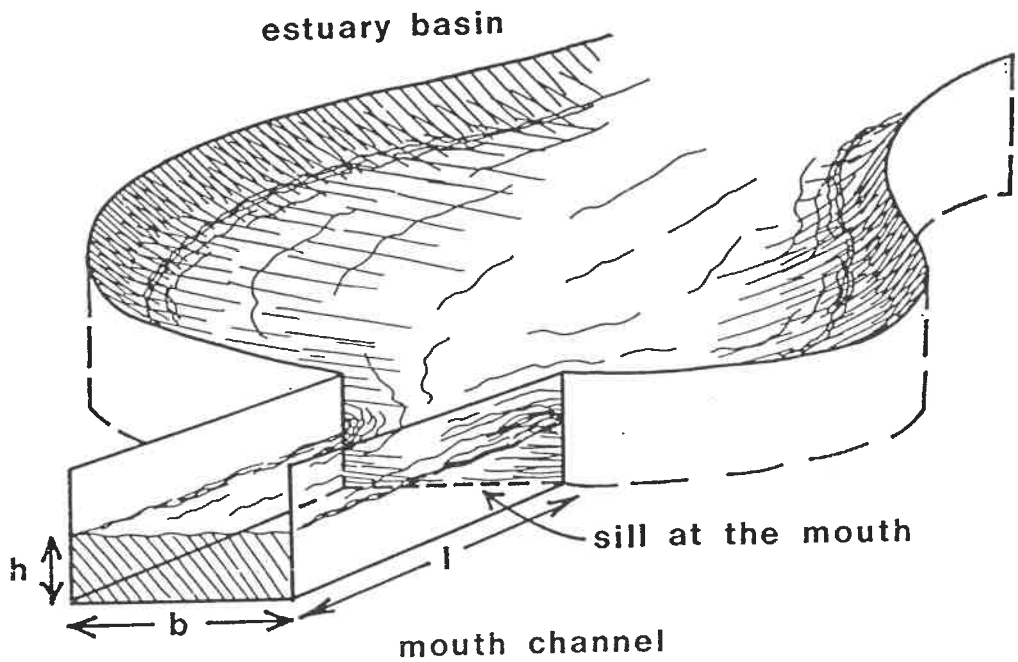

Fig 3.2a Diagram of the mouth modelled as a rectangular channel connected to the estuary at the sill height

For a fluid flowing through a channel constriction/shoal, the critical flow velocity is the velocity at which the Froude Number ($u/(gh)^{1/2}$, where u=velocity, g=gravitational acceleration and h=depth) is unity (Officer 1976). However, in the region of an estuary mouth, whether single-layer flow or multi-layer flow characteristically occurs (this depends on the stratification-circulation in the system as explained in Section 3.3), the exchange of two fluids of different densities takes place. The critical flow condition then occurs when the two layer composite Froude number, defined as $G^2 = F_1^2 + F_2^2$, where $F_i^2 = u_i/(g'h_i)^{1/2}$ and F_i=layer densimetric Froude number, u_i=layer fluid velocity, g'=reduced gravitational acceleration and h_i=layer depth, is unity (Largier & Slinger 1991). In examining Fig 3.2b, in which the flow strength is indicated at different stages of tidal exchange between two fluids of different densities through a constriction and over a sill, it is clear that the nett volume flow is maximum when the critical condition for uni-directional flow applies either at the seaward or landward side of the mouth. This occurs when the densimetric Froude number for the active layer is unity, that is $u_i=(g'h_i)^{1/2}$ where layer i is flowing actively. Note that g'<g, therefore, the critical layer velocity for uni-directional flow over depth h is less when density differences are present than when they are absent. Thus knowing the total effective depth of flow and the mouth cross-sectional area, an upper limit for the maximum critical flow volume may be established.

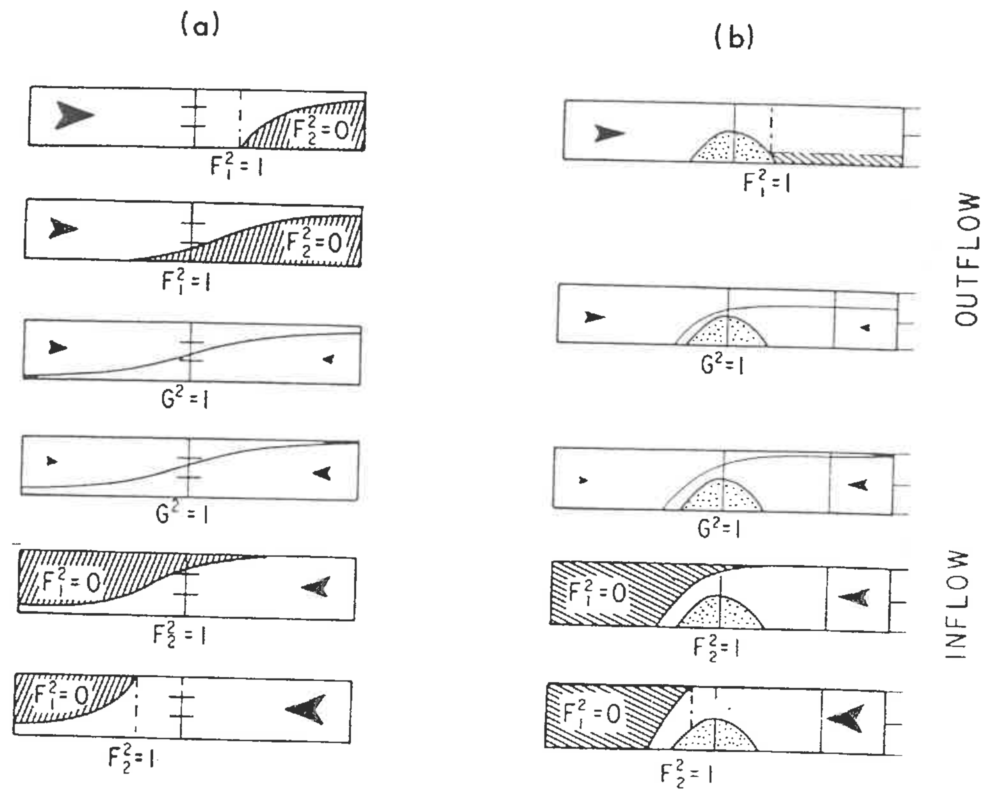

Fig 3.2b A two dimensional schematic of the exchange flow of two fluids of different densities through a contraction (a) and over a sill (b). The arrowheads indicate flow strength, while shading is indicative of a stationary layer. The sea is to the right. Internal hydraulic control of the flow occurs where the composite Froude number G^2 is unity. Under strong inflow or outflow the flow of one fluid is arrested and hydraulic control of the flow occurs where the densimetric Froude number of the active layer is unity. Following Armi & Farmer (1986) and Farmer & Armi (1986), negligible mixing and friction are assumed (adapted from Largier & Slinger (1991)).

Much of the time, the actual flow volume is likely to be less than the maximum condition. Therefore, if the target tidal flux is less than the critical flow volume, no reduction of the target tidal flux occurs i.e. the mouth has no constraining effect on the tidal flow. On the other hand, if the target tidal flux exceeds the critical flow volume, the target tidal flux is moderated by a control factor (which is not unity) to yield the tidal flow:

$$x_{14} = \mathrm{TTFLUX.CONTRF}$$

$$\mathrm{CONTRF} = \mathrm{UC.MCS.CONV} / |\mathrm{TTFLUX}| \quad \text{if } \mathrm{UC.MCS.CONV} < |\mathrm{TTFLUX}|$$

$$= 1 \quad \text{otherwise}$$

$$\mathrm{UC} = (g.h)^{1/2}$$

where x_{14} = tidal flow ($m^3.yr^{-1}$)
TTFLUX = target tidal flux ($m^3.yr^{-1}$)
CONTRF = control factor (unitless)
UC = critical flow velocity ($m.s^{-1}$)
MCS = mouth cross-sectional area (m^2)
CONV = conversion factor ($s.yr^{-1}$)
g = gravitational acceleration ($m.s^{-2}$)
h = effective depth of flow (m)

In summary, therefore, when the mouth is open, the relative water levels of the sea and the estuary determine whether nett tidal inflow or outflow occurs. The magnitude of the tidal flux is determined primarily by the head difference between the sea and a representative water level in the estuary. Additionally, the effects of factors such as the morphology of the estuary basin, bottom friction, gravitational circulation in the estuary and the configuration of the estuary mouth are incorporated in the formulation of the tidal flow (x_{14}).

The intra-tidal momentum effect, which is a term describing the delay in the transition from ebb to flood tide and *vice versa* (slack water) over the estuary as a whole, is not included explicitly in the tidal flux formulation (this would require simulation of axial variations in water level). This implies that accurate velocities cannot be calculated within the estuary. However, the omission of the momentum condition is instrumental in the capability of the present approach to undertake long term simulations effectively.

Finally, the validity of this semi-empirical, mechanistic formulation of tidal flux can only be established by comparison of model predictions and field data and qualitative assessment of the agreement between predictions and the known behavioural responses of estuaries (Sections 4.3 & 4.5).

3.3 Salt Sector

Following the development of equations for the rates affecting the estuarine water volume, the saline content of the water and its rate of change may be determined as salt may be considered a conservative property (Dyer 1973). Invoking the principle of conservation of mass, then, the following differential equation describes the estuarine salt fluxes:

$$\frac{dx_2}{dt} = \Sigma\, x_{2j} \qquad j = 1,2$$

where x_2 = salt (kg)
x_{21} = salt import ($kg.yr^{-1}$)
x_{22} = salt export ($kg.yr^{-1}$)

3.3.1 Salt import

Generally, saline water enters an estuary on the incoming tide. Where active turbulent mixing occurs in the surf zone offshore of the estuary mouth, the inflowing water is of sea salinity. However, if lower salinity water expelled from the estuary on the ebb tide forms a buoyant plume at the mouth, a portion of this water may be returned to the estuary on the flood tide (Dyer 1973, Fischer *et al.* 1979). Thus the quantity of salt entering an estuary depends on the volume of water flowing into the system from the sea and varies according to the salinity of this water (Fischer *et al.* 1979).

In vertically mixed estuaries, tidal flow is landward on the flood tide and seaward on the ebb tide, that is, single-layer flow occurs through the estuary mouth. In partially mixed estuaries, current velocities are highest in the surface layers during ebb tide and in the bottom layers during flood tides (Dyer 1973). These differences in the amplitudes of the surface and bottom tidal currents cause the average current over a complete tidal period (residual flow) to be landward at depth and seaward at the surface (Dyer 1973, Officer 1976, Fischer *et al.* 1979). However, this does not mean that multi-layer flow occurs in the mouth region, apart from for a short interval around the turn of the tide when inertial effects may mean that currents are not uni-directional throughout the water column (Officer 1976). Generally, tidal flows are landward throughout the water column on the flood tide, although the current velocity is greater at depth than in the surface layer/s. Similarly, tidal flows are seaward throughout the water column on the ebb tide, although the current velocity is greater in the surface layer/s than at depth. Thus partially stratified bar-built estuaries exhibit single-layer tidal flows in the mouth region for the most part. Ignoring intra-tidal inertial effects, the salt import rate to these estuaries, therefore, is given by the product of the flood tidal flux and the average salinity of the intruding water.

In a highly stratified estuary, however, multi-layer flow may prevail throughout the tidal cycle (Officer 1976, Fischer *et al.* 1979). Thus low salinity outflow may occur in the surface layer/s while there is nett inflow (flood tide) and saline inflow may occur at depth even when there is nett outflow (ebb tide). As highly stratified estuaries in South Africa commonly have a shallow sill or bar at the mouth (Reddering & Rust 1990, Largier & Slinger 1991), the latter effect is not as well developed as in the wedge-shaped systems of Europe and North America eg. the Duwamish, Vellar and Mississipi Estuaries and Rotterdam Waterway (Officer 1976, Jay & Smith 1988), which have wide, deep mouths. Another feature associated with the presence of complex bottom topography in the form of sills or bars at the mouth is the upstream blocking of ebb tidal outflows observed by Largier & Slinger (1990), Largier *et al.* (1992) and Slinger *et al.* (1993). Consequently, the effect of multi-layer flow during ebb tide may be modelled by reducing the efflux of salt with the ebb tidal outflow, rather than explicitly including salt import during ebb tidal stages (see Section 3.3.2). In similar vein, the effect of multi-layer flow during flood tide (i.e. the simultaneous surface outflow of low salinity estuarine water and influx of highly saline water at depth) is accommodated in the model formulation by reducing the average salinity of the inflowing water volume according to the degree of stratification (x_3) and density-induced circulation (x_4) in the estuary. Thus the quantity of salt entering an estuary is modelled as the product of the volume of water flowing into the estuary from the sea and the reduced salinity. The reduced salinity is a function of the stratification-circulation in the estuary, assuming a value equivalent to ocean salinity when the system is vertically mixed, a slightly lower value (equivalent to the average salinity of the intruding water) when the system is partially stratified and the ebb tidal outflow may not have mixed completely with the sea water, and a still lower value when the estuary is highly stratified and there is a continuous outflow of low salinity surface water. Therefore, the salt import rate is:

$$x_{21} = RS(x_3,x_4).x_{14} \quad \text{if } x_{14} > 0$$
$$= 0 \quad \text{otherwise}$$

where x_{21} = salt import ($kg.yr^{-1}$)
RS = reduced salinity ($kg.m^{-3}$)
x_3 = stratification index (unitless)
x_4 = circulation index (unitless)
x_{14} = tidal flow ($m^3.yr^{-1}$)

3.3.2 **Salt export**

The export of salt from an estuary generally occurs during ebb tides. However, outflow of low salinity surface water (and inflow of highly saline bottom water) may occur continuously in a highly stratified system (Officer 1976, Fischer *et al.* 1979). Additionally, flooding or periods of high river flow may expel all salt from an estuary (Largier & Taljaard 1991, CSIR 1993d). Thus the quantity of salt lost to a system depends upon the magnitude of the outflowing volume and the saline concentration of this water.

If an estuary were uniformly mixed throughout, the salt export rate would be given simply by the product of the outflowing volume flux and the average concentration, that is the ratio of the salt content to water volume of the estuary. However, most well mixed estuaries still exhibit a longitudinal gradient in salinity with fresher riverine water located towards the head of the estuary and sea water at the mouth, as do partially stratified systems (Officer 1976, Day 1981). This axial variation in saline concentration means that under average flow conditions the concentration of the effluent water is higher than the average salinity of the estuary, because the more saline water located near the mouth is expelled from the estuary first. This effect is exacerbated under low river flow conditions when tidal exchange may only affect the lower reaches of the estuary. However, under high river flows the concentration of the outgoing water approaches the average salinity of the estuary as more effective flushing occurs. Ultimately, during a freshwater flood the system is flushed entirely and water of zero salinity (the average concentration under these conditions) is expelled through the mouth of the estuary. Thus the saline concentration of the effluent water compared with the average estuarine concentration varies according to the flushing rate of the estuary, owing to the presence of axial salinity gradients. The effect of these longitudinal gradients in salinity on the salt export rate is modelled by the inclusion of a salinity export multiplier. This is a function of the tidal and fresh water flushing rates (x_5 and x_6) and not of the circulation index, which is indicative of the baroclinic component of the flow regime rather than the volume exchange with the sea. The salinity export multiplier moderates the product of the average concentration and the ebb tidal flux.

However, vertical gradients in salinity and density-induced circulation also affect the export rate of salt from an estuary. In the case of highly stratified estuaries, flushing of saline water is usually less effective than in well mixed and partially stratified estuaries despite continuous outflow of surface water. Flushing of salt in these systems is brought about by the entrainment of the saline bottom layer by the outflowing surface layer (Dyer 1973, Officer 1976, Dyer 1986). Although entrainment enhances the volume delivery of estuarine water to the coastal sea above that purely ascribable to riverine discharge (Fischer et al. 1979), it is not as effective a mixing mechanism as turbulent diffusion which predominates in well mixed estuaries and is active in partially stratified estuaries (Dyer 1986). Additionally, the presence of the bar at the mouth or sills located elsewhere in an estuary are known to cause topographic blocking of the downstream movement of saline bottom water on the ebb tide in South African systems (Largier & Slinger 1991, Largier *et al.* 1992, Slinger *et al.* 1993), further inhibiting the rate at which salt is exported from stratified systems. The reduced salt export under stratified conditions is modelled by the inclusion of a stratification export function which moderates the product of the average concentration, the ebb tidal volume flux and the salinity export multiplier. This function is assigned values of unity for vertically mixed conditions, slightly lower values for partially stratified conditions and still lower values for highly stratified flows. The reduced expulsion of salinity during ebb tides also compensates for any simultaneous influx of saline bottom water which may occur owing to multi-layer flow in the mouth region of a highly stratified estuary - a process which is deemed less significant in the bar-built estuaries of South Africa than in the wedge-shaped systems of Europe and North America (Officer 1976, Jay & Smith 1988) and so is not modelled explicitly.

The salt export rate is given by:

$$x_{22} = (x_2/x_1).x_{14}.\mathrm{SEM}(x_5,x_6).\mathrm{SEF}(x_3,x_4) \quad \text{if } x_{14} < 0$$
$$= 0 \quad \text{otherwise}$$

where x_{22} = salt export ($kg.yr^{-1}$)
x_2 = salt (kg)
x_1 = water volume (m^3)
x_{14} = tidal flow ($m^3.yr^{-1}$)
SEM = salinity export multiplier (unitless)
x_5 = tidal flushing (yr^{-1})
x_6 = fresh water flushing (unitless)
SEF = stratification export function (unitless)
x_3 = stratification index (unitless)
x_4 = circulation index (unitless)

The effects of density differences on the salt flux, therefore, have been accommodated in the salt export rate by the inclusion of the salinity export multiplier and the stratification export function, while the concept of reduced salinity as a function of stratification-circulation allows for these effects in the salt import rate.

3.4 Stratification-Circulation Sector

Density stratification in an estuarine water body influences the exchange of salt between the estuary and the sea (Section 3.3), affecting the residence times of estuarine water masses (Largier 1986, Largier & Slinger 1991) and the efficiency of flushing of an estuary (Taljaard & Slinger 1993, Slinger *et al.* 1993). The link between the density stratification in an estuary and the circulation and mixing in the system traditionally has been used in classifying estuaries, with terms such as highly stratified, salt wedge, partially stratified, well mixed and fjord-like employed in describing different estuarine types (Dyer 1973, Officer 1976). Thus, an indication of the stratification-circulation is an essential component in the description of the physical state of an estuary. Additionally, the stratification and associated circulation are important determinants of the biotic response to the physical conditions of an estuarine system (Day 1981, CSIR 1992b, Hilmer & Bate 1991). The formulation of descriptors of estuarine density stratification and circulation is addressed by first considering the factors influencing density stratification, next reviewing different stratification and circulation indices and finally selecting indices appropriate for inclusion in this dynamic model.

3.4.1 Factors influencing density stratification

Principle influences on density stratification include:

- the magnitude of the river inflow to an estuary,
- the magnitude and extent of the tidal motion, and
- the morphology of the estuary (Officer 1976).

Fresh water inflow at the head of an estuary acts as a source of buoyancy (Fischer 1979). Where fresh water flows are not of such magnitude as to cause turbulent entrainment and expulsion of all salt from an estuary such as occurs in, for example, the Mgeni Estuary and other Natal estuaries during floods (Perry 1989) and the Palmiet and Great Brak Estuaries under high river flows (Largier & Taljaard 1991, CSIR 1993d), the buoyant fresh water tends to overlie the denser saline water derived from the tidal intrusion of sea water. In addition to the mixing occasioned by riverine flow, the oscillatory motion of the tide promotes vertical, lateral and longitudinal mixing (Officer 1976, Fischer 1979). The efficacy of such mixing is affected by the geometry of the estuary (Dyer 1973, Officer 1976). For instance, for a particular tidal range and volume ratio of fresh to saline water, vertical mixing is more effective in a shallow system than in a deep system, while longitudinal mixing is inhibited in a long estuary compared with a short estuary. Additionally, lateral mixing and circulation becomes progressively more important in estuaries as width and sinuosity increase. Thus strong stratification occurs in systems where the supply of buoyancy dominates mixing due to boundary and internal stirring (Largier & Slinger 1991).

Where more effective mixing occurs, the system is less stratified.

It is clear that factors such as the volume and density of the fresh water inflow, the volume and density of inflowing sea water and the shape of the estuary basin must be considered in a comprehensive description of the density stratification and circulation in an estuary. However, factors such as wind and turbulence induced by heating of the water surface also affect stratification by enhancing vertical mixing. In a study of the evolution of thermohaline structure under low fresh water flows in a closed estuary (i.e. in the absence of tidally-induced mixing), Largier and Slinger (1990) established that both wind-induced mixing and heating affected only the surface layer (*c.* 0,75 m) and were not effective in breaking down stratification in the lower layers. Similarly, Mackay and Schumann (1990) discovered that the effect of mixing due to winds was limited to the surface layer (*c.* 0,5 m) of the water column of the Sundays Estuary even under strong wind events. Consequently, the influence of wind and surface heating are not deemed significant in a bulk description of stratification and circulation in southern African estuaries.

The effects of changes in temperature on the densities of river, estuarine and sea water and hence on the stratification and circulation in an estuary also are not considered of sufficient significance for inclusion in a bulk parameter formulation. The densities of sea and estuarine water are influenced by both temperature and salinity (UNESCO 1983), while the density of fresh water (zero salinity) is influenced by temperature alone. The variations in salinity are more extreme in an estuary than in the sea and the effects of temperature differences (for instance, changes in the temperature of the inflowing fresh or sea water) on densities generally are outweighed by the effects of salinity differences (Day 1981, Dyer 1986). The study by Slinger and Largier (1990) confirmed the dominant role that salinity plays in the determination of density structure. Consequently, temperature effects are not incorporated in the modelling of estuarine stratification-circulation and density differences are assumed to originate solely from salinity differences.

Thus it remains to define descriptors of stratification and circulation which incorporate the principal influences in a dynamic manner. To this end, the different stratification-circulation indices published in the estuarine literature are reviewed and their applicability for this purpose analysed.

3.4.2 **Stratification and circulation indices**

There have been many attempts to characterise estuarine stratification and circulation according to universally applicable principles (Simmons 1955, Ippen & Harleman 1966, Hansen & Rattray

1966, Fischer 1972, 1976, Rattray & Uncles 1983, Oey 1984, Prandle 1985, Jay & Smith 1988, Scott 1993). Historically, Simmons (1955) proposed the first stratification index, suggesting the ratio of river flow per tidal cycle to the tidal prism as an indicator of the degree of density stratification in an estuary. According to his investigation, where this ratio is unity or greater an estuary is highly stratified, when the ratio is about 0,25 the estuary is partially mixed and a value of 0,1 and less is indicative of a well mixed system. The fact that estuarine geometry plays a role in the determination of the stratification-circulation was indicated first by Pritchard (1955) and examples (the Mersey Estuary and Southampton Water) where Simmons' analysis fails owing to topographic effects were indicated later by Dyer (1973). Recognising that the ratio of fresh water inflow to tidal prism is not sufficient for the characterization of density stratification, Ippen & Harleman (1966) proposed a stratification number based on the ratio of tidal energy dissipation to potential energy gain per unit mass of water. This parameter measures the amount of energy lost by the tidal wave relative to that used in mixing the water column and is analogous to the inverse of a Richardson number (Dyer 1973). However, calculation of the stratification number requires fairly accurate data on river flow and tidal elevation along an estuary; information which may not be readily available.

In 1966, Hansen and Rattray (1966) defined two non-dimensional parameters of stratification and circulation, which are used in the classification of estuaries (Dyer 1973, Officer 1976, Jay & Smith 1988, Largier *et al.* 1992). The stratification parameter is defined as $\Delta S/\bar{S}$, where ΔS is the difference between the bed and surface tidally averaged salinity and $\bar{S}$ is the depth mean value, while the circulation parameter is defined as the ratio of the residual velocity at the surface to the depth mean value and measures the strength of the baroclinic circulation. Hansen and Rattray (1966) related these parameters to their theoretical analysis of estuarine motion (Hansen & Rattray 1965) and empirically determined a further relationship to two bulk parameters: the ratio of river flow to tidal flow and a bulk densimetric Froude number based on the river flow. The bulk densimetric Froude number is given by:

$$F_m = q_f / (b.d.[(\Delta\rho_H / \rho).g.d]^{1/2})$$

where F_m = bulk densimetric Froude number (unitless)
q_f = fresh water inflow rate ($m^3.s^{-1}$)
b = channel width (m)
d = mean depth (m)
$\Delta\rho_H$ = head to mouth density difference ($kg.m^{-3}$)
ρ = ocean density ($kg.m^{-3}$)
g = gravitational acceleration ($m.s^{-2}$)

The circulation parameter was found to depend primarily on the value of the bulk densimetric

Froude number, but the stratification parameter was dependent on both of the bulk parameters. Hansen and Rattray, therefore, demonstrated conclusively that a stratification-circulation diagram could be used in classifying estuaries according to circulation and salinity structure, but failed to determine two independent bulk parameters characterising estuaries.

This deficiency was redressed when Fischer (1972) defined a non-dimensional number, known as the Estuarine Richardson number (R_E), which compares the stabilizing forces of density stratification to the destabilizing forces of velocity shear. R_E is defined as the ratio of the input of buoyancy per unit width of channel to the mixing power available from the tide:

$$R_E = g.(\Delta\rho_H / \rho).(q_f / b) / u_t^3$$

where R_E = Estuarine Richardson Number (unitless)
g = gravitational acceleration ($m.s^{-2}$)
$\Delta\rho_H$ = head to mouth density difference ($kg.m^{-3}$)
ρ = ocean density ($kg.m^{-3}$)
q_f = fresh water inflow rate ($m^3.s^{-1}$)
b = channel width (m)
u_t = root mean square tidal velocity ($m.s^{-1}$)

The stratification parameter of Hansen and Rattray was found to depend primarily on R_E, indicating that the two independent bulk parameters required for the characterisation of estuaries are the Estuarine Richardson number (yielding information on stratification) and the bulk densimetric Froude number (providing information on circulation). This is evident in Figure 3.4a, where the dependence of the stratification and circulation parameters of Hansen and Rattray on the two bulk parameters F_m and R_E is presented graphically. An additional parameter ν, which arose from Hansen and Rattray's (1965) theoretical analysis of estuarine motion, is plotted in Figure 3.4a. This parameter may be interpreted as indicating the fraction of landward transport of salinity caused by all dispersion mechanisms other than the density-driven circulation (Fischer 1972). Clearly, when the Estuarine Richardson number is large, the estuary is strongly stratified and the flow is dominated by density currents (ν is small). When R_E is small, the estuary is well mixed with negligible vertical gradient in salinity, although the longitudinal gradient in salinity could still be strong. The transition from a well mixed to a strongly stratified estuary occurs in the range $0{,}08 < R_E < 0{,}8$ (Fischer 1979).

Prandle (1985) used a dynamical analysis similar to that of Hansen and Rattray (1966) to arrive at a marginally different scheme, while Rattray and Uncles (1983) modified the application of Hansen and Rattray's linear theory (1965, 1966) to include the effects of Stoke's drift on the circulation parameter. However, the scheme of Hansen and Rattray (1966), as modified by

Fischer (1972) and Rattray and Uncles (1983), provides the most widely recognised classification index to date (Jay & Smith 1988), enabling characterisation of an estuary according to salinity and current measurements at various positions in the estuary or according to two bulk parameters.

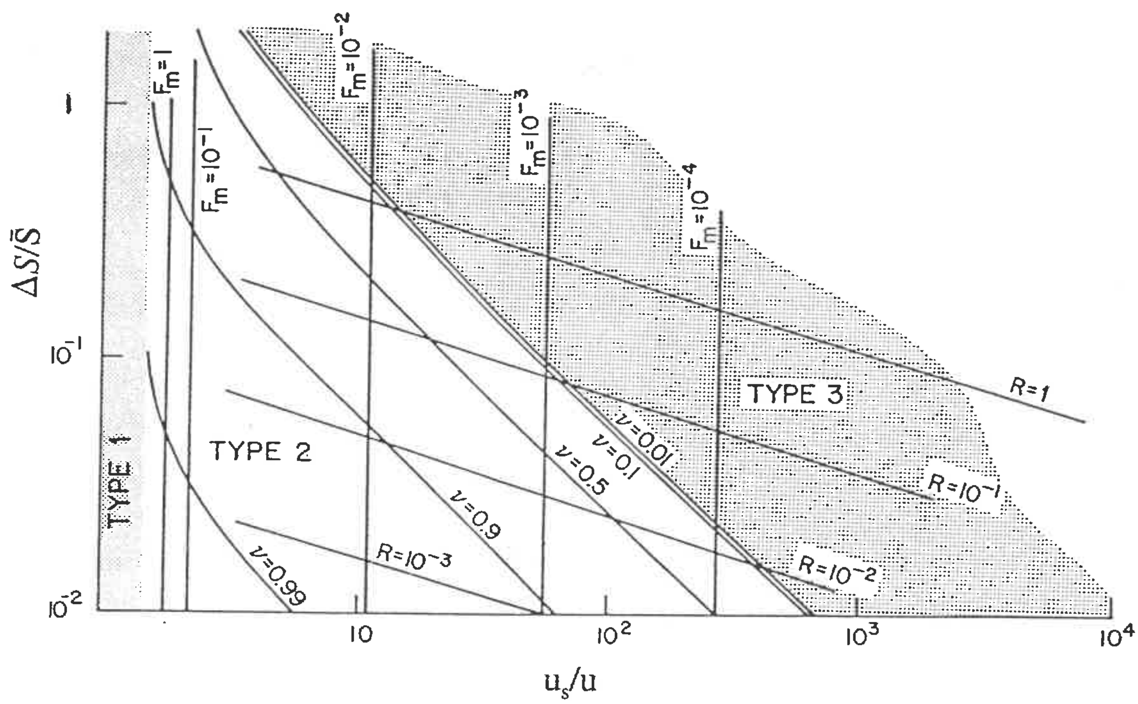

Fig 3.4a Hansen and Rattray's stratification-circulation diagram with lines of constant bulk densimetric Froude number (F_m) and Estuarine Richardson number (R_E) added. The parameter ν, indicating the fractional contribution of all mechanisms except the density-driven circulation to the landward transport of salt, is also indicated. Three types of stratification-circulation state are indicated. Type 1 estuaries are well mixed, Type 2 estuaries are partially mixed and Type 3 estuaries are highly stratified. Salt wedge estuaries are also defined as Type 4, occupying the portion of the diagram where $\Delta S/\bar{S} \geq 1$ and $u_s/u \leq 5$.

However, Jay & Smith (1988) considered the inability of Hansen and Rattray's underlying dynamical analysis to accommodate barotropic and baroclinic non-linearities a major disadvantage, particularly for shallow estuaries where these non-linearities may be important in the generation of the residual circulation. Accordingly, they proposed a two parameter classification system derived from a finite amplitude wave analysis of estuarine circulation in shallow systems (Jay & Smith 1988). The first classification parameter, an internal Froude number (F_B), reflects the stability or nonlinearity of the internal tidal oscillation:

$$F_B = (\Delta d_I / d_I).(\Delta\rho_H / \Delta\rho_V)$$

where F_B = baroclinic Froude number (unitless)
d_I = tidal cycle mean layer depth (m)
Δd_I = interface excursion depth (m)
$\Delta\rho_H$ = head to mouth density difference ($kg.m^{-3}$)
$\Delta\rho_V$ = vertical density difference in mid estuary ($kg.m^{-3}$)

The second parameter is the barotropic Froude number (F_T), which has been used as a measure of the barotropic tidal nonlinearity (Ianniello 1977):

$$F_T = \xi / d$$

where F_T = barotropic Froude number (unitless)
ξ = mean tidal amplitude (m)
d = mean depth (m).

The resulting F_T-F_B plane is subdivided into regions on the basis of the relative dominance of different vertical mixing mechanisms in the time-dependent flow and the barotropic and baroclinic residual flow generation mechanisms. Three regions, corresponding to highly stratified, partially mixed and weakly stratified systems, are distinguished (Fig 3.4b). The position of an estuary in this plane at a particular time is indicative of the prevailing stratification-circulation in the system.

The major advantages of this particular classification scheme, therefore, are:

- the whole estuary is located in one position on the F_T-F_B plane,
- changes in river flow and tidal range cause distinct positional movements of an estuary on the F_T-F_B plane, allowing qualitative prediction of the effects of such changes on the stratification-circulation regime of the system,

However, this scheme is applicable only to shallow estuaries and does not enjoy the wide acceptance of the Hansen and Rattray classification. Moreover, information on the flood-ebb excursion of the interface and the mean layer depth over the tidal cycle are required for application of the scheme. In a sense, if these values are known, F_T and F_B are bulk parameters descriptive of the baroclinic and barotropic non-linearities generating stratification and circulation in the estuary under particular discharge and tidal conditions. But, the data required to calculate these parameters are not as readily available as the information required in the calculation of the bulk densimetric Froude number of Hansen and Rattray (1966) and the Estuarine Richardson number of Fischer (1972). Additionally, no correspondence or relationship of F_T and F_B to F_m and R_E is indicated.

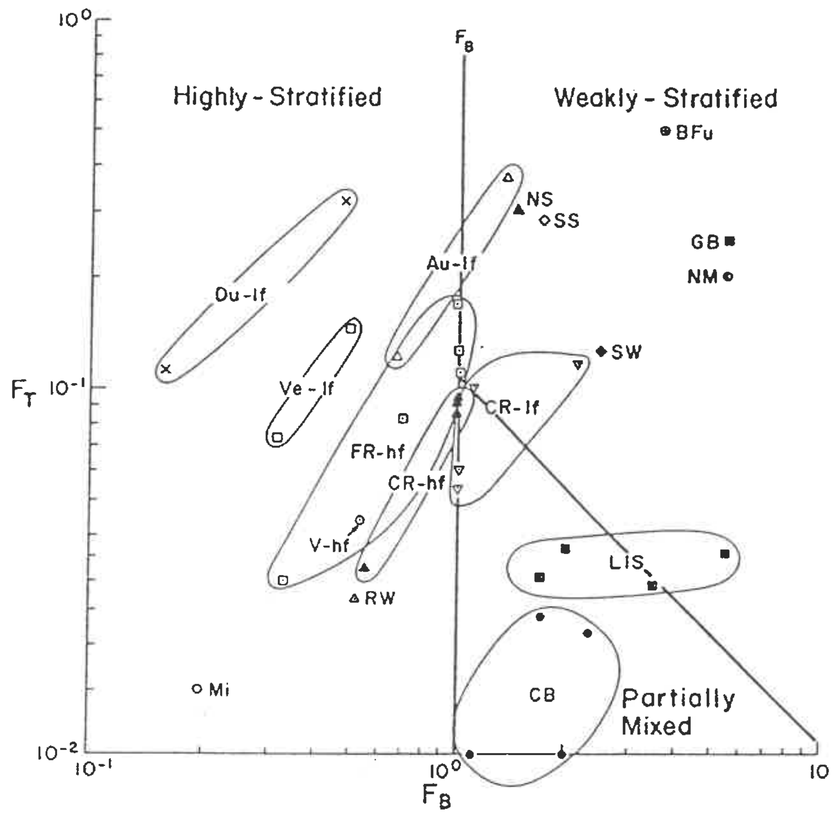

Fig 3.4b Jay & Smith's two parameter classification system for shallow estuaries based on F_B, an internal Froude number indicative of baroclinic non-linearity, and F_T , a Froude number measure of barotropic non-linearity. Symbols are: *lf* = low flow; *hf* = high flow; *Du* = Duwamish Estuary; *Ve* = Vellar Estuary; *Au* = Aulne Estuary; *Mi* = Mississipi River; *Fr* = Fraser River Estuary; *Rw* = Rotterdam Waterway; *CR* = Columbia River Estuary; *CB* = Chesapeake Bay; *LIS* = Long Island Sound; *SW* = Southhampton Water; *NS* = North Santee; *SS* = South Santee; *BFu* = Bay of Fundee; *GB* = Great Bay; *NM* = Narrows of the Mersey (after Jay & Smith 1988).

Whereas Jay and Smith (1988) criticised the fundamental dynamic analyses that led to the development of the Hansen and Rattray classification scheme and its derivatives, themselves adopting an approach which explicitly considered non-linearities in the baroclinic and barotropic forcing (but which only applies to shallow estuaries), Scott (1993) revisited the classical approach. By determining three canonical parameters related to the discharge, salinity gradient and salinity in an estuary, he was able to demonstrate that the stratification and circulation parameters of Ippen & Harleman (1966), Hansen & Rattray (1966) and Fischer (1972) may be expressed as combinations of these three fundamental parameters by assuming certain parameterisations of longitudinal diffusion and vertical mixing. For instance, the choice of the bulk densimetric Froude number, F_m, as an indicator of estuarine circulation is shown to be highly suitable, because its dependence on horizontal and vertical mixing co-efficients is minimal. Additionally, the Estuarine Richardson number, R_E, is shown to be a valid choice of bulk parameter indicative of estuarine stratification, provided that a choice to parameterise the longitudinal and vertical mixing co-efficients in terms of the strength of the tidal forcing is applicable to the estuary concerned. Thus Scott's results (1993) confirm that the bulk parameters F_m and R_E may be used to specify the location of an estuary on the Hansen and Rattray stratification-circulation diagram (Fig 3.4a).

3.4.3 **Dynamic modelling of stratification and circulation**

Thus valid indices of the stratification-circulation in an estuary include those provided by the stratification number of Ippen and Harleman (1966), Hansen and Rattray's (1966) classification scheme including the contributions of Fischer (1972), Rattray and Uncles (1983) and Scott (1993), and the classification scheme of Jay and Smith (1988).

The data required for the determination of Ippen and Harleman's stratification number are the along channel variation in tidal amplitude and accurate data on river inflow. While the latter information is exogenously specified in the estuarine systems model and is thus available for use in such a calculation, the former aspect has been explicitly excluded from consideration in the model (see Sections 3.1.1 & 3.2.4). Consequently, the stratification number is not employed to dynamically indicate the degree of stratification in the estuarine systems model.

Although, Jay and Smith's (1988) criticisms of the Hansen and Rattray classification scheme (1966) are valid, their proposed scheme is not universally accepted nor is it applicable to all estuaries, but only to shallow systems. Furthermore, the information on the flood-ebb excursion of the density interface and the mean layer depth over the tidal cycle is available for very few estuarine systems in South Africa and is not generated by the estuarine systems model. Thus, despite its potential advantages in terms of the qualitative prediction of the stratification-circulation response of a shallow estuary to alterations in discharge and tidal forcing, this classification scheme is also not suitable for use as a dynamic indicator of stratification-circulation in the estuarine systems model.

This leaves us the choice of the most universally accepted and widely applied system, that of Hansen and Rattray (1966), including all valid later modifications. The data required for the calculation of the stratification and circulation parameters of this classification scheme are the tidal mean salinity and surface to bottom differences in salinity, the residual velocity at the surface and the depth mean value. Again, these data are not available for many South African estuaries and are not generated by the model. However, the data required in the calculation of the bulk parameters F_m and R_E are more readily available and, with some adaptations, are generated by the estuarine systems model.

Accordingly, stratification and circulation indices are formulated by developing dynamic bulk parameters analogous to the Estuarine Richardson number and the bulk densimetric Froude number from model variables. Following Scott (1993), the ratio $\Delta\rho_H/\rho$ of the head to mouth density difference to the ocean density is represented by a function of the mean salinity of the system. The ratio q_f/b of freshwater discharge to the estuary width is represented by the ratio of the freshwater inflow rate x_{11} to a characteristic width B and a conversion factor (CONV), which accounts for differences in the temporal units, while the depth is represented by the water level WL. The tidal velocity is given by $|x_{14}| / (MCS.CONV)$ when the estuary mouth is open. When the mouth is closed, a tidal velocity of the order of that likely to have occurred immediately prior to the closure of the mouth, $x_{min} / (MCS.CONV)$, is used. This ensures that the model-generated Estuarine Richardson number will then alter only in response to changes in the freshwater discharge rate. Finally, first order exponential delays are applied to accommodate the tidal mean formulation of Hansen and Rattray (1966). Thus the stratification and circulation indices used in the estuarine systems model are:

$$R_{model} = g.DF(x_2/x_1).(x_{11} / (B.CONV)) / Ut^3$$

$$Ut = |x_{14}| / (MCS.CONV) \quad \text{if } |x_{14}| > x_{min}$$

$$= x_{min} / (MCS.CONV) \quad \text{otherwise}$$

$$F_{model} = x_{11} / (B.CONV.WL.[DF(x_2/x_1).g.WL]^{1/2})$$

$$\frac{dx_3}{dt} = (R_{model} - x_3) / t_1$$

$$\frac{dx_4}{dt} = (F_{model} - x_4) / t_1$$

where R_{model} = analogue Estuarine Richardson number (unitless)
g = gravitational acceleration ($m.s^{-2}$)
DF = density anomaly function (unitless)
x_2 = salt (kg)
x_1 = water volume (m^3)
x_{11} = freshwater inflow rate ($m^3.yr^{-1}$)
B = characteristic estuary width (m)
CONV = conversion factor ($s.yr^{-1}$)
Ut = tidal velocity ($m^3.s^{-1}$)
x_{14} = tidal flow ($m^3.yr^{-1}$)

MCS = mouth cross-sectional area (m^2)
x_{min} = minimum tidal flow ($m^3.yr^{-1}$)
F_{model} = analogue densimetric Froude number (unitless)
WL = water level (m)
x_3 = stratification index (unitless)
x_4 = circulation index (unitless)
t_1 = time delay constant 1 (yr)

The concept of generating these indices dynamically and using them to indicate the changes in stratification-circulation in a system over the medium to long term is novel in estuarine science.

3.5 **Flushing Sector**

Because many estuaries are the recipients of effluents and pollutant discharges, the rate at which water is flushed from an estuary has been the subject of numerous studies (Dyer 1973, Fischer *et al.* 1979, Officer 1980, Robinson 1983, Pilson 1986, Miller & MacPherson 1991, Sanford *et al.* 1992, Asselin & Spaulding 1993, MacKay 1994, Slinger & Taljaard 1996) and is considered by Sanford *et al.* (1992) to be the most important physical influence on water quality in an estuarine system. The flushing time was initially defined as the time required to replace the existing freshwater in an estuary at a rate equal to the river discharge. This concept was introduced by Ketchum (1950), who calculated the flushing time of the New York Bight under different river discharges based on the fraction of freshwater present in the basin (Dyer 1973). This definition assumes the dominant flushing mechanism to be freshwater flow.

The dominant flushing mechanism in an estuary may be the tide (Fischer *et al.* 1979). A tidal flushing or residence time for such a system is commonly expressed in terms of the tidal cycle and is calculated as the ratio of the high tide volume to the intertidal volume or tidal prism (Dyer 1973). The assumptions of this method, known as the simple tidal prism method, include:

- complete mixing of freshwater and salt water inputs occurs in the basin within a tidal cycle i.e. mixing is rapid and thorough, and
- the receiving water body (eg. the coastal ocean) is itself well flushed or vigorously mixed i.e. there is no return flow of estuarine water.

Because these assumptions do not hold for long or stratified estuaries (internal mixing is incomplete) and large estuaries or those on sheltered coasts (external mixing is incomplete), the simple tidal prism method generally yields a flushing time lower than that found in practice (Dyer 1973, Largier 1986, Sanford *et al.* 1992, MacKay 1994).

The simple tidal prism method was subsequently modified to accommodate incomplete longitudinal mixing (Dyer 1973). In this method, the estuary is divided into longitudinal sections corresponding in length to the average excursion of a particle of water on the flood tide and a flushing time for each segment is calculated as the inverse of a volume exchange ratio. The total flushing time for the estuary is then considered to be the sum of the flushing times for the individual segments. This procedure has produced realistic results for a number of estuaries, but is most accurate for long, well mixed estuaries in which the cross-sectional area increases rapidly with distance downstream (Dyer 1973). In similar vein, the fraction of fresh water method first proposed by Ketchum (1950) has been modified to accommodate incomplete longitudinal mixing and forms the basis of many box model assessments of flushing times (Officer 1980, Miller & McPherson 1991, Asselin & Spaulding 1993). Alternatively, longitudinal and vertical dispersion co-efficients may be determined from field data or via semi-analytical formulae and used in standard one and two dimensional transport-dispersion modelling (Fischer *et al.* 1979, Huizinga 1985a, West & Mangat 1986, Slinger & Taljaard 1996). The last option is particularly effective in determining the spatial distribution and temporal fate of a contaminant within a particular water body.

In a developing country such as South Africa, diffuse or non-point source pollution (arising from storm water carrying enhanced levels of pathogens, bacteria, sediments and nutrients) often poses more of a threat to human health (through recreational and informal domestic usage) and estuarine ecosystem functioning than direct or point source pollution (DWAF 1991, CSIR 1993c, MacKay 1994). Furthermore, in 1991, direct effluent discharges were limited to less than 7% of the 273 estuaries along the South African coast (CSIR 1991b). Thus an indication of the flushing of the water body is generally of more relevance than an accurate prediction of the fate of a particular substance. An additional water quality concern arises from the de-oxygenation of saline bottom water which is known to occur in highly stratified and partially mixed South African estuaries under low freshwater flows (Largier 1986, Taljaard & Slinger 1993, Slinger *et al.* 1993). Expulsion of this bottom water may be occasioned by high freshwater flows, or renewal may be brought about by tidal intrusion (Taljaard & Slinger 1993, Slinger *et al.* 1993). Thus, an indication of both the fresh water flushing and the tidal exchange of the water body is needed for an assessment of the water quality of a South African estuary.

Using time-independent longitudinal and vertical dispersion co-efficients to simulate the dynamic freshwater and tidal flushing of an estuarine water body is not an appropriate method. The fraction of fresh water method and its derivatives likewise are not suitable as fresh water flushing is assumed to occur continuously and strongly and this assumption is invalid for many South African estuaries, which experience low fresh water inputs and mouth closure at times. Furthermore, knowledge of antecedent inflows is a prerequisite in the determination of the extent of influence of freshwater and the degree of flushing occasioned by river flows. Therefore, a first order exponential delay of the inflow rate is considered to provide an appropriate estimate of the freshwater flushing:

$$\frac{dx_5}{dt} = (x_{11}/x_{11n} - x_5) / t_2$$

where x_5 = fresh water flushing (unitless)
x_{11} = fresh water inflow rate ($m^3.yr^{-1}$)
x_{11n} = fresh water inflow rate normal ($m^3.yr^{-1}$)
t_2 = time delay constant 2 (yr)

Because the freshwater run-off may itself be a major source of contaminants eg. the run-off to the Mgeni Estuary (CSIR 1993c), it is necessary to establish whether throughflow of this water is occurring within a reasonable period of time or whether mouth constriction or closure is inhibiting the fresh water flushing and causing an accumulation of poor quality water. In calculating the tidal exchange rate of the water body, the lack of longitudinal definition in the estuarine systems model precludes the use of the modified tidal prism method. This leaves the simple tidal prism method, which yields a flushing time that is the inverse of the tidal exchange rate (Dyer 1973). Accordingly an instantaneous fractional tidal exchange rate may be defined as the ratio of the tidal flow through the mouth and the water volume of the estuary. The tidal flushing is then given by a first order exponential delay of the instantaneous fractional tidal exchange rate:

$$FEX = |x_{14}| / x_1$$

$$\frac{dx_6}{dt} = (FEX - x_6) / t_3$$

where FEX = fractional tidal exchange rate (yr^{-1})
x_{14} = tidal flow ($m^3.yr^{-1}$)
x_1 = water volume (m^3)
x_6 = tidal flushing (yr^{-1})
t_3 = time delay constant 3 (yr)

This state variable indicates for instance whether the water body is flushed tidally 365 times per year on average, 10 times or only once, whereas the fresh water flushing variable provides an indication of the extent of freshwater influence. Note that incomplete internal and external mixing within an estuary will cause an overestimate of the tidal flushing. However the advantages of the formulation include the use of bulk parameters in the calculation of tidal flushing and the fact that increased tidal flushing owing to an event such as a freshwater flood or reduced flushing because of mouth closure are automatically accommodated. In the case of a freshwater flood, for instance, x_{14} would be large and negative for longer than a tidal cycle, but this would not pose any difficulties in the calculation of tidal flushing, which would just be higher than usual. In the event of mouth closure, the tidal flushing would approach zero with the persistence of the mouth closed condition.

An estimate of head to mouth differences in salinity may be derived from knowledge of the freshwater flushing and the salt content of the estuary. The assessment provided by the axial salinity function is not accurate (hence could not be used in implementing the modified tidal prism method), but is useful as a descriptor of saline conditions in the estuary and in estimating the success or otherwise of faunal recruitment.

$$ASD = ASF(x_2,x_5)$$

where ASD = axial salinity difference ($kg.m^{-3}$)
ASF = axial salinity function ($kg.m^{-3}$)
x_2 = salt (kg)
x_5 = fresh water flushing (unitless)

The state variables of fresh water and tidal flushing thus provide a basis for assessing the likelihood of de-oxygenation, stagnancy, nutrient accumulation and bacterial contamination of particular estuaries and predicting fish and invertebrate recruitment to these systems.

3.6 Sediment Sector

Estuarine sediment is derived primarily from the nearshore marine environment and the catchment. Marine sediments are characteristically coarser and more sandy than fluvial sediments which contain a higher proportion of mud and clay particles (Day 1981). In South Africa, the fluvial sediment input to the estuaries of the western and southern Cape is low, whereas the rivers of the eastern Cape, Transkei and Natal carry a heavy sediment load particularly when in flood (Day 1981). In contrast, the supply of marine sand generally is plentiful along the entire coast and many estuaries experience mouth closure when sand chokes the entrance channel (Reddering & Rust 1990). As the sediment responsible for closure and constriction of an estuary mouth is marine sand and not the fluvial sediment characterising the upper and middle reaches, the effects of accretion and erosion of sand at the mouth of the estuary as well as artificial breaching of the estuary mouth are the only aspects of estuarine morphometry considered in the model (see Section 2.1 (ii) & (iii)).

The configuration of the mouth of an estuary is determined by the competition between the scouring/erosive action of outflowing water (riverine and tidal outflow) and the action of the flood tidal currents and waves in transporting and depositing marine sand in the inlet region. The sharp reduction in wave action caused by the constriction of the entrance channel commonly results in considerable sand deposition (Day 1981), particularly under low river flow conditions and the sand sill at the mouth may then build up substantially. Under floods and high flow conditions this material is scoured out (deepening the mouth) and occasionally the sand bar is eroded away completely as occurred in the Mgeni Estuary during the October 1987 floods (Perry 1989), to be re-established rapidly once the river flows decrease (Reddering & Rust 1990). Such dramatic changes in estuarine mouth morphometry are not addressed by the estuarine systems model. Instead, the height of the sill at the mouth is regarded as representative of mouth condition and changes in the sill height are determined from the sand mass accretion and erosion rates by assuming a rectangular entrance channel connected to the estuary basin at the sill height (Fig 3.2a). The management action of mouth breaching may also affect the height of the sill at the mouth. Thus the rate of change of the sill height is given by the following differential equation:

$$\frac{dx_7}{dt} = \Sigma\, x_{7j} \qquad j = 1,3$$

where x_7 is the sill height (m), x_{71} and x_{72} are the accretion and erosion rates of the sill in $m.yr^{-1}$, respectively and x_{73} is the breaching rate ($m.yr^{-1}$).

3.6.1 **Erosion of the sand sill**

The bars at the mouths of South African estuaries consist of sand particles of different sizes packed together. The individual particles range in size from coarse grains of up to 1 mm to very fine grains of about 62,5 μm, while silt and clay particles (smaller grain sizes) are usually absent. The grain size composition and the shapes of the sand particles (eg. irregular, smooth, spheroidal) determine how densely they are packed together. Usually, a packing density of about 60% prevails, that is the ratio of space occupied by sand grains to the total volume is 60%. Conversely, the porosity, or ratio of space occupied by the fluid to the total volume, is usually about 40%. The lack of silt and clay particles, which adhere to one another and reduce the erodibility of the bed, means that the sandy entrance channel can be treated as a bed of non-cohesive sediment. Thus, the grain size distribution and the porosity of the sand in the entrance channel of an estuary are the two characteristics of the sediment which affect the erosion rate. However, it is the capacity of the outflowing water to erode and transport the sediment in the entrance channel which primarily determines the rate of erosion of the sand sill.

The erosion and transport of sandy bed sediment has been studied extensively. A variety of approaches has been adopted, from those of Einstein (1950) and Bagnold (1956, 1966) who considered the basic physics of sediment motion, through purely experimental approaches to dimensionless analysis (Ackers & White 1973, Yalin 1963). Comprehensive overviews are provided by Yalin (1977) and Dyer (1986). The formulae generally applied to sediment transport in estuaries are those of Engelund and Hansen (1967), Ackers and White (1973), Engelund and Fredsoe (1976), Van Rijn (1984a & 1984b) and Yang (1973). However, there is considerable disparity between results when these formulae are applied to the prediction of sediment transport and morphological response in particular estuaries (CSIR 1991a, DHI 1992, CSIR 1994a). In a recent review, Nakayo (1990) confirmed the earlier findings of White *et al.* (1975), as supplemented by Yang (1976), and re-established that the formulae of Engelund and Hansen (1967), Ackers and White (1973) and Yang (1973) are amongst the most reliable. To compensate for the lack of accuracy, it has become practice in estuarine applications either to apply between three and four predictive formulae and average the results of those in reasonable agreement or to customise a formula for site specific application using measured data. Examples of the former approach include studies on sedimentation in the Kromme Estuary (CSIR 1991a) and on the effect of the width of the sill at the mouth of the Touw Estuary on flood levels in the Wilderness system (CSIR 1994a). However, the latter approach is advocated by Dyer (1986), who evinces considerable scepticism regarding the reliability of the predictive formulae. Since the estuarine systems model is being formulated with the objectives of ease of application, as little dependence on measured data and as much general applicability as possible (see Sections 2.1 & 2.3), this intensely site specific approach cannot be adopted. Thus, the different formulae (hereafter referred to only by the authors' names) commonly applied to estuaries are examined for suitability of use in the estuarine systems model based on their data requirements, generic applicability and their performance in the two South African estuarine case studies.

In the Kromme sedimentation study (CSIR 1991a), the formulae of Engelund-Hansen and Ackers-White were found to agree reasonably well, whereas Yang's formula yielded considerably higher sediment transport rates. In the Wilderness study (CSIR 1994a), the formulae of Engelund-Hansen, Ackers-White and Van Rijn were in good agreement, while the predictions of the Engelund-Fredsoe formula differed substantially from the others. Thus, on the basis of the reliability of their predictions in the South African context, the formulae of Engelund-Hansen and Ackers-White appear to be worthy of consideration for inclusion in the estuarine systems model and the Van Rijn formula remains a possibility. Both the Engelund-Hansen and the Ackers-White formulae calculate total sediment transport and do not distinguish between sediment transported as bed load (rolling and saltation of particles) or suspended load. In contrast, the Van Rijn formula treats the bed and suspended loads separately. Additional factors such as the bed form, the reference bed concentration, the fall velocity and the representative particle size for the suspended load, amongst others, have to be calculated and confirmed using calibration data when applying and verifying the results of this formula (DHI 1992). As the total sediment transported by the outflowing water is of interest rather than the mode by which it is transported, the additional data and verification requirements for the effective application of the Van Rijn formula make it unattractive as an option for use in the model. The Engelund-Hansen formulation, on the other hand, requires the iterative solution of four equations at each time step (DHI 1992). While it yields relatively reliable results, so does the readily applicable Ackers-White formula. Consequently, the latter formula was chosen for application in the estuarine systems model with the following caveats:

(i) it is understood that the accuracy of the results supplied is reasonable at best, but that trends are indicated reliably,

(ii) the formula is semi-empirical in nature so, rather than including details of every parameter, an explanation of the concept and general principles will be given, and

(iii) the formula is not sacrosanct, but bearing Dyer's (1986) scepticism in mind, parameter values may be modified to reflect site-specific conditions where the calibration data indicates this to be necessary.

The Ackers-White sediment transport formula

The effects of sediment size and mobility on the capacity of the stream flow to transport sediment are included in the formula by means of three dimensionless quantities, namely the dimensionless grain diameter, the sediment mobility number and the dimensionless transport capacity. The dimensionless grain diameter is calculated as:

$$D = DIAM.[g.(s-1) / \upsilon^2]^{1/3}$$

where D = dimensionless grain diameter (unitless)
$DIAM$ = grain diameter (m)
g = gravitational acceleration ($m.s^{-2}$)
s = mass density of sand relative to fluid (unitless)
υ = kinematic viscosity of fluid ($m^2.s^{-1}$)

The mobility of the sediment is derived from the ratio of the appropriate shear force on unit area of the bed to the immersed weight of a layer of grains. Thus the sediment mobility number depends on the particle size of the sediment, the velocity of the flow and the configuration of the channel.

$$S_{mob} = [v_*^n / (g.DIAM.(s-1))^{1/2}].[U / (5.66.\log_{10}(10.h / DIAM))]^{1-n}$$
$$v_* = (g.h.i)^{1/2}$$
$$i = |TWL - WL| / l$$

$$n = 1 - 0.56.\log_{10}D$$
$$U = |x_{14}| / (MCS.CONV)$$

where S_{mob} = sediment mobility number (unitless)
v_* = shear velocity ($m.s^{-1}$)
n = transitional grain size exponent (unitless)
g = gravitational acceleration ($m.s^{-2}$)
DIAM = grain diameter (m)
s = mass density of sand relative to fluid (unitless)
U = mean velocity of flow ($m.s^{-1}$)
h = effective depth of flow (m)
i = hydraulic gradient at the mouth (unitless)
TWL = tidal water level (m)
WL = water level (m)
l = length of the estuary mouth (m)
D = dimensionless grain diameter (unitless)
x_{14} = tidal flow ($m^3.yr^{-1}$)
MCS = mouth cross-sectional area (m^2)
CONV= conversion factor ($s.yr^{-1}$)

Note that n=0 for coarse sediment and for fine sediment n=1. This corresponds to a transitional range from D=1 to D=60 and the grain sizes for sand in water at about 15 °C corresponding to these limiting dimensionless grain diameters are 40 μm and 2,5 mm, respectively.

The capacity of the flow to transport sediment is considered to depend on the sediment mobility number and the dimensionless grain diameter. The concept of a threshold of sediment movement is incorporated in the parameter A, which is the minimum value of the sediment mobility number for which transport occurs. The parameter A varies according to the grain size of the sediment. Thus, the empirical formula for sediment transport capacity, derived from almost a thousand flume experiments, is:

$$ST_{cap} = C.[S_{mob}/A - 1]^m \quad \text{if } S_{mob} > A$$
$$= 0 \text{ otherwise}$$
$$\log_{10}C = 2.86.\log_{10}D - (\log_{10}D)^2 - 3.53$$
$$A = 0.23/D^{1/2} + 0.14$$
$$m = 9.66/D + 1.34$$

where ST_{cap} = dimensionless sediment transport capacity (unitless)
C = sediment transport coefficient (unitless)

S_{mob} = sediment mobility (unitless)
A = threshold value (unitless)
m = exponent in sand transport function (unitless)
D = dimensionless grain diameter (unitless)

As the sediment mobility number increases above the threshold value, the capacity of the flow to transport sediment increases. According to Ackers and White (1973), the volumetric sediment transport rate is then given by:

$$X_{vol} = ST_{cap}.(DIAM/h).(U/v_*)^n$$

where X_{vol} = volumetric sand transport rate (10^{-6} m^3 sand per unit volume flow rate)
ST_{cap} = dimensionless sediment transport capacity (unitless)
DIAM = grain diameter (m)
h = effective depth of flow (m)
U = mean velocity of flow ($m.s^{-1}$)
v_* = shear velocity ($m.s^{-1}$)
n = transition exponent (unitless)

Thus, the Ackers-White formula can be used to derive a volumetric sand transport rate, based on knowledge of the ebb tidal flow through the mouth and on the sediment grain size characteristics of the entrance channel. However, the configuration of the estuary mouth and the porosity of the sandy bed of the estuarine inlet still have to be taken into account before the erosion rate of the sand sill can be determined.

As mentioned previously, the entrance channel is conceived as a rectangular channel of set width and length (site specific parameters), but variable height (Fig 3.2a). Erosion of sand is assumed to occur evenly across the bed of the channel, so that the erosion rate of the sand sill is given by the quotient of the eroded volume and the bed area of the channel, thus accommodating the effect of the configuration of the inlet. The eroded sediment volume, in turn, is the product of the Ackers-White sediment transport rate and the ebb tidal outflow, adjusted to accommodate the porosity of the sandy bed. Therefore, the erosion rate of the sill is given by:

$$x_{72} = X_{vol}.x_{14} / (w.l.(1- Ps)) \quad \text{if } x_{14} < 0$$
$$= 0 \quad \text{otherwise}$$

where x_{72} = erosion rate of the sill ($m.yr^{-1}$)
X_{vol} = volumetric sand transport rate (10^{-6} m^3 sand per unit volume flow rate)
x_{14} = tidal flow ($m^3.yr^{-1}$)

w = width of the estuary mouth (m)

l = length of the estuary mouth (m)

Ps = porosity of the sandy bed (unitless)

3.6.2 **Accretion of the Sand Sill**

The dynamic processes involved in the suspension, transport and deposition of marine sand under the waves and currents active in an estuary mouth are complex. It is evident, however, that on the flood tide the primary transporting medium for sand in the entrance channel of an estuary is the flood tidal current. The sediment transport rate on the flood tide could possibly be determined as for the ebb tidal current, that is, by using the Ackers-White formula (Section 3.6.1). However, the presence of waves is known to increase the concentration of sediment in suspension (Dyer 1986, Nadaoka *et al.* 1988) and to cause elevated concentrations to be transported into the entrance channel where constriction and shallowing cause a sharp reduction in wave activity (Day 1981). Enhanced deposition of sediment results (O'Brien 1969, Day 1981, Huizinga 1994). These effects cannot be ignored. A standard application of the Ackers-White formula in this situation, therefore, is not appropriate. Rather, the effects of wave stirring and site-specific aspects of the estuary mouth which influence sediment transport and deposition must be accommodated.

Wave stirring

Waves are an effective stirring mechanism, intermittently generating obliquely descending turbulent eddies when breaking or when propagating shoreward after breaking, and so causing sediment suspension at much lower equivalent velocities than a steady current (Dyer 1986, Nadaoka *et al.* 1988, Smith & Mocke 1993). Thus when stirring by waves has occurred, sediment can be transported by steady currents of relatively low velocity, currents which themselves may not exceed the threshold of movement of the particles of the sandy bed (Dyer 1986). This implies that under combined waves and currents, the mobility of the sediment is enhanced compared with under the same current without waves. In analysing experimental data, Owen and Thorn (1978) found that they were able to quantify the factor by which the transport rate under a combined wave and current regime exceeded that under the same current without waves. They termed this wave stirring factor a sand flux multiplier and determined that it assumed a value of 25 for waves with root mean square velocities of 25 $cm.s^{-1}$, lower values when the wave velocity was lower and higher values when the wave velocity was higher. In a more recent investigation, Smith and Mocke (1993) confirmed that there is a consistent correlation between sediment suspension and the passage of groups of higher waves with associated stronger orbital velocities. The effect of wave stirring in enhancing the mobility of the sediment is incorporated in the estuarine systems model in the form of a wave stirring multiplier which is an increasing function of wave height.

Site specific topography

There are additional factors which affect the quantity of sediment in suspension at an estuarine entrance. Firstly, the slope of the beach and the width of the breaker zone affect suspended sediment concentrations. For instance, at the mouth of the Great Brak Estuary, the beach slopes gradually and there is a wide breaker zone. The maximum concentration of suspended sediment in the surf zone, therefore, occurs some distance offshore of the mouth in the mid-breaker zone (CSIR 1990a). In contrast, at the mouth of the Mgeni Estuary, the beach is steep and the large waves break on an offshore bar, progressing shoreward as a bore. These bores and smaller unbroken waves penetrate to the shore, where the turbulence induced by their passage or breaking causes the suspension of considerable quantities of sediment in the immediate vicinity of the mouth of the estuary (CSIR 1992c). This is deemed the primary reason that the mouth of the Mgeni Estuary has been known to close when river flows are of the order of 7 $m^3.s^{-1}$, whereas the mouth of the Great Brak Estuary can remain open under river flows of the order of 0,5 $m^3.s^{-1}$ (Huizinga 1994). The effect of beach topography on the mobility of the sediment (that is, the ease with which it is suspended) is modelled by the inclusion of a beach slope factor, a site specific parameter moderating the wave stirring multiplier.

Secondly, another factor which may influence the amount of sediment in suspension in the water entering the entrance channel of the estuary is the presence of rocky headlands or outcrops. Recent investigations of the closure of the mouths of selected South African estuaries, has confirmed that the on-offshore movement of sediment is the primary mechanism resulting in mouth closure, rather than the longshore transport of sediment (Huizinga 1994). Thus, wave direction, which is of importance in the calculation of the direction of longshore transport, is unlikely to influence mouth closure significantly. However, topographic sheltering, such as provided by a rocky headland, may reduce wave heights at the estuary mouth considerably and thus reduce the sediment in suspension in the breaker zone. This effect is included by reducing the beach slope factor accordingly.

The quantity of sediment transported by the flood tidal currents and the waves at the mouth of an estuary is assumed to be given by the Ackers-White formula with the dimensionless mobility number modified by the product of the wave stirring multiplier and the beach slope factor. This modification accommodates the effects of waves and all relevant site specific topographic aspects on the estuary mouth. Finally, the quantity of sediment deposited in the channel is calculated as the difference between the quantity that would have been transported under the action of the flood tidal current alone (that is, the standard Ackers-White formulation) and the quantity transported under the enhanced sediment mobility associated with a combined wave and current regime (that is, the modified Ackers-White formulation). Taking into account the rectangular form of the channel and the porosity of the deposited sandy layer, the accretion rate of the sill is:

$$
\begin{aligned}
x_{71} &= (X_{mod} - X_{vol}).x_{14} / (w.l.(1 - Ps)) \quad \text{if } x_{14} > 0 \\
&= 0 \quad \text{otherwise} \\
X_{mod} &= ST_{mod}.(D/h).(U/v_*)^n \\
ST_{mod} &= C.[S_{mob}.WSM.BSF/A - 1]^m \quad \text{if } S_{mob}.WSM.BSF > A \\
&= 0 \quad \text{otherwise} \\
WSM &= WSM(WH(t))
\end{aligned}
$$

where x_{71} = accretion rate of the sill ($m.yr^{-1}$)

X_{mod} = modified volumetric sand transport (10^{-6} m^3 sand / unit volume flow rate)

X_{vol} = volumetric sand transport rate (10^{-6} m^3 sand / unit volume flow rate)

x_{14} = tidal flow ($m^3.yr^{-1}$)

w = width of the estuary mouth (m)

l = length of the estuary mouth (m)

Ps = porosity of the sandy bed (unitless)

ST_{mod} = modified dimensionless sediment transport capacity (unitless)

D = dimensionless grain diameter (unitless)

h = effective depth of flow (m)

U = mean velocity of flow ($m.s^{-1}$)

v_* = shear velocity ($m.s^{-1}$)

n = transition exponent (unitless)

C = sediment transport coefficient (unitless)

S_{mob} = sediment mobility (unitless)

WSM = wave stirring multiplier (unitless)

BSF = beach slope factor (unitless)

A = threshold value (unitless)

m = exponent in sand transport function (unitless)

$WH(t)$ = wave height (m)

3.6.3 The state of the estuary mouth

Knowing the sill erosion and accretion rates, the natural height of the sill may be determined (Section 3.6) and the degree of openness of the mouth ascertained. When the sill height exceeds the greater of the tidal or estuary water levels, the effective depth of flow at the mouth is zero (Section 3.2.4) and the mouth is closed. The marine influence on the estuarine water body then is curtailed until the mouth is breached. This may occur artificially or naturally. The artificial breaching of the mouth is facilitated by means of an exogenous function known as the breaching function. Thus:

$$x_{73} = BF(t)$$

where x_{73} = breaching rate ($m.yr^{-1}$)
$BF(t)$ = breaching function ($m.yr^{-1}$).

Natural breaching of the mouth takes place when fresh water flows cause the water level to increase and overtopping and scouring of the mouth by the outflowing volume of water occurs. The sill height then decreases below the estuarine or tidal water levels and the effective depth of flow becomes positive, indicating that the mouth is open to tidal exchange. The greater the difference between the tidal or estuarine water levels and the sill height, the deeper the mouth of the estuary and the better the exchange between the estuary and the sea. Thus the sill height, the last of the state variables for which a model equation is derived, is critical to the determination of the mouth condition of the estuary.

This completes the formulation of the physical dynamics component of the estuarine systems model.

3.7 Management Evaluation Component

The physical dynamics component of the model simulates the time-varying response of the state variables to particular management policies involving mouth breachings and water releases. The purpose of the management evaluation component is to enable appraisal of the efficacy of these management policies in the maintenance of estuarine character and functioning. This component of the model, therefore, facilitates assessment of the effects of one policy compared with another and evaluation of the desirability of these effects.

3.7.1 Considerations in the development of the management evaluation component

The state variables comprising the physical dynamics component of the model provide a comprehensive picture of the physical character of the estuary. However, every aspect of the physical dynamics is not of equal importance to management, but particular issues are deemed of more relevance in particular estuaries. For instance, the condition of the mouth is of secondary importance in an assessment of the response of the Great Berg Estuary to water release policies, because the mouth is fixed in position and permanently open, whereas salinities in the estuary are a primary consideration (DWAF 1993a). In contrast, the state of the mouth is germane to the assessment of the response of the small, periodically closed Great Brak Estuary (CSIR 1990a).

To enable selection of relevant management issues on a site specific basis and yet prevent inconsistency in policy comparison, a generic set of key issues related to alterations in freshwater flow to South African estuaries was generated and accompanying management performance indices formulated.

In formulating the management performance indices, the following aspects were considered important:

- Estuaries are inherently dynamic systems, therefore the assessment of their physical state upon imposition of a particular policy must be formulated in a dynamic manner,
- The response of the estuary to a management policy must be summarised into readily understandable indices, which contain sufficient information to enable effective decision making, that is, there must be a compromise between ease of understanding and amalgamation of information,
- As the timing and/or duration of events such as mouth openings or inundation is often critical for recruitment and water quality, the inclusion of these factors must be facilitated,
- As management objectives may differ from estuary to estuary (as do the relevant issues), the inclusion of specific objectives in the performance indices should be avoided. However, the indices should be formulated so that effects favourable to the system are scored consistently, that is a direction of increasing benefit should be indicated clearly and consistently,
- A comparative means of assessing the effects of policies on an issue will facilitate understanding of the benefits of one policy in relation to another. This may be accomplished by selecting a uniform standard of comparison for a particular estuary at the outset i.e. choosing a reference policy,
- Where several issues need to be considered to form a full picture of the performance of management policies, the facility to consider the relevant issues separately or in a composite fashion must be readily available, and
- Where value judgements or ratings occur these must be stated explicitly in order to maintain transparency in decision making.

The key issues and related management performance indices described below are deemed to satisfy the above requirements and provide a set of indices from which those appropriate to an assessment of the consequences of reduced fresh water flow to a particular estuary may be selected. Thereafter, various means of comparatively evaluating the merits of management policies are indicated briefly.

3.7.2 **Key management issues in South African estuaries**

The distinctive nature of South African estuaries and the development pressures they experience (Section 1.2), are the principal determinants of the issues considered key to an assessment of the response of estuaries to management actions such as fresh water releases and mouth breachings. Clearly, the condition of the mouth is a key issue. Generally, the percentage closure of a mouth in response to alterations in flow is estimated using simulated run-off data (CSIR 1992b, Slinger 1994). However, it is not sufficient to know only whether the mouth is open or closed for a certain percentage of the time. Information on factors such as:

- the constriction of the mouth i.e. the degree to which it is open or closed,
- the duration of mouth closure,
- the timing of mouth closure, and
- the frequency of mouth closure,

is often necessary for determining whether the character of the mouth has altered and for facilitating prediction of the response of the ecosystem to the hydrodynamics and morphology of the system. For instance, an estuary which is open at intervals throughout the year for 30 percent of the time on average, has a different hydrodynamic regime and ecosystem response from an estuary of the same shape and size which is open 30 percent of the time, but continuously at a particular time of year. An index which incorporates the complexities of mouth condition and facilitates comparative assessment of the effectiveness of management policies in maintaining the character and functioning of an estuary is required.

Investigations of the effects of alterations in fresh water flow on salinity distributions in estuaries are commissioned when abstraction of water from the catchment is considered (CSIR 1992b, Huizinga & Slinger 1994). The critical issues related to salinities are:

- the occurrence of hypersalinities,
- the persistence of hypersalinities,
- the stratification-circulation state of the estuary, and
- axial salinity differences, particularly in long estuaries.

The formulation of management performance indices reflecting these aspects is required for an adequate evaluation of possible alterations in physical character and hydrodynamic functioning of an estuary. Such indices would also provide a basis for prediction of the ecosystem response to the physical conditions.

Alterations in the pattern of fresh water inflow and mouth closure frequently result in differences in the duration and frequency of inundation of areas of the estuary. For instance, tidal variation is absent when the mouth is closed and the intertidal areas do not experience regular inundation and exposure. If the water level is fairly low, the lower intertidal zone is submerged and the elevated intertidal zone exposed. If the water level is high, large areas of the estuary can be submerged for long periods. Estuarine floral communities such as salt marshes, mangroves and intertidal macrophytes (eg. *Zostera* or *Ruppia*) respond differently to inundation and exposure events. Mangroves are sensitive to inundation as they have aerial roots, whereas intertidal salt marsh is more robust. In a similar fashion, estuarine fauna such as the marsh crab and the sand prawn require tidal inundation and exposure of their habitat to thrive. In contrast, the mud prawn is unaffected by prolonged periods of inundation. It is evident that to enable prediction of the ecosystem response to inundation and exposure events, a management performance index reflecting the occurrence and persistence of high water levels and the extent of tidal variation is required.

In South African estuaries, the presence of sills and/or constriction of the mouth is known to inhibit the outflow of basal water on the ebb tide and restrict the renewal of this water by seawater intrusion on the flood tide (Taljaard & Slinger 1993, Slinger *et al.* 1994). In such systems, reduced freshwater flow can result in an alteration in the dominance of flushing methods from those of strong fresh water flows causing scour and expulsion of old water to reliance on episodic invasion of the estuary by seawater, owing to elevated sea levels and high waves (Slinger & Taljaard 1993). Because the bottom waters of these estuaries commonly exhibit hypoxia, this change in the frequency and extent of flushing of the water body could increase the occurrence, persistence and extent of hypoxia and even lead to large-scale anoxia. A management performance index reflecting the frequency and magnitude of flushing of the water body is required. The extent of alteration in the hydrodynamic functioning of the estuary can then be assessed and the possibility of the occurrence of hypoxia and anoxia estimated.

In summary, the condition of the mouth, the inundation and exposure of areas of the estuary, the occurrence and persistence of hypersalinities, the stratification-circulation state, the axial salinity differences and the frequency and extent of flushing are key issues in the determination of the physical character and hydrodynamic functioning of an estuary and the response of its associated ecosystem to the application of different management policies. Accordingly, performance indices aimed at facilitating comparative assessment of the effects of management policies (involving freshwater releases and mouth breachings) on these key issues were developed.

3.7.3 Formulation of the management performance indices

The Management Performance Index for Mouth Condition

As freshwater deprivation is generally the fate of South African estuaries, a mouth condition index was formulated to score positively the deeper the mouth, the more frequently and the longer the period it is open under a particular management policy. The facility to incorporate aspects such as a critical season or time period when the mouth must be open (eg. for fish recruitment to occur), a critical duration of the mouth open phase (eg. for adequate exchange of the water body), or a critical depth of the mouth sufficient for active tidal exchange, was also included in the formulation of the mouth condition index. If the timing, duration or depth of the mouth are not critical in a particular estuary, this facility need not be invoked, but may be nullified by setting the critical duration to zero, the critical time interval to the full interval over which the evaluation takes place and by taking all sill height values into account. Thus the mouth condition index is given by:

$$\begin{aligned} MC^J &= x_{7c} - x_7^J \text{ if } x_{7c} > x_7^J \text{ and } t \in [a1,b1] \text{ and } t_{dur1} \geq dur1 \\ &= 0 \text{ otherwise} \end{aligned}$$

where MC^J = mouth condition under policy J (m)
x_{7c} = critical sill height (m)
x_7^J = sill height under policy J (m)
$a1$ = initial critical time 1 (yr)
$b1$ = final critical time 1 (yr)
t_{dur1} = duration of mouth deepening event (yr)
$dur1$ = critical duration 1 (yr)

The aggregate value of the mouth condition index under a particular management policy J is evaluated over the selected time period and is then compared with the aggregate value over the same time period under a selected reference policy R. The resulting unitless index is termed the management performance index 1 and provides the means whereby a comparative assessment of the effects of different management policies on the condition of the mouth can be undertaken. A value of the management performance indices greater than one indicates that policy J benefits the system above the reference policy. The higher the management performance index 1, the more beneficial the policy J for the condition of the mouth of the estuary.

$$PI_1 = \int_T MC^J \, dt \Big/ \int_T MC^R \, dt \quad \text{for } T = [t_i, t_f]$$

where PI_1 = management performance index 1 (unitless)

T	= time interval over which management performance is evaluated (yr)
t_i	= initial time (yr)
t_f	= final time (yr)
MC^J	= mouth condition under policy J (m)
MC^R	= mouth condition under the reference policy R (m)

The Management Performance Index related to Salinity Distributions

The presence of strong axial salinity gradients in an estuary is indicative of the existence of a brackish water component (often lacking in severely disturbed estuaries), while gradients in excess of 20 $kg.m^{-3}$ have been associated with enhanced recruitment of fish (Dr A Whitfield, *pers. comm.*). In consequence, a salinity index which scores large estimated head to mouth salinity differences highly was formulated. The facility to score differences in excess of a critical minimum was included to accomodate systems where management performance is assessed by criteria such as the persistence of an axial gradient in excess of 20 $kg.m^{-3}$ and the absence of hypersalinities. Additionally, potentially significant factors for estuaries such as the timing and duration of a salinity event were included in the formulation of the salinity index. Where timing or duration are not important to an assessment of the management performance of an estuary, these may be excluded voluntarily from the salinity index as from the mouth condition index. It must be borne in mind that the salinity index does not provide an accurate reflection of the persistence of axial salinity gradients, but an informed estimate based on knowledge of the freshwater flushing and salt content of the water body.

$$SI^J = ASD^J - ASD_c \text{ if } ASD^J > ASD_c \text{ and } t \in [a2,b2] \text{ and } t_{dur2} \geq dur2$$
$$= 0 \text{ otherwise}$$

where	SI^J	= salinity index under policy J ($kg.m^{-3}$)
	ASD^J	= axial salinity difference under policy J ($kg.m^{-3}$)
	ASD_c	= critical axial salinity difference ($kg.m^{-3}$)
	a2	= initial critical time 2 (yr)
	b2	= final critical time 2 (yr)
	t_{dur2}	= duration of salinity event (yr)
	dur2	= critical duration 2 (yr)

By integrating the salinity index under a particular management policy J over a selected time interval and dividing this value by the comparable integral under the reference policy R, a unitless index known as the management performance index 2 is generated. This index enables a quantitative comparison of the effects of different management policies on salinity distributions to occur. A value greater than one indicates that a particular policy benefits the system above the reference policy. The greater the management performance index 2, the larger the axial salinity gradients and the lower the likelihood of hypersalinities, that is, the more beneficial the management policy.

$$PI_2 = \int_T SI^J \, dt \Big/ \int_T SI^R \, dt \quad \text{for } T = [t_i, t_f]$$

where PI_2 = management performance index 2 (unitless)
T = time interval over which management performance is evaluated (yr)
t_i = initial time (yr)
t_f = final time (yr)
SI^J = salinity index under policy J ($kg.m^{-3}$)
SI^R = salinity index under the reference policy R ($kg.m^{-3}$)

The Management Performance Index for Stratification State

The stratification-circulation state is an integral component of the physical character of an estuary and is a determining factor in the structure of its associated ecosystem. However, in the management arena there is little understanding of circulation processes in estuary, whereas the concept of stratification is appreciated. This led to the decision to describe the physical response of the water column to imposed management policies solely in terms of stratification. Because a reduction in freshwater flow to an estuary means a reduction in the buoyancy input to the system and hence a reduction in the prevailing state of stratification of the water body, an index of the stratification state was formulated to score positively the higher the degree of stratification. Where it is important that the state of stratification be maintained above a certain level, for instance that the water body remain highly stratified, this may be accomodated by setting a non-zero critical minimum value. Similarly the facility to include aspects such as the timing and duration of stratification events is incorporated. Thus, the stratification state is given by:

$$SS^J = x_3^J - x_{3c} \quad \text{if } x_3^J > x_{3c} \text{ and } t \in [a3, b3] \text{ and } t_{dur3} \geq dur3$$
$$= 0 \quad \text{otherwise}$$

where SS^J = stratification state under policy J (unitless)
x_3^J = stratification index under policy J (unitless)
x_{3c} = critical stratification index (unitless)
$a3$ = initial critical time 3 (yr)
$b3$ = final critical time 3 (yr)
t_{dur3} = duration of stratification event (yr)
$dur3$ = critical duration 3 (yr)

The aggregate of the stratification state under a policy J is compared with the aggregate under the reference policy R over a selected time period to yield an index termed the management performance index 3. This index facilitates quantitative comparison of the characteristic stratification states resulting from the application of different management policies. A value for the index in excess of unity implies that the policy J is more favourable to stratification in the system than the reference policy.

$$PI_3 = \int_T SS^J\,dt \Big/ \int_T SS^R\,dt \quad \text{for } T = [t_i, t_f]$$

where PI_3 = management performance index 3 (unitless)
T = time interval over which management performance is evaluated (yr)
t_i = initial time (yr)
t_f = final time (yr)
SS^J = stratification state under policy J (unitless)
SS^R = stratification state under the reference policy R (unitless)

The Management Performance Index for Flushing

The water quality in an estuary where the water body is exchanged frequently is generally better than a comparable estuary (in terms of size, basin shape and quality of the inflowing water) where the water body is exchanged infrequently. As exchange of the water body of South African estuaries occurs primarily by tidal flushing when they are subjected to freshwater deprivation (Slinger *et al.* 1993), an index of flushing state which scores positively the greater the tidal flushing rate under a particular policy, was formulated. Provision was made for including factors such as a minimum acceptable tidal flushing rate, timing and/or duration of a tidal flushing event to accomodate estuaries where the tidal flushing must exceed a critical minimum value, prevail for longer than a critical minimum period and occur at a certain time of year for the ecological functioning of the estuary not to be impaired.

$$FS^J = \begin{cases} x_6^J - x_{6c} & \text{if } x_6^J > x_{6c} \text{ and } t \in [a4, b4] \text{ and } t_{dur4} \geq dur4 \\ 0 & \text{otherwise} \end{cases}$$

where FS^J = flushing state under policy J (yr^{-1})
x_6^J = tidal flushing under policy J (yr^{-1})
x_{6c} = critical tidal flushing (yr^{-1})
$a4$ = initial critical time 4 (yr)
$b4$ = final critical time 4 (yr)
t_{dur4} = duration of tidal flushing event (yr)
$dur4$ = critical duration 4 (yr)

The ratio of the integral of the flushing state under a particular management policy J to the integral of the flushing state under the reference policy R over a selected time period forms the management performance index 4. This index provides a means of comparing the efficiency of different policies in enhancing the tidal flushing of an estuary. A value for the index greater than one implies that tidal flushing is more efficient under policy J than under the reference policy. The policies maximising tidal flushing are considered most beneficial.

$$PI_4 = \int_T FS^J \, dt \Big/ \int_T FS^R \, dt \quad \text{for } T = [t_i, t_f]$$

where PI_4 = management performance index 4 (unitless)
T = time interval over which management performance is evaluated (yr)
t_i = initial time (yr)
t_f = final time (yr)
FS^J = flushing state under policy J (yr^{-1})
FS^R = flushing state under the reference policy R (yr^{-1})

The Management Performance Index related to Inundation and Exposure

To facilitate the assessment of the efficacy of a management policy in meeting the inundation and exposure requirements of estuarine flora and fauna, inundation and exposure indices were formulated. The inundation index scores positively the greater the extent of inundation above a certain critical level. Similarly, the exposure index scores positively the greater the exposure below a selected critical level. This allows evaluation of both the degree of inundation of the elevated salt marsh communities (which require flooding at regular intervals) and the exposure of intertidal macrophytes, for instance, owing to the imposition of a particular management policy. If the critical water levels for inundation and exposure are set equal to the mean tidal level in the estuary, then the degree and extent of tidal variation under the imposed management policy may be assessed. Additionally, provision is made for the inclusion of aspects such as timing and duration of the inundation and exposure events as critical issues. Therefore, the inundation and exposure indices are:

$$II^J = WL^J - WL_{ci} \quad \text{if } WL^J > WL_{ci} \text{ and } t \in [a5,b5] \text{ and } t_{dur5} \geq dur5$$
$$= 0 \quad \text{otherwise}$$
$$EI^J = WL_{ce} - WL^J \quad \text{if } WL_{ce} > WL^J \text{ and } t \in [a6,b6] \text{ and } t_{dur6} \geq dur6$$
$$= 0 \quad \text{otherwise}$$

where II^J = inundation index under policy J (m)
WL^J = water level under policy J (m)

Wl_{ci}	=	critical water level for inundation (m)
a5	=	initial critical time 5 (yr)
b5	=	final critical time 5 (yr)
t_{dur5}	=	duration of inundation event (yr)
dur5	=	critical duration 5 (yr)
EI^J	=	exposure index under policy J (m)
WL_{ce}	=	critical water level for exposure (m)
a6	=	initial critical time 6 (yr)
b6	=	final critical time 6 (yr)
t_{dur6}	=	duration of exposure event (yr)
dur6	=	critical duration 6 (yr)

The value of the aggregates of the inundation and exposure indices under a particular management policy J over a selected time interval and the corresponding aggregates under the reference policy R are compared to yield management performance indices 5 and 6, respectively. If inundation of certain areas of an estuary is considered desirable, then those policies yielding large values of performance index 5 are considered beneficial to the system. Where exposure of specific areas is deemed necessary, those management policies generating high values of performance index 6 are favourable to the estuary. Where both inundation and exposure requirements exist (or the existence of intertidal areas is a requirement) those policies yielding reasonably high values of both management performance indices are the most favourable to the system.

$$PI_5 = \int_T II^J \, dt \Big/ \int_T II^R \, dt \quad \text{for } T = [t_i, t_f]$$

$$PI_6 = \int_T EI^J \, dt \Big/ \int_T EI^R \, dt \quad \text{for } T = [t_i, t_f]$$

where	PI_5	=	management performance index 5 (unitless)
	T	=	time interval over which management performance is evaluated (yr)
	t_i	=	initial time (yr)
	t_f	=	final time (yr)
	II^J	=	inundation index under policy J (m)
	II^R	=	inundation index under the reference policy R (m)
	PI_6	=	performance index 6 (unitless)
	EI^J	=	exposure index under policy J (m)
	EI^R	=	exposure index under the reference policy R (m)

3.7.4 Evaluating Management Performance

The issues relevant to an assessment of the effects of various management policies on a particular estuary are selected on a site specific basis and need not include all of the management performance indices generated thus far. At present the significant issues are identified through:

- preliminary studies conducted or commissioned by the Department of Water Affairs and Forestry when development of a river system is under consideration,
- investigations conducted or commissioned by the Department of Water Affairs and Forestry when further development of a river system is planned or revision of existing management policies is under consideration,
- official complaints or publicly expressed concerns regarding the functioning or management of an estuary by members of the public or local /regional authorities.

Once the issues of concern have been identified, these are related back to driving variables or physical environmental considerations, enabling relevant indices for the evaluation of management performance to be selected. For instance, the concern of farmers about the salt content of water used for irrigation in the upper reaches of the Great Berg Estuary caused the axial distribution of salinities to be identified as a significant management issue (DWAF 1993a).

Clearly, value judgements are involved in the selection of significant management issues and relevant indices. These judgements may be on the part of the public, users of the system eg. farmers, officials of the Department of Water Affairs and Forestry or estuarine scientists. Whichever means of selection or method of judgement is used, it is important that the reasons for the selection of one management issue above another are documented, because this assists in clarifying the implicit value judgements.

Once the significant management issues for a particular estuary or estuarine type have been selected and associated management performance indices implemented, a consistent set of descriptors of the response of the system to particular management policies is generated. These provide quantitative criteria whereby evaluation of the acceptability of the different responses can be made. This evaluation may be undertaken in the following ways:

- By implementing the multi-criteria decision making process as advocated by Stewart *et al.* (1993). In this method, different interest groups representing wide public interests would be involved in explicit trade-offs and rankings of policies based on the values of the management performance indices under the various management policies. This approach is in agreement with the Integrated Environmental Management Procedure (DEA 1992), which has as its philosophical basis that responsible decision making can only be undertaken in a transparent and participatory fashion.
- Where the significant issues have already been identified in consultation with the public and a representative committee is in place, only the issues of concern need be considered.

Comparative judgements as to acceptability may then be undertaken in this forum. This is a modified version of the aforementioned approach.

- Where there is one overriding issue of concern, the evaluation of acceptability need only be based on optimising the relevant management performance index.
- Where the relative importance of one issue in relation to another is known, a composite index may be generated by combining different management performance indices according to agreed weightings. This should only be undertaken when those involved in the decision making are comfortable with the concept and in conjunction with an analysis of the sensitivity of the resulting policy ranking to the selected weighting. This approach may be combined with those already mentioned when there is sufficient information and/or consensus.

Thus a number of evaluation methods exist, ranging from wide public participation and consideration of all available information to numerical optimisation of single or composite management performance indices. No one method is considerably better than the rest. Rather, it is important to select a process of evaluation appropriate to the level of sophistication of the decision makers. Management is a people-related process and decision making is enhanced when there is a feeling of confidence and control of processes and decisions. Thus it is more important to select an appropriate process and generate positive emotions in those involved than to implement methods that would yield optimal results in a purely scientific sense (eg. numerical optimisation of performance indices).

In summary, the management evaluation component allows one to compare the effects of different management policies by summarising the time series output of the model in a management-related fashion. It further facilitates evaluation of the efficacy of management policies by providing indices of management performance which may be used in rating the acceptability of the estuarine response.

This completes the development of the management evaluation component of the estuarine systems model. A listing of the equations for both the physical dynamics and the model evaluation components of the model is contained in Appendix A.

4. MODEL IMPLEMENTATION AND VALIDATION

The physical dynamics component of the estuarine systems model comprises a system of seven first-order, inhomogenous, non-linear, ordinary differential equations, that is:

$$\frac{dx_j}{dt} = F_j(\mathbf{x}, \mathbf{p}, t) \quad j = 1, 2, ..., 7$$

where $\mathbf{x} = (x_1, x_2, ..., x_7)^T$ and $\mathbf{p} = (p_1, p_2, ..., p_{37})^T$ are the state variables at time t and the parameters of the system, respectively. The management evaluation component of the model, on the other hand, involves the appraisal and possible optimisation of functions of these state variables and parameters over a portion of seven dimensional state space. Therefore, implementation of the management evaluation component of the model must follow implementation of the physical dynamics component. The implementation and validation of the physical dynamics aspects of the model will thus be undertaken first.

4.1 Implementation and Validation Procedure

The estuarine systems model was developed for application to South African estuaries, with the distinctive geophysical features of these systems and their associated management requirements in mind (Section 1.2). Thus, implementation of the model on at least one South African estuary (and preferably more) is vital to the determination of the success of the modelling endeavour. However, establishing the site specific applicability of the model is not sufficient to generate confidence in the validity of the model. The effects on the simulation results of inaccuracies in the calibration data and/or uncertainties in the parameter values need to be understood to establish the degree of confidence which may be placed in model results. Additionally, the ability of the model to simulate observed physical characteristics of estuaries (eg. permanently open or intermittently closed mouth, distortion of the tidal signal, generation of overtides) needs to be explored. Therefore, the implementation and validation of the physical dynamics component of the model involved the following phases:

(i) selecting appropriate sites for implementation,

(ii) assigning parameter values and determining function equations so that model outputs reflect the physical conditions of the selected sites, i.e. site specific calibration,

(iii) choosing an appropriate numerical technique for solving the differential equations,

(iv) testing the sensitivity of the system response to uncertainty in parameter values, and

(v) investigating aspects of the dynamic behaviour simulated by the model and comparing these with known behavioural responses in estuaries.

Details of the execution of each of these phases are given in this chapter, while the implementation of the management evaluation component of the model is described in the next chapter (Chapter 5).

4.2 Selection of Case Study Sites for Model Implementation

More than one site specific application is required to demonstrate the applicability of the estuarine systems model to South African systems. However, implementation of the model on three or more case studies involves the collation and detailed interpretation of considerable amounts of data without substantial gain in understanding. Two case studies, therefore, were deemed sufficient for the purpose of demonstrating that the model is applicable to South African estuaries, in general, while eliminating unnecessary detail.

The factors considered in the selection of the case study sites included:

- the availability of data for testing and calibration,
- the type of management problem associated with the estuary,
- the Whitfield category of the estuary,
- the physical character of the system eg. stratification-circulation regime, and
- current estuarine research foci.

The two case study sites chosen were the Great Brak Estuary and the Kromme Estuary, more detailed site descriptions of which follow. However, the major reasons for their selection included that both have been studied intensively over a period of years resulting in the availability of a wide range of data. Additionally, both systems are impounded immediately upstream of the estuaries and have substantially reduced fresh water inflows. Strategies for water releases which benefit the physical and ecological environment of the downstream estuaries while still ensuring that the water demands of other user sectors are met, are a management necessity. Furthermore, where the Great Brak is a temporarily open system, the Kromme Estuary is permanently open, that is, the estuaries fall into different Whitfield categories. The Great Brak Estuary exhibits stratification in the deep portions of the upper and middle reaches, whereas the water column of the Kromme Estuary is generally well mixed with hypersalinities occurring in the upper reaches. Finally, the co-ordinated research programme on decision support for the conservation and management of estuaries (CERM 1992) decided by consensus to focus attention on these two systems. As implementation of the estuarine systems model forms a component of the research project and the results from the estuarine systems model are necessary to the success of this research endeavour (Slinger 1996), it seemed sensible to select the same case studies.

4.2.1 The Great Brak study site

The Great Brak Estuary, located on the southeastern coast of Africa approximately 24 km east of Mossel Bay, drains a forested, semi-arid catchment 192 km^2 in extent (Morant 1983). The rainfall of the region is weakly seasonal with peaks in spring and autumn. Although the natural mean annual run-off is estimated to be 34 x 10^6 m^3 (Bruwer 1987), the area is subject to droughts and intermittent flooding and the annual run-off may be as large as 44,5 x10^6 m^3, as in 1962/63, or as little as 4,3 x 10^6 m^3, as in 1979/80 (Morant 1983). In 1990, the construction of the Wolwedans Dam with a capacity of 23 x 10^6 m^3 some three kilometres upstream of the estuary, exacerbated the effects of agricultural development and afforestation of the catchment by further decreasing the supply of fresh water to the estuary (CSIR 1990a, Slinger *et al.* 1994). Presently, the estuary receives an average fresh water inflow of between 3 and 30% of the natural mean annual run-off.

In common with many South African estuaries the Great Brak Estuary is a small system with a high tide area of about 0,6 km^2 and a length from head to mouth of 7,4 km (Figure 4.2a). The tidal prism (of order 3 x 10^5 m^3) is of insufficient volume to maintain an open mouth at times of low river flow when high wave energies and a plentiful supply of marine sand combine to ensure that sand accumulates in the lower reaches (CSIR 1992a, Slinger *et al.* 1994). A sand berm is then built between the estuary basin and the sea and the mouth closes (Huizinga 1994).

When the mouth is closed, salinities in the estuary generally exceed 20 mg.l^{-1} in all but the head reaches (Figure 4.2b), where salinities of between 5 and 10 mg.l^{-1} have been measured. Stratification of the water column is limited to the deeper pools (depths between - 0,5 and -3,5 m to MSL), where dissolved oxygen levels of below 20% saturation and elevated levels of dissolved reactive phosphate and ammonia occur (CSIR 1992a, Taljaard & Slinger 1993, Slinger *et al.* 1994). The estuary persists in this state until the mouth is breached, either naturally or mechanically.

Upon breaching, water drains from the estuary scouring the entrance channel. Tidal influence is re-established and the water of the lower reaches is renewed by active exchange with the sea. The middle reaches of the estuary are characterised by weak dispersion and long residence times and dense saline water persists in the deep pools of the upper and middle reaches (Slinger *et al.* 1994). Complete flushing or renewal of the bottom water is occasioned episodically when either a major flood or seawater intrusion event occurs (CSIR 1993d, Slinger & Taljaard 1993).

The condition of the mouth and the water quality of the system were identified as issues of serious concern when a management plan for the Great Brak Estuary was first developed in 1990

(CSIR 1990a). The management plan involves the release of 2 x 10^6 m^3 per annum from the Wolwedans Dam together with mechanical breaching of the mouth when necessary. However, the effects of the construction of the impoundment on the downstream estuary were not addressed quantitatively in this study. Rather, the impacts of the development were assessed qualitatively, mitigatory uses of the allocated quantity of water were investigated and monitoring of key aspects of the physical and biological environment of the estuary recommended. The subsequent development of the estuarine systems model and the monitoring programme have supplied a predictive tool and data for assessing the consequences of the reduced fresh water flow to the Great Brak Estuary and comparing the efficacy of different uses of the total annual allocation of water to the estuary. The Great Brak Estuary is thus a suitable study site for application and testing of the model.

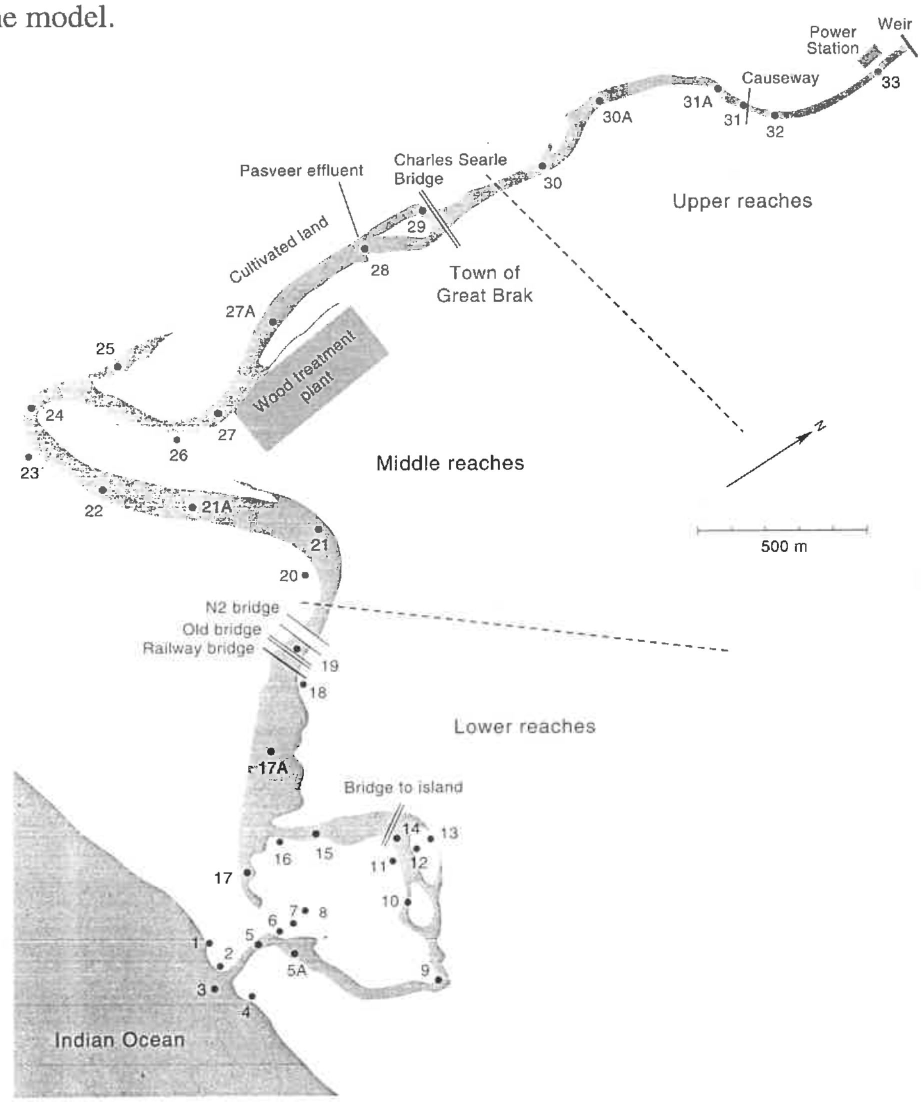

Figure 4.2a Map of the small, intermittently closed Great Brak Estuary situated on the southeastern coast of Africa. Hydrographic station positions are indicated by the numbers located along the channel of the estuary.

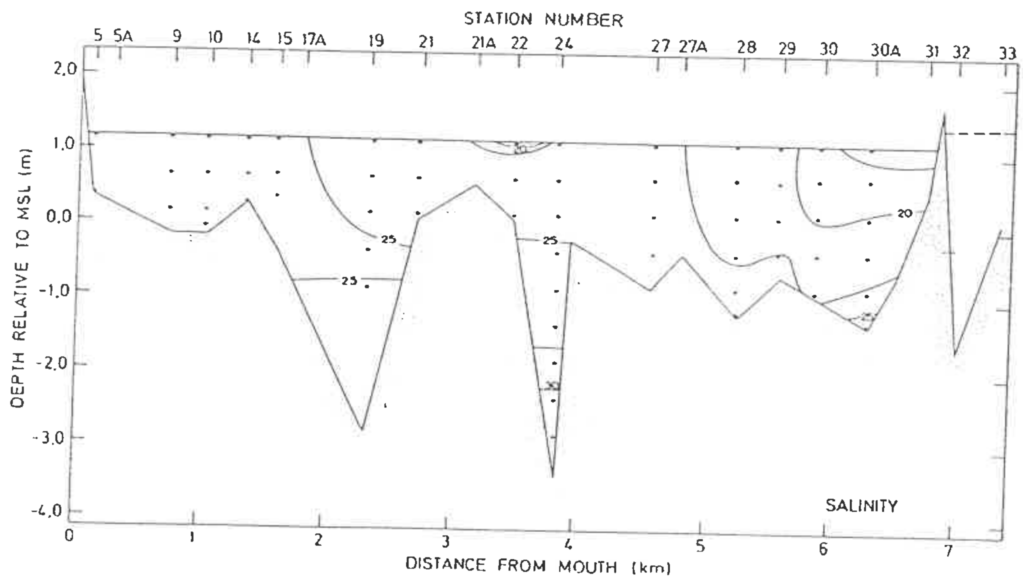

Fig 4.2b Longitudinal salinity profile ($kg.m^{-3}$) of 28, 29 November 1990 when the mouth of the Great Brak Estuary was closed. The river is to the right of station 31 and the sea to the left of the closed mouth.

4.2.2 **The Kromme study site**

The Kromme Estuary (Figure 4.2c), located on the east coast of South Africa approximately 80 km west of Port Elizabeth, is a permanently open system discharging into St Francis Bay (Bickerton & Pierce 1988). For 6,6 km from the mouth, the estuary is characterised by channel depths of 1,5 m and a sandy bottom substrate (CSIR 1991a). Upstream of this, the estuary deepens to between 3 m and 5 m and current velocities generally are lower than 0,3 $m.s^{-1}$ (values of the order of 1 $m.s^{-1}$ are common in the mouth area) (Slinger 1989). Salt marshes cover the banks of the estuary in the middle and lower reaches, while the channel meanders between vegetated cliffs in the upper reaches. The head of tidal influence is a located at a rocky sill about 14 km upstream of the mouth.

The construction of the Churchill and Mpofu Dams on the Kromme River, the latter in 1984, has led to the situation in which, apart from dam overflows under floods, the only freshwater supplied to the Kromme Estuary is the 2×10^6 m^3 per annum allocated for ecological purposes (DWAF 1993b, CSIR 1994c). At present this is released in quantities of $1{,}67 \times 10^5$ m^3 over one to two days per month.

Owing to the limited freshwater inflow to the estuary, little variation in salinities usually exists along its length. During an intensive two week field exercise conducted by the CSIR in August 1988, the estuary was persistently vertically well mixed (Slinger 1989). This finding is borne out by extensive sampling of salinities conducted by the University of Port Elizabeth over years (Prof D Baird *pers. comm.*, Prof T Wooldridge *pers. comm.*). On occasions, hypersalinities of the order of 45 $mg.l^{-1}$ have been measured in the upper reaches. As a result of the reduction in freshwater flow, the estuary is considered by ecologists to be a sheltered arm of the sea having a relatively low productivity compared with other permanently open systems (Adams & Talbot 1992, DWAF 1993b, Baird & Heymans 1994). Specific concerns include a possible increase in sedimentation in the system owing to a reduction in flooding, decreased recruitment and migration of estuarine fish because cues associated with fresh water flow are reduced and low primary productivity arising from the reduced influx of nutrients (Prof E Schumann *pers. comm.*, Dr A.Whitfield *pers. comm.*, Adams & Bate 1994).

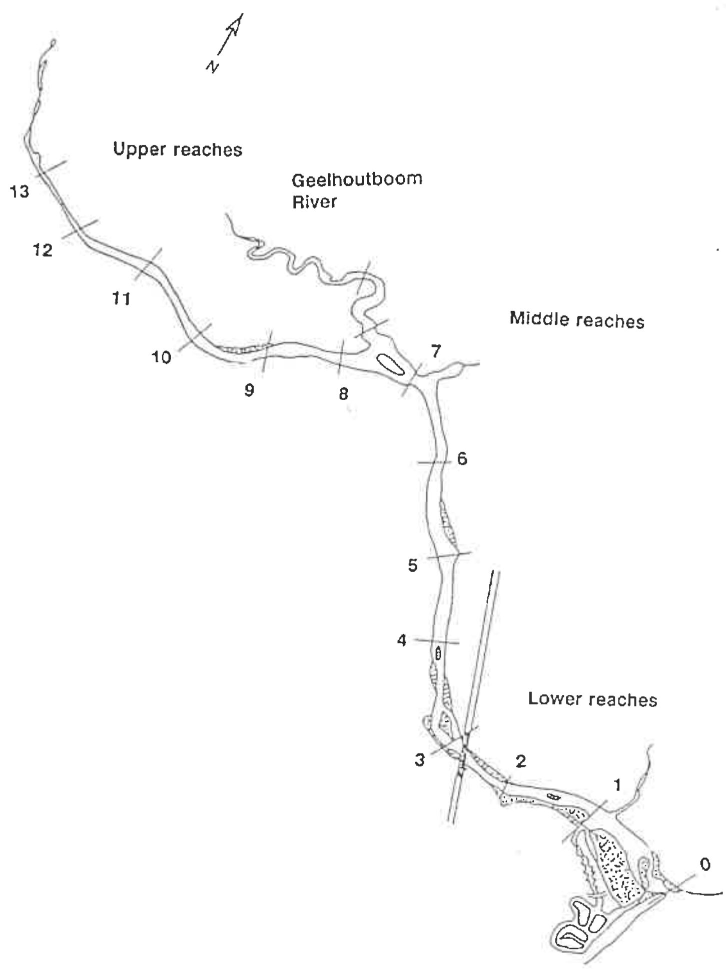

Fig 4.2c Map of the permanently open Kromme Estuary. The shallow lower reaches and deeper upper and middle reaches of the estuary are indicated while the numbers along the length of the estuary are indicative of the distance (in km) upstream of the mouth.

The optimal usage of the annual allocation to the Kromme Estuary is currently an area of concern for resource managers. During deliberations on the water resources of the Algoa Region, ecologists were unanimous in supporting further investigation of the use of this water and even suggested that the allocation to the estuary be reviewed with a view to increasing the fresh water supplied to the system (DWAF 1993b). The estuarine systems model is designed to address this type of management decision making and the Kromme Estuary was selected as an ideal case study.

4.3 Calibration of the Model on the Great Brak and Kromme Estuaries

Before simulations can be executed by the estuarine systems model, feasible parameter values must be selected, endogenous and exogenous function equations determined and an appropriate numerical method for solving the differental equations of the model chosen. Thereafter, the sectors of the model can be calibrated sequentially by holding interacting state variables constant eg. assuming that the height of the sill at the mouth remains constant for initial calibration of the water volume sector. Finally, full calibration of the model is achieved when all interactions between sectors are permitted and good correspondence between the observed site specific behaviour and the simulated response is obtained through fine tuning and adjustment of the parameter and function values. This calibration procedure is depicted in Figure 4.3a and its application to the Great Brak and Kromme Estuaries is described below. Details of the selection of the Runge-Kutta-Verner 5^{th} and 6^{th} order method for the solution of the differential equations are given in Appendix B.

4.3.1 Calibration of the water volume sector

Exogenous Functions

In calibrating the water volume sector, the focus is to achieve close agreement between the water level variations generated by the model and the water level variations recorded in the estuary under specific inflow and tidal conditions. Additionally the tidal flux computed by the estuarine systems model must correspond in magnitude and direction with the volume flows through the mouth either computed using a one-dimensional hydrodynamic model, where this is available, or estimated from field data. Specification of the exogenous fresh water inflow and tidal functions is thus necessary.

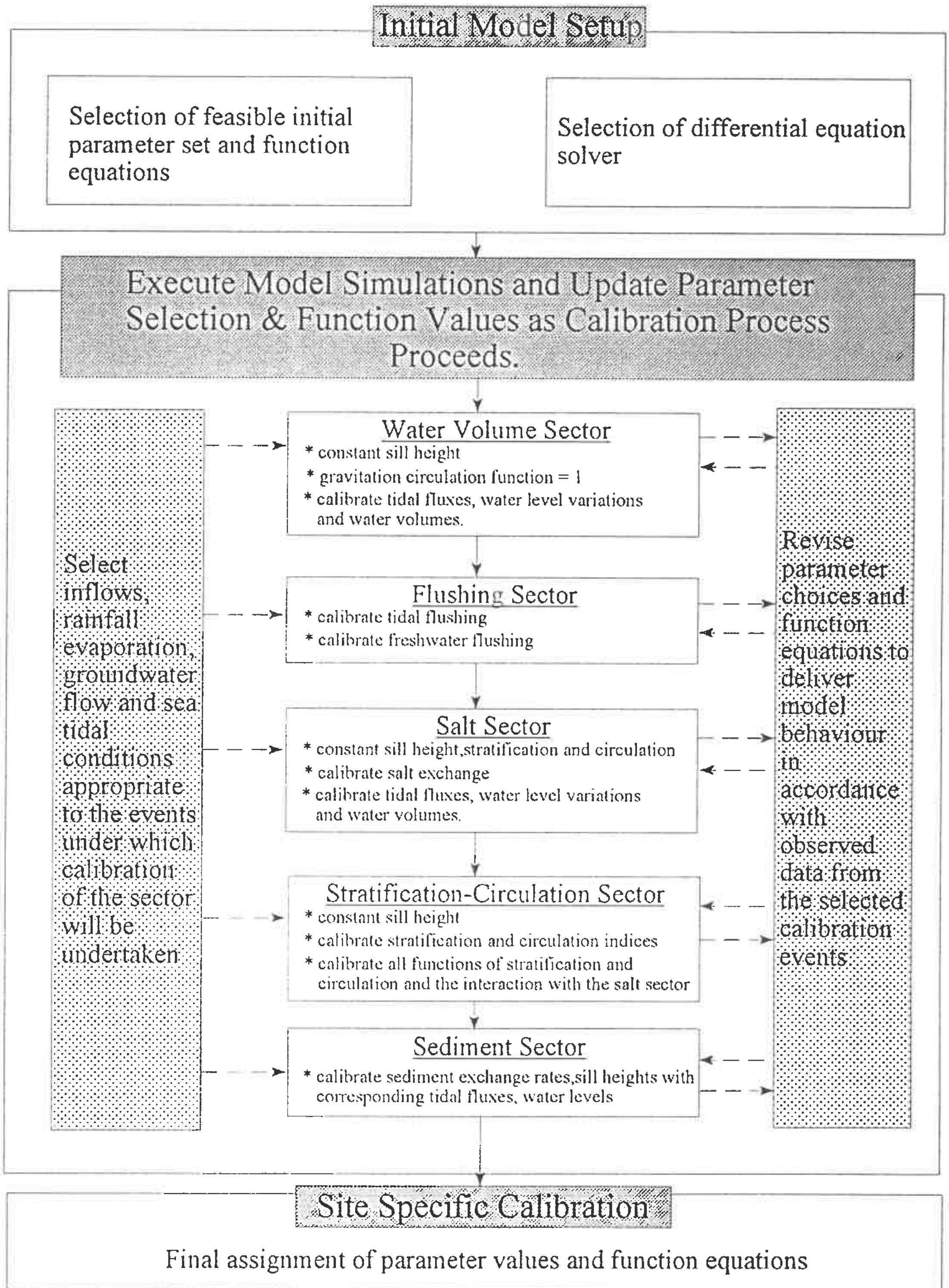

Fig 4.3a The procedure to be followed in site specific calibration of the Estuarine Systems Model

To facilitate the inclusion in the model of the current or proposed release strategies from impoundments, the inflow function is specified as the sum of component functions: a base flow function and flood function/s as depicted in Figure 4.3b. Thus:

$$IF(t) = BF(t) + \Sigma_i FF_i(t, st_i, dur_i, V_i, F_i, I_i)$$
$$BF(t) = BN.BSM(t)$$

where $IF(t)$ = inflow function ($m^3.yr^{-1}$)
$BF(t)$ = base flow function ($m^3.yr^{-1}$)
FF_i = flood function i ($m^3.yr^{-1}$)
st_i = start time i (yr)
dur_i = duration i (yr)
V_i = flood volume i (m^3)
F_i = fractional time to peak (unitless)
I_i = repeat interval i (yr)
BN = base flow normal ($m^3.yr^{-1}$)
BSM = base flow seasonal multiplier (unitless)

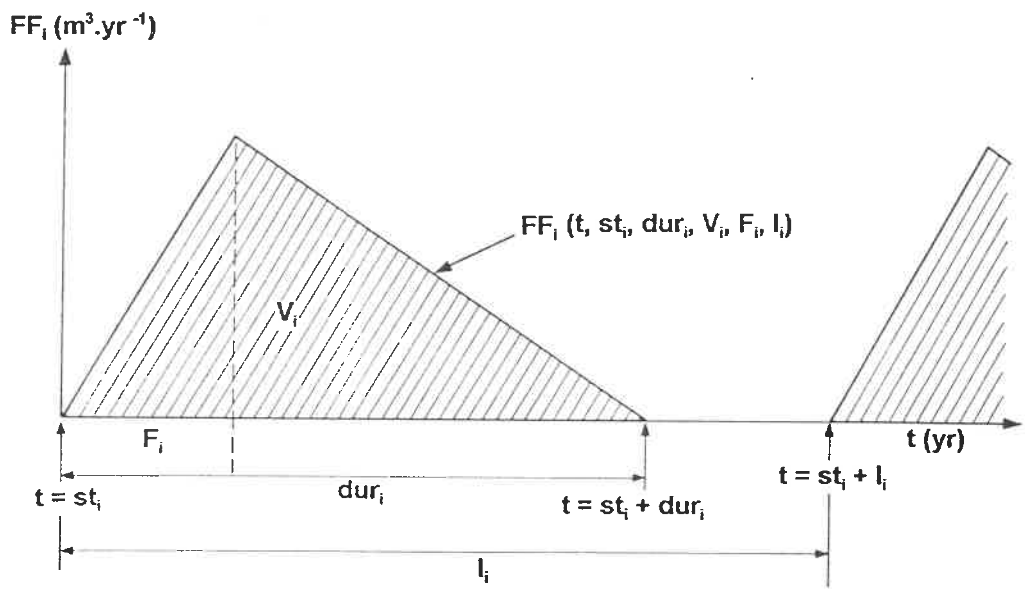

Figure 4.3b Diagrammatic representation of the flood function used to simulate fresh water releases to an estuary

The values of the parameters and component functions determining the inflow to an estuary are seldom fixed throughout application of a model to a particular case study, but are altered to reflect the inflow at the time of data gathering when calibrating or to reflect the management policy being tested. The particular choices made when calibrating different aspects of the model or conducting management policy simulations will be stated in each case.

For calibration and model testing purposes, the tide function is assumed adequately represented by a simple cosine form with semi-diurnal and spring-neap features, that is:

$$TF(t) = MSL + HLA.\cos(2\pi t/AT1).[1 + SNA.\cos(2\pi t/AT2)]$$

where TF(t) = tide function (m)
MSL = mean sea level (m)
HLA = high-low amplitude (m)
AT1 = semi-diurnal time scale (yr)
SNA = spring-neap amplitude (unitless)
AT2 = spring-neap time scale (yr).

The values of the parameters MSL, HLA, AT1, SNA, AT2, which are derived from Table II of the Tide Tables published annually by the SA Navy Hydrographer (SAN 1994), are given in Table 4.3a.

As very few estuaries in South Africa have accurate rainfall and evaporation measurements taken in their vicinity, it would be artificial to specify exogenous rainfall and evaporation functions in great detail. Instead the form of the rainfall and evaporation functions is taken as the product of the annual average values and seasonal multipliers, which average unity over a hydrological year:

$$RF(t) = RN.RSM(t)$$
$$EF(t) = EN.ESM(t)$$

where RF = rainfall function ($m.yr^{-1}$)
RN = rainfall normal ($m.yr^{-1}$)
RSM = rainfall seasonal multiplier (unitless)
EF = evaporation function ($m.yr^{-1}$)
EN = evaporation normal ($m.yr^{-1}$)
ESM = evaporation seasonal multiplier (unitless)

Thus the annual average rainfall and evaporation values are additional parameters, and the seasonal variation in evaporation and rainfall are additional function values, required for calibration of the estuarine systems model. For the Great Brak and Kromme Estuaries the necessary information was obtained from Pitman *et al.* (1981), CSIR (1990) and CSIR (1994) and is included in Tables 4.3a & b.

Knowledge of the influence of groundwater flow on the dynamics of South African estuaries is limited to a study on the Gamtoos Estuary (Pearce & Schumann 1994) and includes estimates of seepage losses from certain systems (CSIR 1990a, 1994a). In view of the dearth of data on this aspect of estuarine hydrodynamics, an estimated constant, small seepage loss (termed the groundwater normal (Table 4.3a)) is assumed to occur from the Great Brak Estuary through the extensive sand spit separating it from the sea (CSIR 1990a). The influence of groundwater flow on the dynamics of the Kromme Estuary cannot be quantified or even reliably estimated from available data and therefore is set at zero (Table 4.3a) in the full awareness that this may not reflect the real situation.

Table 4.3a Exogenous parameter values for the Great Brak and Kromme Estuaries

PARAMETER	SYMBOL	UNITS	GREAT BRAK ESTUARY	KROMME ESTUARY
Mean sea level	MSL	m	0.24	0.24
High-low amplitude	HLA	m	0.58	0.530
Semi-diurnal time scale	AT1	yr	0.001407915	0.001407915
Spring-neap amplitude	SNA	-	0.509	0.519
Spring-neap time scale	AT2	yr	0.039421613	0.039421613
Rainfall normal	RN	$m.yr^{-1}$	0.5056	0.08667
Evaporation normal	EN	$m.yr^{-1}$	1.315	0.12725
Groundwater normal	GN	$m^3.yr^{-1}$	-157680.0	0.0

The exogenous functions of inflow, tide, precipitation, evaporation and groundwater may be altered to more accurately simulate reality eg. by the inclusion of stochasticity, or to reflect a desired condition for testing eg. a period of drought. When applying the model to an individual estuary, however, it is necessary to ensure that the function values conform to the bio-geographic characteristics of the region in which the estuary is located i.e. the appropriate seasonality in rainfall is exhibited. No such restraints need be applied when investigating the response of estuarine systems at a theoretical level.

Table 4.3b Data values of the rainfall seasonal multiplier, the evaporation seasonal multiplier and the wave seasonal multiplier for the Great Brak and Kromme Estuaries. As these functions are annually periodic, only data values for the first simulation year, commencing 1 October, are supplied. Intermediate function values are derived by linear interpolation of the known data values.

TIME (yr) FROM 1 OCT	GREAT BRAK ESTUARY			KROMME ESTUARY		
	RSM(t)	ESM(t)	WSM(t)	RSM(t)	ESM(t)	WSM(t)
0.00000	1.233	0.825	1.139	2.083	0.853	1.000
0.04167	1.134	0.943	1.085	1.096	1.029	0.950
0.12500	1.510	1.375	0.950	1.085	1.328	0.950
0.20830	0.871	1.768	0.950	0.531	1.525	0.950
0.29167	0.862	1.571	1.085	0.519	1.556	1.000
0.37500	0.931	1.375	1.085	0.208	1.383	1.000
0.45830	1.440	1.178	1.139	0.035	1.375	1.050
0.54167	0.912	0.884	1.194	1.292	0.880	1.100
0.62500	0.912	0.638	1.356	0.588	0.589	1.150
0.70830	0.489	0.540	2.007	2.273	0.589	1.250
0.79167	0.686	0.432	1.628	0.196	0.487	1.150
0.87500	1.172	0.589	1.356	1.108	0.582	1.050
0.95830	1.332	0.707	1.248	3.069	0.676	1.000
1.00000	1.233	0.825	1.139	2.083	0.853	1.000

Endogenous Functions and Parameter Values

Functions descriptive of the shape of the estuary basin need to be specified at the outset. These include the water level function, the storage capacity function and the surface area. Function values are determined by linear interpolation of published data in the case of the Great Brak Estuary (CSIR 1990) and survey data in the case of the Kromme Estuary (Mr L van der Merwe *pers. comm.*), as listed in Table 4.3c. The surface area of these estuaries (Table 4.3d) is assumed constant for the purposes of calculating the evaporation from the estuary and the direct precipitation to the estuary. This is justifiable as the area of tidal flats within the usual range of tidal variation of both estuaries is insufficient to cause substantial alterations in surface area with variation in water level.

A further endogenous function requiring specification is the velocity function, which promotes asymmetry between the flood and ebb tides. This is defined in terms of an analytical form of the table functions of classical systems dynamics (Forrester 1961, 1969) due to Uys (1985), which has the properties depicted in Figure 4.3c. The parameters of this sigmoidally increasing function

Table 4.3c Data values of the water level and storage capacity functions for the Great Brak and Kromme Estuaries. Intermediate function values are derived from the known data values by linear interpolation.

GREAT BRAK ESTUARY			KROMME ESTUARY		
WLF(x_1)	SCF(WL)		WLF(x_1)	SCF(WL)	
x_1 (m^3)	WL (m + MSL)	SCF (10^{11} $m^3.yr^{-1}$)	x_1 (m^3)	WL (m + MSL)	SCF (10^{12} $m^3.yr^{-1}$)
1.0	-1.0	0.00004	1.0	-5.0	0.256
5.0	-0.2	0.529	70014.5	-4.0	0.604
15000.0	0.0	0.582	331129.5	-3.0	1.845
33000.0	0.2	1.361	1081057.0	-2.0	3.991
92200.0	0.4	2.393	2517845.0	-1.0	6.283
245300.0	0.8	3.188	3520684.0	-0.5	7.254
454000.0	1.2	4.062	4123523.0	0.0	8.225
706200.0	1.6	4.863	5299139.5	0.5	9.149
1005800.0	2.0	5.713	6474756.0	1.0	10.073
1354500.0	2.4	5.713	9183208.0	2.0	11.017

are subject to the restrictions that the maximum function value, A, must exceed unity, while the minimum function value, B1, must lie between zero and one. In the case of the velocity function (Figure 4.3c), the independent variable is defined as the ratio of the actual head gradient to a usual or average head gradient value, known as the head gradient normal (HGN) and the function is determined as:

$$
\begin{aligned}
VF(v) &= T(A1,B1,G1,v) \quad \text{if } v \geq 0 \\
&= T(A2,B1,G2,-v) \quad \text{otherwise} \\
T(A,B1,G,v) &= A / [1 + (A/B1 - 1).\exp(-c.v^q)] \\
c &= \log_e[(A/B1 - 1) / (A - 1)] \\
q &= G.A / [(A - 1).c] \\
v &= (TWL - WL) / L . HGN \quad \text{if } |x_{14}| = 0 \\
&= 0 \text{ otherwise}
\end{aligned}
$$

where VF = velocity function (unitless)
v = independent variable (unitless)
T = analytical table function (unitless)
A1 = maximum flood velocity function value (unitless)
B1 = minimum velocity function value (unitless)
G1 = flood velocity function gradient (unitless)

A2	= maximum ebb velocity function value (unitless)
G2	= ebb velocity function gradient (unitless)
A	= maximum table function value (unitless)
G	= table function gradient at v=1 (unitless)
c,q	= intermediate calculated values (unitless)
TWL	= tidal water level (m)
WL	= water level (m)
L	= length (m)
HGN	= head gradient normal (unitless)

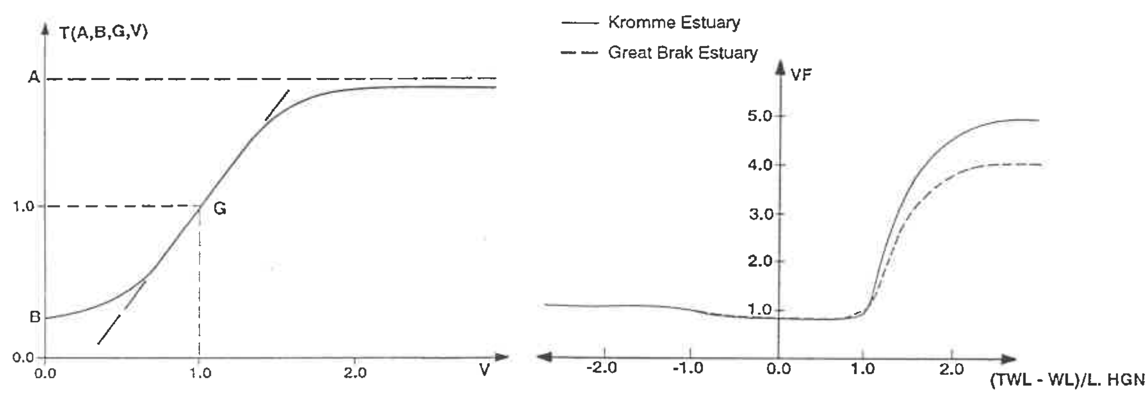

Figure 4.3c Diagrammatic representation of the asymmetrical velocity function, VF, defined in terms of two analytical table functions of the form T(A,B,G,V)

Thus, six additional parameters are required to specify the velocity asymmetry on the flood and ebb tides. These and other endogenous parameter values yielding adequate calibrations of the water volume sector of the estuarine systems model are listed in Table 4.3d. The following aspects of the parameter values and functions are noteworthy:

- during initial calibration of the water volume sector, the sill height is set at an initial value and remains fixed throughout the simulation,
- the effects of density-induced circulation on the tidal flux are ignored in the initial calibration of the water volume sector and the gravitational circulation function is unity,
- the Kromme Estuary is substantially deeper than the Great Brak Estuary and its storage capacity at equivalent depths far exceeds that of the Great Brak Estuary.
- the Kromme Estuary is approximately double the length of the Great Brak Estuary, therefore the characteristic estuary length must be greater. Additionally, the lower reaches of the Kromme Estuary are very shallow with channels meandering through extensive sand banks. Thus the characteristic length over which the hydraulic head acts for the Kromme Estuary is large, being set at 1 km, whereas that of the Great Brak Estuary is only 250 m,

Table 4.3d Endogenous parameter values for the Great Brak and Kromme Estuaries and some initial values of state variables used in the model calibration.

PARAMETER	SYMBOL	UNITS	GREAT BRAK ESTUARY	KROMME ESTUARY
Surface area	SA	m^2	470500.0	2756342.0
Length	L	m	250	1000
Width of the mouth	w	m	18	72,5
Head gradient normal	HGN	-	0.002	0.0004
Maximum flood velocity function value	A1	-	4.0	5.0
Minimum velocity function value	B1	-	0.85	0.85
Flood velocity function gradient	G1	-	1.0	2.5
Maximum ebb velocity function value	A2	-	1.05	1.05
Ebb velocity function gradient	G2	-	0.5	0.2
Initial value of sill height	$x_7(0)$	m	0.6	0.09
Time delay constant 3	t_3	yr	0.009855405	0.009855405
Initial value of tidal flushing	$x_6(0)$	yr^{-1}	415	222
Time delay constant 2	t_2	yr	0.015	0.03
Fresh water flow rate normal	x_{11n}	$m^3.yr^{-1}$	$4,5 \times 10^6$	1.4×10^7
Salinity export normal	x_{22n}	-	1.2	1.15
Characteristic estuary width	B	m	55	120
Time delay constant 1	t_1	yr	0.004223745	0.01126332
Grain diameter	DIAM	m	0.45×10^{-3}	0.25×10^{-3}
Porosity	Ps	-	0.4	0.4
Beach slope factor	BSF	-	1.0	1.0
Length of the estuary mouth	l	m	10	80

- in similar fashion, the width of the mouth at the Kromme Estuary is 72,5 m, whereas that at the Great Brak is 18 m, and
- the velocity function, which promotes asymmetry between flood and ebb tides, is formulated as a combination of two analytical table functions, but continuity is maintained as they are assigned the same minimum function value, B1.

Calibration Results

For the Great Brak Estuary, the calibration of the water level variations and tidal fluxes in response to tidal variation in the sea and fresh water inflows was undertaken using data recorded over the spring tidal period of 23 and 24 November 1988 (CSIR 1990a). These data were also used in the calibration of a one dimensional hydrodynamic model implemented on the Great Brak Estuary. The only inflow to the estuary at the time was a constant base flow of 0,1 $m^3.s^{-1}$. Therefore, the components of the inflow function were assigned the following values: BN was set equivalent to 0,1 $m^3.s^{-1}$, BSM was set at unity and no floods occurred. The sea tidal variation, measured at Mossel Bay Harbour, attained a minimum level of -0,58 m to MSL on 23 November and a maximum level of 1,3 m to MSL on 24 November. The difference in amplitude of the sea tide compared with the average variation for this coastal area as predicted in the Tide Tables (SAN 1994) was accomodated in the model by setting the meteorological function to $1.2\cos(2\pi t/0.019178082)$ for a period of only seven days and to unity otherwise and so ensuring an enhanced spring tidal amplitude. The resulting water level variations are presented in Figure 4.3d together with the measured variation in water level in the lower estuary on 23 and 24 November 1988. Characteristic features of the hydrodynamics of the Great Brak Estuary include:

- the strong reduction in tidal amplitude within the estuary (approximately 15% of that of the sea (CSIR 1990a)),
- minimum water levels within the estuary of greater than 0,55 m to MSL during mouth open phases (CSIR 1994b). These minima sometimes increase to 0,8 m to MSL and more immediately prior to mouth closure,
- increased low water levels within the estuary at spring tide compared with neap tide. This phenomenon, termed super-elevation, has been observed in other South African systems with constricted mouths eg. the Mfolozi Estuary (CSIR 1990d), and
- flood tides of 4,5 hour duration and ebb tides of 7,8 hour duration, with high water in the estuary lagging that in the sea by about 30 minutes.

Despite the occurrence of an unusually high and asymmetrical tide in the sea on 23, 24 November 1988, these characteristic features are reflected by the estuarine systems model. The amplitude of variation of the simulated water level in the estuary is about 15% of the sea tide. The simulated high waters in the estuary basin lag those measured in the lower estuary (the recorded water levels in Fig 4.3d), as expected. The simulated low water levels correspond well with the measured low water levels and the duration of flood and ebb tides agree. In addition, although tidal fluxes were not measured at the time that the water levels were recorded, these were simulated by a one dimensional hydrodynamic model. The estuarine systems model was calibrated to yield tidal fluxes of corresponding magnitude with a maximum simulated flood tidal flux of approximately 12 $m^3.s^{-1}$ and a maximum simulated ebb tidal flux of less than 9 $m^3.s^{-1}$. Other characteristic features of the Great Brak Estuary are evident in Figure 4.3d, where a slight increase in low water levels over spring tide as opposed to neap tide is discernible and the inability of the tide to effectively penetrate the estuary over neap tides when the sill at the mouth is 0,6 m to MSL is clear. Note that the sill height was set at this level during initial calibration of the water volume sector, but full calibration of the interaction of water flow and the transport, erosion and deposition of sand will be undertaken during calibration of the sediment sector (Section 4.3.5).

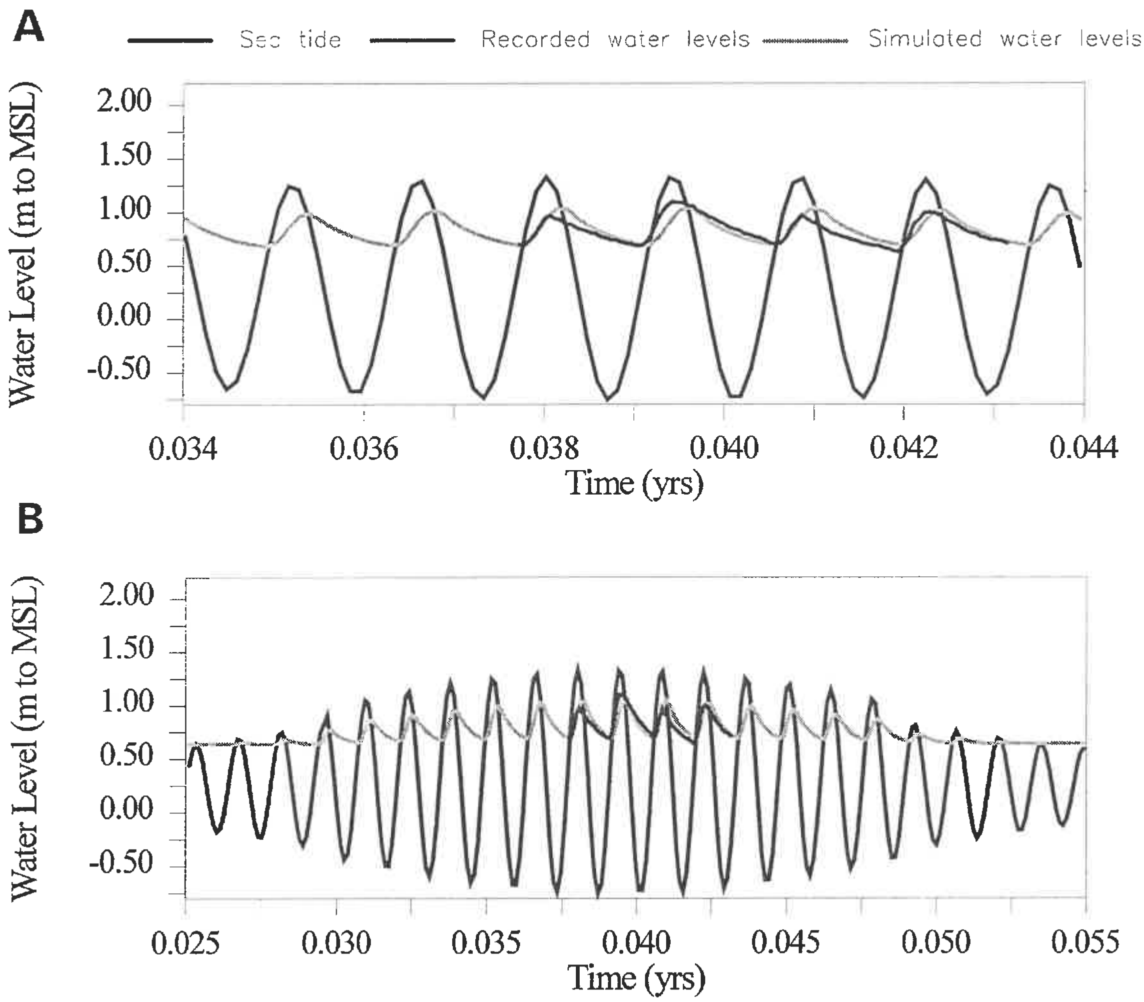

Figure 4.3d Comparison of simulated and recorded water level variations in the Great Brak Estuary for periods of approximately 4 (A) and 11 (B) days over a spring tide analogous to that of 23 and 24 November 1988. The simulation commenced at 08:00 on 9 November.

For the Kromme Estuary, the calibration of the water volume sector was undertaken using data collected from 12 to 24 August 1988 (Slinger 1989, CSIR 1994c). Water level variations were recorded at five positions in the estuary over this period. A spring high tide level of 1,01 m to MSL was predicted for 17:00 on 14 August 1988 with a low tide level of -0,61 m to MSL occurring at about 10:30 in the morning. The tide function was assigned parameter values as indicated in Table 4.3a, although the diurnal modulation in high tidal levels which occurred in the sea during the calibration period is not then reflected. The riverine flow to the estuary at the time was a constant base flow of 0,1 $m^3.s^{-1}$. Therefore, the components of the inflow function were assigned the following values: BN was set equivalent to 0,1 $m^3.s^{-1}$, BSM was set at unity and no floods occurred. The simulated estuarine water levels and the sea tide are depicted in Figure 4.3e together with the recorded water levels within the mouth of the estuary and 10 km upstream of the mouth for 3,65 days from 17:00 on 12 August. The simulation commenced on 1 August. The following observations can be made regarding the measured and simulated water levels:

- water levels within the estuary varied from 0,2 m to MSL to 0,9 m to MSL over spring tide and from about 0,05 m to MSL to 0,4 m to MSL over neap tide,
- low water levels over spring tide exceeded those measured over neap tide by about 0,15 m, indicating that super-elevation at spring tides is a feature of the Kromme Estuary,
- the simulated water level variation in the estuary has an amplitude about 43% of that of the sea tide, while the amplitudes of variation at the mouth and 10 km upstream are approximately 53% and 39% of the sea tide, respectively,

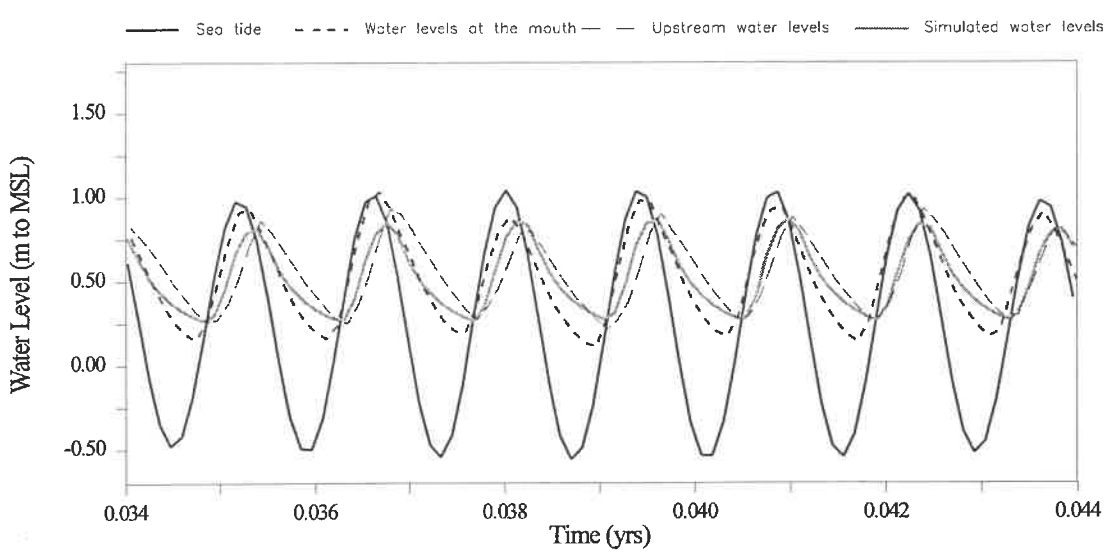

Figure 4.3e Comparison of simulated and recorded water level variations at two positions in the Kromme Estuary (at the mouth and 10 km upstream) for a period of approximately 4 days over a spring tide analogous to that of 12 to 15 August 1988

- simulated low water levels in the estuary correspond closely with the low water levels measured 10 km upstream of the mouth, indicating that the effect of the mouth in retarding draining of the estuary is well reflected,
- the timing of simulated high water levels within the estuary basin is intermediate to the occurrence of high water at the mouth and 10 km upstream in the estuary,
- the duration of the modelled flood tide is approximately 4 hours and that of the ebb tide 8,3 hours, which agrees well with field observations, and
- the slight difference in high water levels on alternate days which is evident in the recordings is a reflection of the diurnal modulation of the sea tide. This feature is not reflected in the simulated water levels as it was omitted in the parameterization of the relevant exogenous forcing function, the tide function.

Thus, the estuarine systems model produces water level variations which accord well with those recorded in the Kromme Estuary. In addition the magnitude of the tidal fluxes agrees with those computed by a one-dimensional hydrodynamic model (CSIR 1994c) as depicted in Fig 4.3f. The maximum simulated ebb tidal flux is 133 $m^3.s^{-1}$ and the maximum flood tidal flux is about 175 $m^3.s^{-1}$.

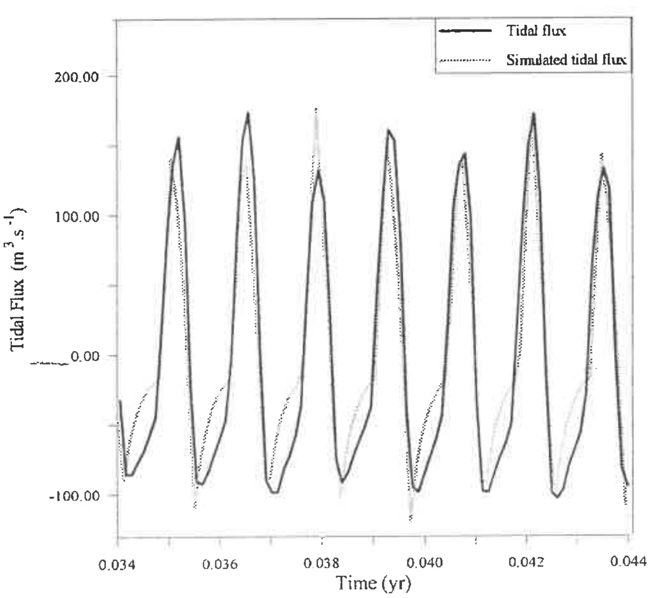

Figure 4.3f Comparison of the tidal flux simulated for the Kromme Estuary by the estuarine systems model (light line) and that computed by a one-dimensional hydrodynamic model (dark line)

4.3.2 Calibration of the flushing sector

Tidal Flushing

An important time scale in the tidal flushing of the small, bar-built estuaries of South Africa is the spring-neap time scale. For instance, under low fresh water flows in the Palmiet Estuary the spring tidal intrusion of sea water along the base of the estuary is of sufficient magnitude to renew the old sea water trapped in deep scour holes in the upper estuary, whereas the extent of sea water intrusion is more limited on neap tides (Largier 1986, Slinger & Largier 1990). Similarly, the enhanced intrusion of sea water during spring tides as opposed to neap tides plays a role in the flushing of the water of the middle reaches of the Great Brak Estuary. Renewal of old water in the deep scour holes of the lower middle reaches may be effected on a spring tide should the mouth be open, whereas neap tides are ineffectual in flushing any but the lower reaches of the estuary (Taljaard & Slinger 1993). Therefore, on the one hand, the time delay constant 3 (t_3) has to be selected so that the variation in tidal flushing in the estuary over the spring neap tidal cycle is discernible. On the other hand, the time delay constant 3 must be of sufficient magnitude to aggregate the semi-diurnal tidal variations. Accordingly t_3 was set at seven times the semi-diurnal tidal period for both the Great Brak and the Kromme Estuaries (Table 4.3d). In calibrating the tidal flushing variable, then, it remains to select the initial value, $x_6(0)$, so that a state of dynamic equilibrium is achieved fairly rapidly. For the Great Brak Estuary, this value is set as 415 yr^{-1} and a repetitive cycle of approximately 28 day period, which represents a state of dynamic equilibrium in the tidal flushing in the absence of seasonality in the freshwater inflow or wave conditions, is achieved within four weeks of the start of the simulation. For the Kromme Estuary, $x_6(0)$ is set as 222 yr^{-1}. The value for the Great Brak Estuary is understandably greater than for the Kromme Estuary (despite its more constricted mouth), because under similar freshwater inflow conditions, a higher proportion of the total volume of the smaller Great Brak Estuary is exchanged by the tide.

Fresh Water Flushing

The efficacy of fresh water flushing of an estuary is dependent primarily on the maximum flow rate and duration of a fresh water flow event. The higher the fresh water flow rate and the more sustained the flow, the more effective the flushing of the water body. Therefore, the time delay constant 2 (t_2) must be of sufficient magnitude to capture the lingering effects of a high volume, fast flowing flood, yet be small enough to reflect the difference between such a flood and an inflow of similar volume, but longer duration. Accordingly, for the larger Kromme Estuary the time delay constant 2 was chosen as 0,03 yr (that is, 11 days) and the value selected for the smaller Great Brak Estuary was 0,015 yr or 5,5 days (Table 4.3d). A further parameter of concern in calibrating the fresh water flushing variable is the fresh water inflow rate normal (x_{11n}). For the Kromme and Great Brak Estuaries, this parameter was set at approximately 2,5% of the maximum flow rates of their annual floods, with that of the Kromme Estuary being 1,4 x 10^7 $m^3.yr^{-1}$, while for the Great Brak Estuary this parameter is 4,5 x 10^6 $m^3.yr^{-1}$. The initial values of the freshwater flushing variables, $x_5(0)$, were then assigned as 0,1 and 1,8 for the Kromme and Great Brak Estuaries, respectively.

4.3.3 **Calibration of the salt sector**

The salinity structure measured on 30 November 1988 represents an open mouth situation accompanied by low fresh water inflows (CSIR 1990a), a typical condition for the small Great

Brak Estuary. Freshwater influence was limited to the head reaches of the estuary and the system was predominantly saline with slight vertical stratification (Figure 4.3g). Active tidal exchange occurred in the lower reaches of the estuary, while little renewal or exchange of water occurred in the middle reaches. This typical situation was used in the calibration of the salt sector of the Great Brak model as no time series of concurrent salinity and inflow data under steady open mouth conditions was available and data from a closed mouth situation could not be used in the calibration of the salt import and export rates. The following values were assigned:

- a base flow incorporating seasonality and averaging 0,2 $m^3.s^{-1}$ was considered representative of the fresh water inflow at the time,
- no floods or high waves were included as these would naturally cause an alteration in the mouth configuration and this sector is not yet calibrated,
- the rainfall, evaporation and groundwater influences were determined as described in Section 4.3.1,
- the exogenous tide function was taken as a simple cosine series incorporating the semi-diurnal and spring-neap features characteristic of the Mossel Bay area (Table 4.3a),
- the initial condition for mean salinity was set at 29 $kg.m^{-3}$ to accord with observed conditions (Figure 4.3g), and
- a value of 34,5 $kg.m^{-3}$ (sea salinity) was assigned to the reduced salinity function and a value of unity was assigned to the stratification export function, because the stratification-circulation sector is not yet calibrated.

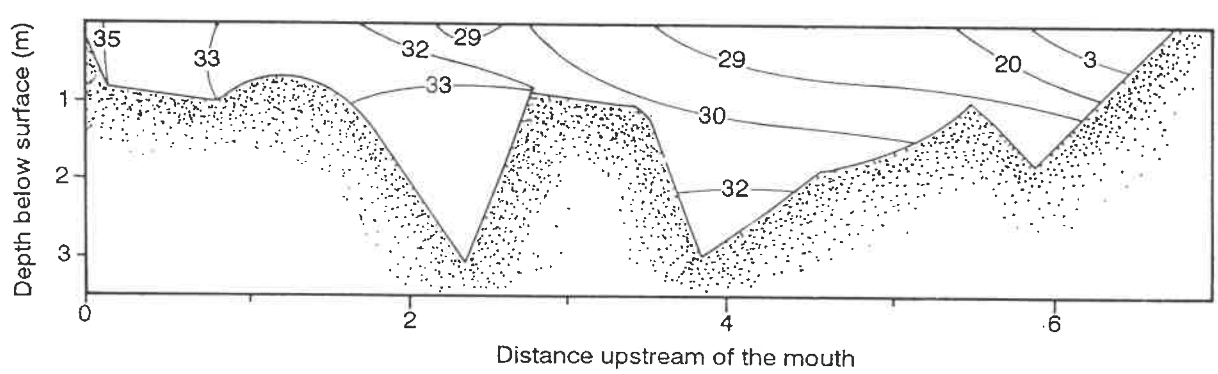

Figure 4.3g The salinities ($kg.m^{-3}$) measured on the spring flood tide of 30 November 1988 are considered typical of the Great Brak Estuary under low inflow and open mouth conditions. The river is to the right and the sea is to the left of the mouth.

Thereafter, the form of the salinity export multiplier was selected as the product of a salinity export normal (x_{22n}) and a two dimensional function similar in form to the table functions of classical system dynamics (Forrester 1961, 1969). After assigning values to this function, preliminary simulations were conducted and the simulated mean salinities were compared with the typical salinities encountered in the Great Brak Estuary under low freshwater flows and open mouth conditions. The latter step was repeated with adjustments in the SEM(x_5,x_6) function values until reasonable agreement was attained between simulated and typical mean salinities. The value of the salinity export normal for which the agreement was closest is listed in Table 4.3d, while the salinity export multiplier is depicted in Figure 4.3h. The simulated mean salinities for the Great Brak Estuary under open mouth conditions are presented in Figure 4.3i.

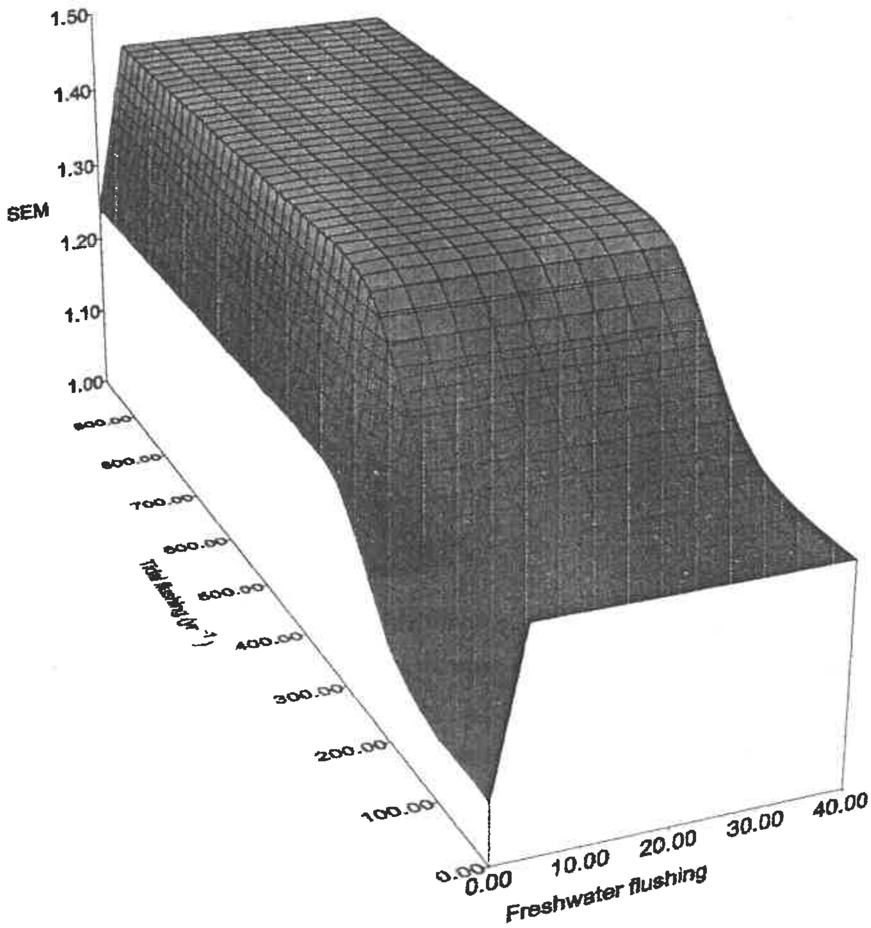

Figure 4.3h The salinity export multiplier, a two-dimensional table function of both the fresh water and tidal flushing variables

The simulated mean salinities range from 21 to about 30 kg.m^{-3} in the estuary when the fresh water inflow is lowest, that is during April, May and June, whereas mean salinities range from between 10 and 12 kg.m^{-3} to about 28 kg.m^{-3} during November and September when higher fresh water inflows occur. These salinity ranges are considered representative of the behaviour of the Great Brak Estuary under steady open mouth conditions and relatively low inflows (18,5% of the natural mean annual run-off). The fact that the average salinities in the estuary are strongly affected by the spring-neap tidal cycle with enhanced salinities on spring tides and lower salinities on neap tides also concurs with observed behaviour.

The axial salinity function for the Great Brak Estuary was formulated to yield head to mouth differences in salinity ranging from 2 to 12,5 kg.m^{-3} in the driest months of April, May and June, but varying between about 15 and 35 kg.m^{-3} during the wettest month, November (Fig. 4.3i). No propensity for hypersalinities is indicated for the Great Brak Estuary. These features accord with observations of the estuary undertaken intermittently over a six year period (CSIR 1990a, 1992a, 1993d, 1994b) and the salt sector of the model is considered calibrated for the Great Brak Estuary under steady mouth and stratification-circulation conditions.

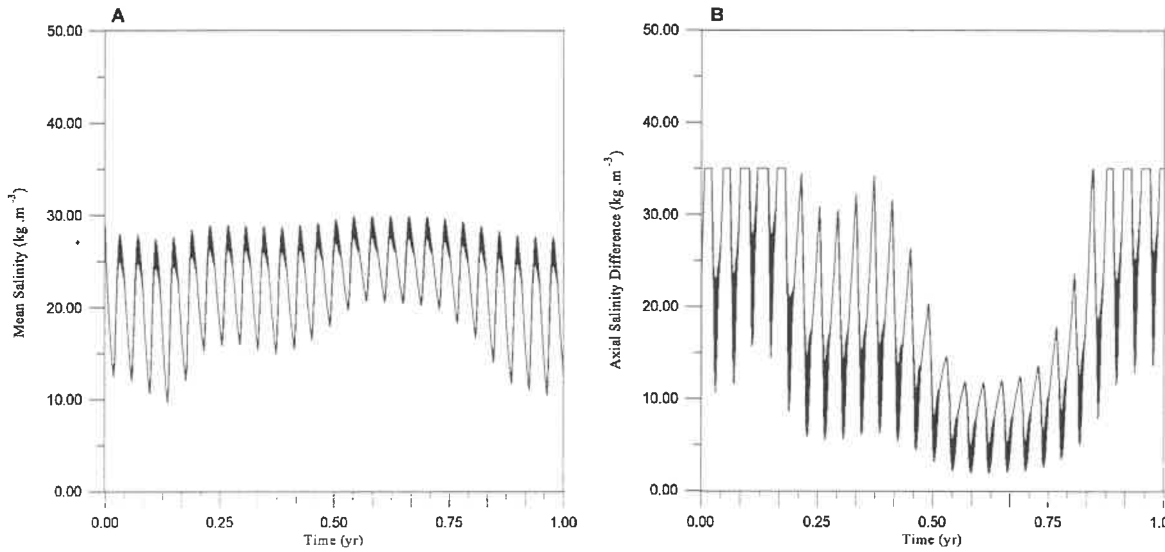

Figure 4.3i Simulated mean salinities (A) and axial salinity differences (B) in the Great Brak Estuary over one hydrological year under open mouth conditions and low flows.

It is noteworthy that the lack of appropriate calibration data for the salt sector (concurrent salinity and inflow data under steady open mouth conditions) means that there is some uncertainty regarding the parameter values of the salinity export multiplier for the Great Brak Estuary. The parametrisation of the axial salinity difference function of two model variables, the salt content and the freshwater flushing of the estuary, is also subject to uncertainty. The effects of these uncertainties on the Great Brak model results will be investigated at a later stage.

Concurrent inflow and salinity data are available for the Kromme Estuary for the period 1 October 1993 to 2 February 1994 (CSIR 1994c). Salinities were measured along the length of the estuary at spring tide on 16 October 1993, at neap tides on 21 November 1993, 21 December 1993, 2 February 1994 and 4 March 1994. Although data on the freshwater inflow to the system are not available for the period from 2 February to 4 March 1994, a low flow rate of 0,003 m^3.s^{-1} is considered reflective of conditions at this time (Mr Govani *pers. comm.*). Accordingly, calibration of the salt sector of the model was undertaken by:

- setting the inflow function values equal to the known freshwater inflows from 1 October 1993 to 4 March 1994,

- assigning rainfall, evaporation and groundwater influences as described in Section 4.3.1,
- setting the exogenous tide function (the downstream open boundary condition) to a simple cosine series incorporating the semi-diurnal and spring-neap features characteristic of the Port Elizabeth region (Table 4.3a),
- selecting an initial condition of mean salinity 20 kg.m^{-3} for the calibration run,
- setting the reduced salinity to the ocean salinity of 34,5 kg.m^{-3} and the stratification export function to unity (as the stratification-circulation sector is not yet calibrated),
- assigning appropriate values to the salinity export multiplier SEM(x_5,x_6),

and then comparing the simulated mean salinities with the average measured values. The latter step was repeated with adjustments in the SEM(x_5,x_6) function values until good agreement was attained between simulated and measured mean salinities. The value assigned to the salinity export normal is listed in Table 4.3d, while the salinity export multiplier for the Kromme Estuary is depicted in Figure 4.3h. The freshwater flow rate over the calibration period and the comparison between the simulated mean salinities and the average measured salinities are presented in Figure 4.3j.

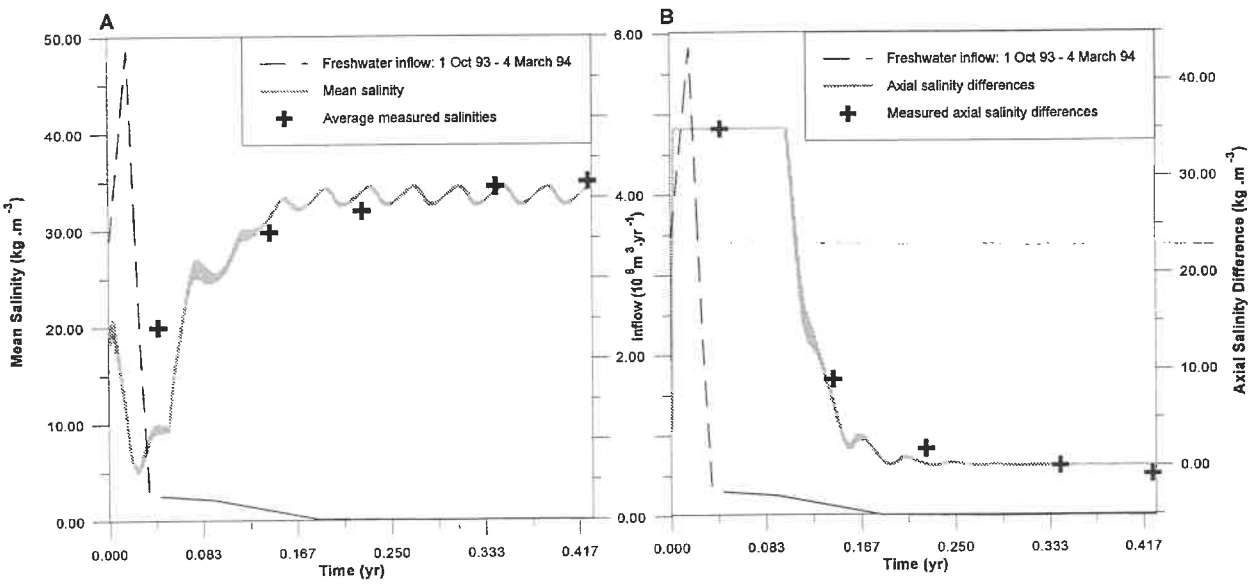

Figure 4.3j Comparison between the simulated mean salinities (A) and axial salinity differences (B) in the Kromme Estuary and those calculated from salinity measurements on five occasions over the period 1 October to 4 March.

Only on 16 October 1993 do the simulated mean salinity and the average measured value differ by more than five percent. This is understandable as sampling didn't occur over neap tide on this occasion, but rather over spring tide when large variations in salinity with the tidal cycle are common. As the time of sampling at each hydrographic station position was not recorded, these effects could not be corrected. Thus, less emphasis was placed on obtaining agreement between the simulated and measured salinities on 16 October 1993 than on the other sampling occasions.

The axial salinity function of mean salinity and fresh water flushing was determined as for the Great Brak Estuary and yielded values which correspond fairly well with the head to mouth salinity differences recorded in the Kromme Estuary over the period from 16 October 93 to 4 March 94 (Figure 4.3j). The slight hypersalinity recorded in the estuary by March 1994 is not reflected in the simulated axial salinity differences. However, the fact that hypersalinities are likely to occur may be deduced from the persistence of an axial salinity gradient of zero.

4.3.4 Calibration of the stratification-circulation sector

The most important aspect of the calibration of the stratification-circulation sector is the formulation of the density anomaly function. Both the stratification and the circulation indices depend upon this function, which is dependent in turn upon the value of the mean salinity. The form of the function, depicted in Figure 4.3k, is derived from empirical knowledge of the relationship between the average salinity conditions in the small, narrow estuaries common to southern Africa and the associated horizontal density differences (Largier 1986, Slinger 1989, MacKay & Schumann 1990, Largier & Taljaard 1991, CSIR 1993b, CSIR 1994c, Slinger *et al.* 1994). Although the information base for the function derivation covers highly stratified to well mixed estuaries (the Palmiet and Kromme Estuaries, respectively) as well as large, permanently open and small, intermittently closed systems (the Great Berg and Great Brak Estuaries, respectively), amongst others, considerable uncertainty remains regarding the representative nature of this function for all South African estuarine systems. The influence which uncertainties in the function values exert on model results will be investigated at a later stage in order to determine the necessity, or otherwise, of improving the determination of this function.

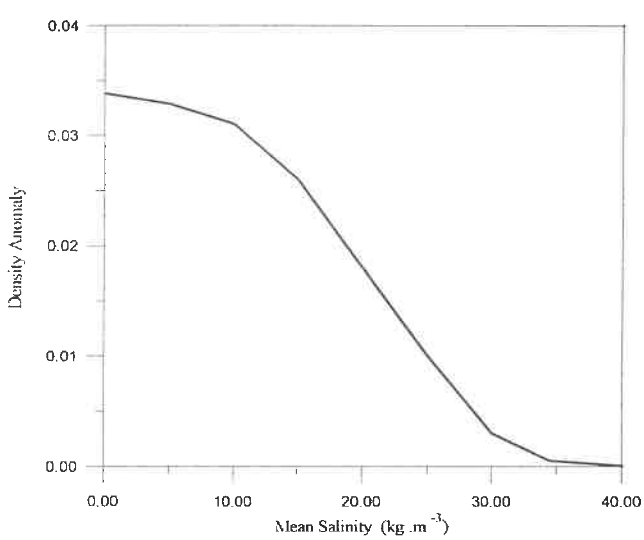

Figure 4.3k The form of the density anomaly function of mean salinity

Site-specific parameters requiring specification for the full calibration of this sector are the average width of the estuary, B, and the time delay constant 1, t_1. The former parameter is obtained from bathymetric surveys of the estuary or from maps of the physiography of the estuarine system where accurate surveys are not available. Thus the width of the larger, longer Kromme Estuary is set at 120 m, the estimated average width of the channel at 0,09 m to MSL, and the value for the Great Brak Estuary is set at 55 m, the average width at 0,6 m to MSL (CSIR 1990a, Mr L van der Merwe *pers. comm.*). The latter parameter, t_1, is selected so that variations in stratification and circulation with freshwater flow events and the spring-neap tidal cycle may be detected. In the larger Kromme Estuary the time constant is set at eight times the semi-diurnal tidal period, whereas in the smaller Great Brak system the time constant is set at three times the semi-diurnal tidal period (Table 4.3d).

The authenticity of these choices of parameter values for the Great Brak Estuary was established by simulating a freshette of similar magnitude and duration to the flood release of 29 and 30 November 1990. On this occasion, $3,8 \times 10^5$ m^3 was released from the Wolwedans Dam over a thirteen hour period, causing the water level in the estuary to increase from 1,09 m to MSL to 1,61 m to MSL (CSIR 1992a). Twenty hours after the mechanical breaching of the mouth, the water level had decreased to 0,82 m to MSL and seven days later a minimum water level of 0,58 m to MSL was attained. As the sediment sector of the Estuarine Systems Model is not yet calibrated for the Great Brak Estuary, the scour at the mouth was not simulated at this stage. Instead, a constant sill height of 0,6 m to MSL and an aseasonal base flow of 0,1 $m^3.s^{-1}$ were assumed, a freshette of $3,8 \times 10^5$ m^3 was discharged to the estuary over 13 hours on 29 and 30 November and the effects on the water levels, mean salinities and the stratification and circulation indices were simulated. The simulated maximum efflux was 15,5 $m^3.s^{-1}$, which agrees fairly well with the maximum outflow rate of 16,6 $m^3.s^{-1}$ calculated from hourly observations of the November 1990 flood release (CSIR 1992a). The simulated mean salinities varied between 22 and 32 $kg.m^{-3}$ prior to the release (reflective of 0,1 $m^3.s^{-1}$ fresh water inflow, open mouth conditions), but decreased sharply as the fresh water released from the impoundment reached the estuary exhibiting a minimum salinity of about 6 $kg.m^{-3}$ (Figure 4.31). Within one and a half

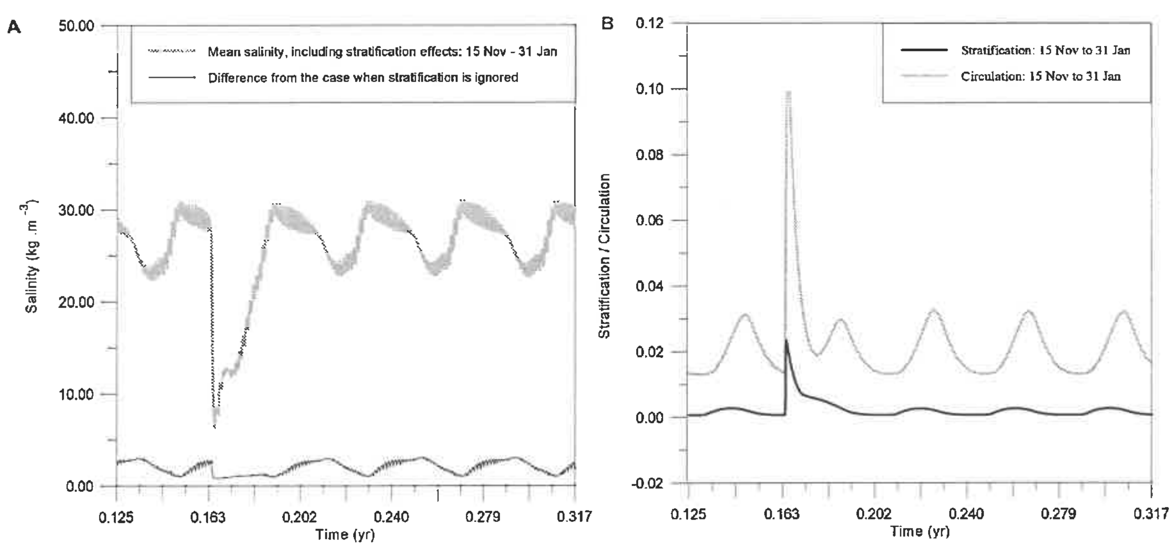

Figure 4.31 Differences in the simulated mean salinities (A) when the stratification-circulation effects (B) associated with a freshette to the Great Brak Estuary are considered.

weeks of the release, the average salinities in the estuary once again varied between 22 and 32 kg.m^{-3}, that is within the usual range of variation of salinities under minimal inflow, open mouth conditions. The predicted short-lived effect of the release on the water body accords well with observations of the water column structure of the Great Brak Estuary (Taljaard & Slinger 1993).

The maximum values of the stratification and circulation indices occurred about a day and a half after the release commenced and the stratification-circulation state predicted was that of a partially mixed system (Figures 4.31 & 3.4a) with top to bottom salinity differences approximately 16% of the depth mean salinity. The delay in attaining a maximum is an artefact of the formulation of these indices as first order exponential delays of the analogue Estuarine Richardson Number and the Froude Number, respectively, but is reflective of the real situation in the Great Brak Estuary in that the buoyant freshwater spreads slowly over the more saline bottom layers of the estuary and the stratification state is maximal about one to two days after a freshwater influx. Within 10 days the stratification-circulation state returned to the less stratified, yet still partially mixed situation common to the Great Brak Estuary under minimal inflow, tidal conditions. In this situation, the top to bottom salinity differences are commonly less than 5% of the mean salinity. Despite the fact that the intense stratification in isolated deep holes in the Great Brak Estuary is not reflected by the model, the predictions of the stratification-circulation state of the water body are considered generally reflective of the real situation of a freshwater flow event because the deep holes contribute less than 9% of the volume of the Great Brak Estuary at 0,6 m to MSL (Table 4.3c).

The reduced salinity function of the stratification and circulation indices was formulated so as to yield values between 35 and 34,5 kg.m^{-3} when the estuary is well mixed (stratification index > 0.03) gradually decreasing to values of about 14 kg.m^{-3} when the estuary is strongly stratified (stratification index > 0.5). The stratification export function of the stratification and circulation indices, on the other hand, is formulated as a two-dimensional table function with a maximum value of 1,008 under vertically mixed conditions and a minimum value of 0,92 when the estuary is highly stratified. The effects of these functions of stratification and circulation on the mean salinity of the Great Brak Estuary are depicted in Figure 4.31.

The greatest influence of the stratification-circulation state on the mean salinities occurred during the freshette and at neap tides. This is to be expected, because stratification in the estuary is more intense at times of enhanced freshwater input (the freshette) or reduced tidal influence (neap tides). The deviation in mean salinities owing to consideration of the stratification-circulation state is less than 20% at the time of the release and less than 12% at neap tides.

Data on inflows and salinities in the Kromme Estuary from 1 October 93 to 4 March 94 were used in the calibration of the stratification-circulation sector of the Kromme model (CSIR 1994c). During this period, the stratification index for the Kromme Estuary exceeds 0,08 only for about 15 days (Figure 4.3m), indicating that top to bottom differences in salinity in excess of 1 kg.m^{-3} existed in the water column when the mean salinity was less than 10 kg.m^{-3} (Figures 3.4a & 4.3j). However, this state was temporary and as the freshwater inflow to the system decreased, the stratification index declined to its customary value of well below 0,08, indicating that the water column of the estuary was again vertically mixed. This accords with field observations from August 1988, when little freshwater inflow occurred and a maximum salinity difference of 1 kg.m^{-3} was measured in the water column on the late spring ebb tide when the mean salinity was 33 kg.m^{-3} (Slinger 1989). This implies that the Estuarine Richardson Number was 5 x 10^{-2} and less throughout the field exercise, which occurred under the low flow conditions typical of the Kromme Estuary. Thus, despite limited data, the stratification index yields results which correspond with the current understanding of the present stratification state of the Kromme Estuary.

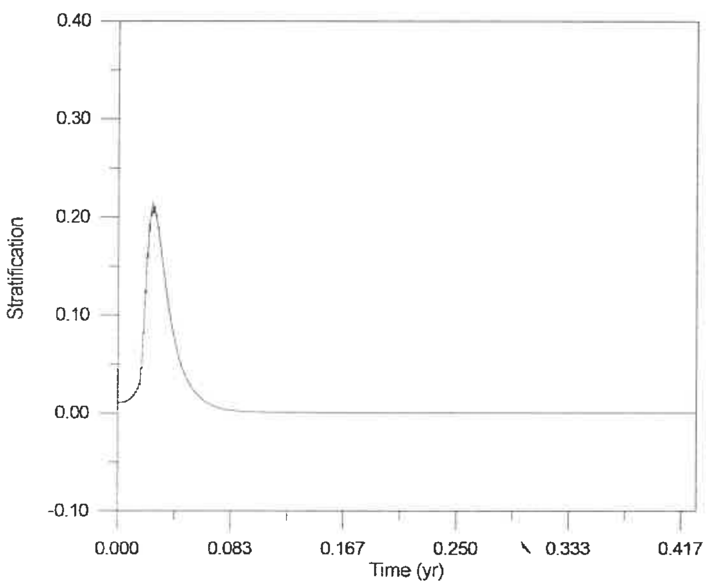

Figure 4.3m The response of the stratification index to the freshwater flows which entered the Kromme Estuary over the time period 1 October 1993 to 4 March 1994.

In the formulation of the circulation parameter, freshwater inflow to an estuary is assumed to occur continuously (Hansen & Rattray 1965, 1966) and the rating of circulation in terms of freshwater flow is incorporated in the circulation index derived by the model. In the case of the Kromme Estuary where inflows may be zero for much of the year (the present run-off situation), the circulation index is set at 1 at these times and the characterisation of the estuary depends largely on the stratification. Clearly, the estuary is predominantly a vertically mixed system.

In contrast to the Great Brak Estuary, the characteristically well mixed state of the Kromme Estuary means that the reduced salinity function, RS(x_3,x_4), assumes the value of the ocean salinity the majority of the time. In similar vein, the stratification export function, generally maintains a value of unity indicating that there is no reduction in the export of salt when a system is vertically mixed.

4.3.5 **Calibration of the sediment sector**

The release of 3,8 x 10^5 m^3 of water to the Great Brak Estuary on 29 and 30 November 1990 and the accompanying scour of marine sediment from the mouth of the estuary was used in the calibration of the sediment sector (CSIR 1992a). The release of 2,2 x 10^5 m^3 from 08:00 to 15:30 on the 29 November, followed by a further release of 1,6 x 10^5 m^3 on 30 November 1990, was simulated using two flood functions of the requisite total volumes. The tide function was assigned a simple cosine form incorporating semi-diurnal and spring-neap features with spring high tide timed to occur three days after the mechanical breaching of the mouth. Breaching commenced two hours before spring high on 30 November and was incorporated in the model as a breaching rate of -1980 $m.yr^{-1}$ of four hour duration. With the grain diameter, DIAM, set to 0,45 mm, the median sediment grain size for the Great Brak Estuary, the Ackers-White formula for the prediction of the sediment volume transported by the outflowing water could be implemented (Section 3.6.1). The following points are relevant in calibrating the scour at the estuary mouth:

- The concept of a threshold for sediment movement is central to the Ackers-White formula for sediment transport. The threshold value is determined once a value is assigned to the grain diameter parameter, DIAM.
- The tidal flux is calibrated for steady open mouth conditions, so the width of the estuary mouth has already been assigned a value (Table 4.3d). Minor adjustments can be made in this parameter value to fine tune the calibration of the scour, but major adjustments would affect the calibration of the water volume sector and thus are not feasible.
- The parameter l, the length of the mouth channel, has to be assigned a value so that the volume of sediment scoured from the mouth area causes an appropriate reduction in the height of the sill.

In attempting to assign an appropriate and realistic value to l, it became evident that, according to the Ackers-White formula, the observed velocities would not result in the observed scour. While the Ackers-White formula has proved reliable in predicting volumetric sediment transport under high flow conditions (White *et al.* 1975, Nakayo 1990), the shear velocities arising from the flood and mechanical breaching of the Great Brak Estuary were indicated as insufficient to generate significant sediment movement, yet in reality extensive scour occurred (CSIR 1992a). Accordingly, slight modifications to the Ackers-White formula were made for the case of low flows, which were considered as flow rates of less than 2,5 $m.s^{-1}$ through the mouth (the situation in South African estuaries when fresh water flooding is not occurring). These encompassed reducing the threshold value for sediment movement by 25% and allowing the hydraulic head to act over 80% of the length of the mouth channel rather than over the whole area available for scour and deposition. Both these assumptions are considered reasonable and the simulation results which were then produced with a mouth channel length of 10 m bear this out.

The recorded alterations in water levels in the estuary caused by the flood release, the associated breaching of the mouth and the resultant scour of the sill are depicted in Figure 4.3n as is the simulated water level response. The simulation commenced on 9 November. Good agreement between the simulated and observed water levels was achieved. The simulated water level decreased from 1,63 to 0,94 m to MSL immediately after the breaching, whereas a decline from 1,61 to 1,01 m to MSL was observed in reality. A low water level of 0,82 m to MSL was attained twenty hours subsequent to the breaching in the real situation. However, this was transitory as the amount of sediment in suspension in the mouth area was considerable and deposition occurred, causing the sill height to increase slightly. These local effects are not simulated by the Estuarine Systems Model, but the scouring of the sill to the extent that low water levels of about 0,86 m to MSL occurred over the three subsequent low tides is well simulated by the model. This is highly satisfactory as it is low water levels rather than high water levels which are most indicative of the retarding effect of the sill on the drainage of water from an estuary. Additionally, the maximum simulated tidal efflux of 17,7 $m^3.s^{-1}$ occurred five hours after the mechanical breaching of the spit at the mouth, that is three hours after spring high tide. This agrees with the observed time of maximum outflow and is in good agreement with the outflow rate of 16,6 $m^3.s^{-1}$ calculated from hourly observations.

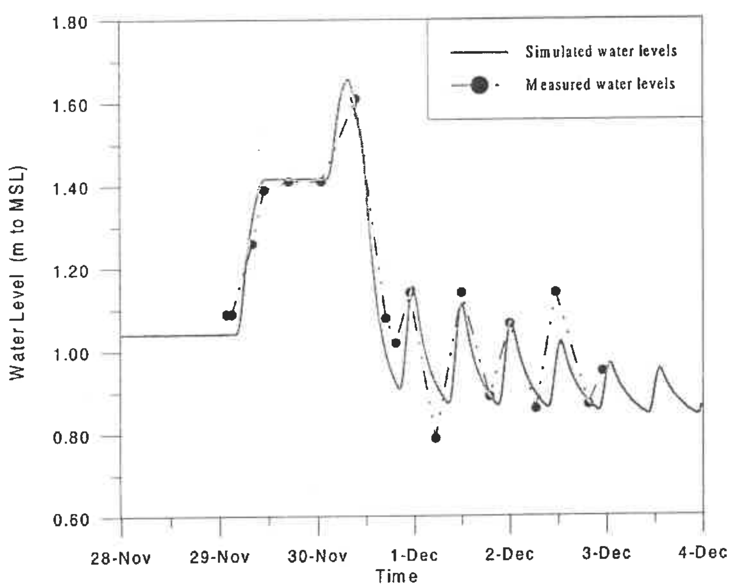

Figure 4.3n Comparison between the measured and simulated water levels (from 17:00 on 28 November) in the Great Brak Estuary following a flood release and associated breaching of the mouth on 29 and 30 November.

Estimates of the mean salinities in the system were derived by volume-weighted averaging of salinity measurements taken in the Great Brak Estuary on 28, 29, 30 November and 1, 4 December 1990 (Taljaard & Slinger 1993). The mean salinity in the estuary prior to the release was approximately 29 $kg.m^{-3}$. With the influx of fresh water to the closed estuary this decreased to about 12 kg.m^{-3} immediately prior to the mechanical breaching of the mouth. On the day following the release, low salinity surface water continued to flow from the estuary, but saline water was present in the bottom layers. The mean salinity was consequently estimated at between 15 and 18 kg.m^{-3}. On the flood tide, however, strong intrusion of saline water occurred and mean salinities increased to about 20 kg.m^{-3}. Three days later, the middle reaches of the estuary exhibited mean salinities of 25 kg.m^{-3}, that is, salinities typical of the middle reaches of the Great Brak Estuary under open mouth conditions. As depicted in Fig 4.3o, the simulated mean salinities accord reasonably with these estimates. The reduction in mean salinity associated with the influx of freshwater is evident as is the decrease in salt content from about 14 to 12 kg.m^{-3} associated with the outflow from the estuary. The strong influx of salt on the day following the release is well modelled, with mean salinities of 19 kg.m^{-3} occurring late on the flood tide of 1 December 1990. By 4 December, mean salinities in the estuary varied between 26 and 28 kg.m^{-3}. This is considered reasonably representative of reality, because the high salinity water of the more extensive lower reaches would counteract the lower mean salinities of the upper reaches and increase the mean salinity above the value of 25 kg.m^{-3} estimated for the middle reaches alone.

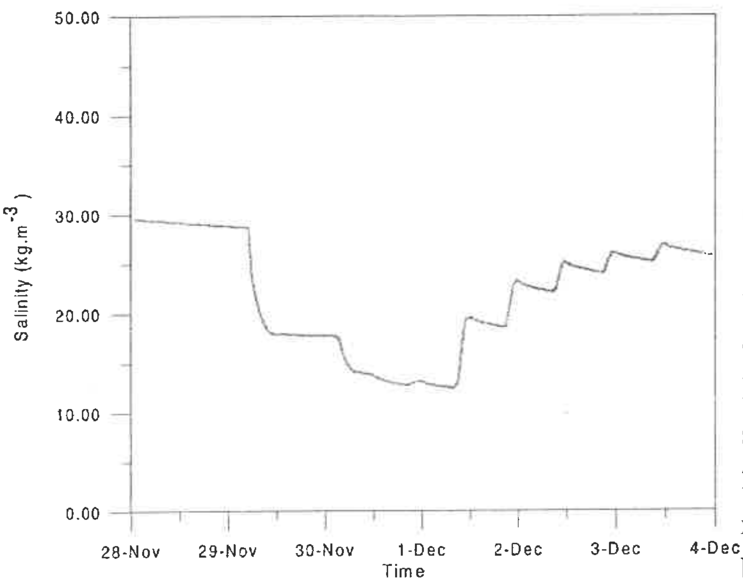

Figure 4.3o Simulated mean salinities in the Great Brak Estuary for a six day period from 17:00 on 28 November incorporating a flood release, mouth breaching and the subsequent intrusion of new sea water.

The stratification and circulation indices, which both attain maxima of the order of 0,06, similarly exhibit behaviour representative of the real situation. The vertical differences in salinity during the breaching operation are indicated as some 20 to 30% of the mean water column salinity. This is borne out by salinity measurements, particularly in the middle reaches of the estuary where salinity differences of 6 kg.m^{-3} and greater occurred over a water column with mean salinity 18 kg.m^{-3}. The estuary remained in a partially mixed state throughout the release.

Thus, with full interaction allowed between the sediment and other sectors of the model, the scouring of the estuary mouth owing to a fresh water release and the subsequent intrusion of saline water into the Great Brak Estuary is satisfactorily simulated. It remains to consider deposition of marine sediment in the mouth in response to high waves. For this purpose, a wave seasonal multiplier and a wave factor were included in the model formulation, as was the beach slope factor. In the case of the Great Brak Estuary, the beach slope factor was assigned a value of unity (Table 4.3d) as the zone of maximum sediment suspension owing to breaking waves is not in the vicinity of the mouth. Selected function values for the wave seasonal multiplier of the Great Brak Estuary are given in Table 4.3b. Data for this function were obtained from a wave height analysis for the region (Rossouw 1989). The effect of implementing this function on the dynamics of the Great Brak mouth was investigated using a base flow of 34 x 10^6 $m^3.yr^{-1}$ with seasonality (that is, the natural mean annual run-off) and a wave factor of unity at this stage so as to investigate the longer term seasonal dynamics, rather than an event-driven response. The variation in the height of the sill at the mouth is depicted in Fig 4.3p. The high wave conditions during the winter months (May to July) cause sand to be deposited in the mouth area and the sill height at the mouth increases. This is counteracted by the scouring effect of the base flow, which is able to erode the sill when the wave conditions are lower (August to mid-April). A dynamic balance between these seasonal forces is attained after about three years and a repetitive annual cycle of deposition and erosion of the sill is evident in years three to five of the simulation. At this time the minimum sill height achieved is about 0,59 m to MSL and the maximum is 0,61m to MSL and the average is approximately 0,6 m to MSL. This concurs with estimations of the likely average height of the sill at the Great Brak Estuary prior to extensive development of the catchment.

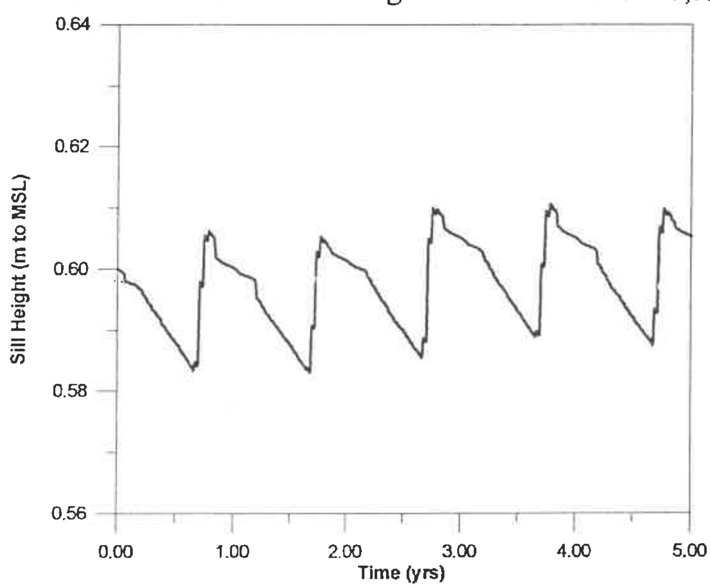

Figure 4.3p Variations in sill height over a five year period occasioned by seasonality in the wave conditions and the freshwater inflow, which amounted to 34 x 10^6 m^3 per annum.

The effects of seasonal changes in wave height on the sill height at the mouth of the estuary, therefore, are accommodated in the model. However, it is not the seasonal alteration in wave height which closes an estuary mouth, although the increased wave heights in winter make closure of the estuary mouth more likely during this season, but a high wave event such as a storm. The effect of such a high wave event was modelled by invoking the wave factor. The results of a simulation with the wave factor set at 100 for a period of three days are presented in Fig 4.3q. The response of the mouth of the Great Brak Estuary to high waves was well simulated in that the mouth closed and the berm across the mouth rose to a height of 2,6 m to MSL. Breaching of the mouth was undertaken when the water level rose to 1,82 m to MSL, the usual practice prior to the construction of the Wolwedans Dam. Following the breaching and associated scour of the mouth, the sill height decreased to 0,55 m to MSL and tidal exchange was re-established. Thereafter, the predominant factor influencing the mouth condition was the seasonal change in wave height (the wave factor was then unity). Thus, the Estuarine Systems Model effectively models the full range of behaviour exhibited by the mouth of the Great Brak Estuary, namely scour of the sill at the mouth following a fresh water release, shallowing of the sill in response to higher wave conditions during the winter months, and closure of the mouth in response to high wave events. Therefore, the sediment sector of the model is considered to be calibrated for the Great Brak Estuary.

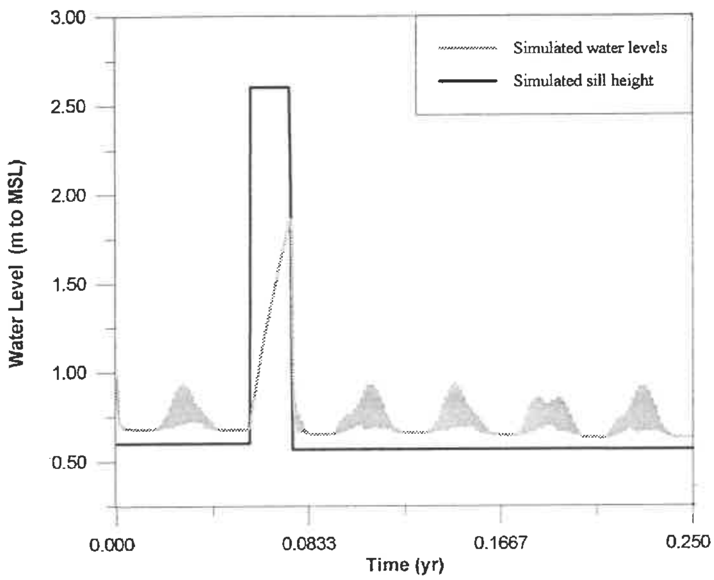

Figure 4.3q Simulated closure and breaching of the mouth of the Great Brak Estuary and the associated variation in water levels over a three month period in response to a high wave event of three day duration.

Unlike the Great Brak Estuary, the mouth area of the Kromme Estuary is characterised by sediment with a median grain size of 0,25 mm (CSIR 1991a). Despite the presence of such fine sediment, studies on sedimentation in the system indicate that major sediment scour occurs in the estuary during floods of return period 10 years and greater, but that little significant scour of sediment occurs under normal tidal conditions (Reddering & Esterhuysen 1983, CSIR 1991a). The volumes of sediment transported from the mouth area by flood flows with return periods of 5 and 10 years as predicted in the CSIR study and those simulated in the Estuarine Systems Model will agree because the standard Ackers-White sediment transport formula (Ackers & White 1973) is used in both cases. However, in calibrating the sediment sector of the model, the volume of sediment eroded or deposited has to be linked with the change in height of the sill at the mouth. It is inappropriate to undertake this linkage using flood flows as the Estuarine Systems Model does not explicitly accomodate the delays associated with the inundation of floodplains and other effects specific to flood models. Consequently, as for the Great Brak Estuary, the calibration of this sector primarily involved setting the value of the parameter l, the length of the mouth channel, so that the erosion and deposition under normal tidal flows (less than 2,5 $m.s^{-1}$ through the mouth) is realistic. Various tidal flow and wave conditions were tested. These caused changes in sill height of at most 0,02 m for a reasonable l value of 72,5 m (selected from a range of 65 m to 80 m) even when the modifications to the Ackers-White formula to accomodate low flows were incorporated. This concurs with the findings of earlier studies that very little scour of sediment occurs under normal tidal conditions and indicates that reasonable confidence can be placed in the calibration of this aspect of the Kromme Model for these conditions.

Because the height of the sill at the mouth is robust to changes in tidal flows and wave conditions, the sediment sector of the Kromme model exerts little direct influence on the predicted water levels, salinities, tidal flushing and stratification-circulation state. Thus, with full interaction between different sectors of the model permitted, the simulation results are considered representative of the known hydrodynamic behaviour of the Kromme Estuary.

This completes the site specific calibration of the physical dynamics component of the estuarine systems model according to the procedure depicted in Figure 4.3a for both the Great Brak and Kromme Estuaries. Prior to the implementation of the management evaluation component of the Estuarine Systems Model, however, it is necessary first to investigate the effects of uncertainty in parameter values on model results and secondly to establish whether the model is able to reproduce typical estuarine dynamic behaviour

4.4 Parameter Sensitivity Analysis

A characteristic of models of the system dynamics type is that parameters have often never been measured specifically and tend to vary in value over the considerable time span of the simulation, behaving more as variables than stable quantities (Tank-Nielsen 1980). Because system dynamics models are generally solved numerically rather than analytically, only particular solutions are obtained and changes in parameter values cause changes in the numerical values computed during simulations. The models, therefore, exhibit numerical sensitivity. A thorough investigation of the effects of parameter uncertainty is required before confidence can be placed in the reliability of simulation results. This investigation forms an intrinsic part of the modelling effort (Wong 1980) and just as the selection of the technique and the formulation of the model are undertaken with both the natural behaviour of the system and the purpose of the model in mind, so must the sensitivity analysis relate to these aspects of the modelling endeavour.

An investigation of the sensitivity of the results of the Great Brak and Kromme Models to parameter perturbations is necessary in view of the fact that very few parameter values are known precisely. For these models, then, the sensitivity analysis effort must relate both to the representative simulation of the physical dynamics of the estuaries and the evaluation of the efficacy of different management policies involving water releases and mouth breachings. This resolves into two major questions, namely:

- How do uncertainties in the assignment of parameter values affect the simulation results of the physical dynamics components of the models? and
- To what extent do these effects influence the management recommendations or decisions?

As the management evaluation component of the estuarine systems model has not yet been implemented, attention will focus on the effects of parameter perturbations on the physical dynamics components of the two model applications at this stage.

The physical dynamics component of the estuarine systems model was formulated to yield representations of the physical state of an estuary over time scales of months to years in response to fresh water inflows and mouth breachings. The physical state of the estuary is indicated in terms of the water volume, the salt content, the stratification-circulation state, the fresh water and tidal flushing of the water body and the sill height at the mouth at any time. The quotient of the salt content and the water volume, two state variables, is an auxiliary variable which provides an indication of the mean salinity in the estuary at any time, while the likely head to mouth differences in salinity across the estuary under different inflow and mouth conditions are indicated by a further auxiliary variable, the axial salinity difference. Thus this sensitivity analysis must address the influence of uncertainties in the parameter values on the simulated values of the seven state variables and the predicted mean salinities and axial salinity differences.

4.4.1 The sensitivity analysis approach

System dynamics models such as the estuarine systems model can generally be described by a system of n non-linear differential equations of the form:

$$\frac{dx_j}{dt} = F_j(\mathbf{x}, \mathbf{p}, t) \quad j = 1, 2, ..., n$$

where $\mathbf{x} = (x_1, x_2, ..., x_n)^T$ and $\mathbf{p} = (p_1, p_2, ..., p_m)^T$ represent the state variables at time t and the parameters of the system, respectively. Normalized sensitivity functions, N_{jk}, defined as:

$$N_{jk} = (\partial x_j / \partial p_k) \cdot (p_k / x_j) \quad j = 1, 2, ..., n \quad k = 1,2, ..., m$$

then yield the approximate percentage change in the variable x_j at time t in response to a one percent increase in the value of the parameter p_k.

This traditional approach (Tomovic 1963) has been applied extensively over the thiry-two years since its inception to indicate the effect of individual parameter perturbations on model results. It is particularly applicable to models with few state variables and a limited number of parameters, but can become unwieldy when applied to large models with many parameters. These factors led Drewes and Slinger, amongst others, to modify the traditional method when undertaking sensitivity analyses of large models (Drewes 1987, Slinger 1988) by considering the effects of perturbations of selected parameters on a subset of the state variables and by considering the effects of perturbations of the rate equations rather than the parameters themselves. Although these methods proved applicable in the cases concerned, they are not required for the estuarine systems model because there are only seven state variables, two auxiliary variables of relevance to the model output and eleven endogenous parameters. The small number of endogenous parameters is misleading, however, as many of the functions which would be parameterised as table functions in a standard system dynamics application are calculated by linear interpolation of pre-determined function values in the estuarine systems model eg. SCF(WL) and GCF(x_4). The effects of uncertainties in these function values also need to be considered in a comprehensive sensitivity analysis of the physical dynamics component of the model. Such an analysis can be accommodated by the inclusion of additional parameters for each of these functions. This increases the number of parameter perturbations to be considered in the application of the traditional approach to twenty-one or greater, depending on whether one or more parameters are required per function.

In establishing numerical sensitivity, the effects on model output of small perturbations in the exogenous functions and the initial values of the seven state variables also need to be considered. A particular choice of exogenous functions and initial values results in a solution trajectory of the model in seven dimensional state space. It is the extent of perturbation from the standard trajectory of the individual state variables at any time, owing to a parameter perturbation, which is captured by the normalised sensitivity function. The inclusion of perturbations of the exogenous function parameters in the sensitivity analysis at this stage will provide an indication of the sensitivity of the model results to a narrow range of external forcing. The sensitivity of the model output to small perturbations of the exogenous function parameters yielding the standard trajectory (described below) is included here for completeness sake and because undue sensitivity will highlight possible deficiencies in the model structure. These considerations increase the number of parameter perturbations to be undertaken and hence the number of simulations of the Great Brak and Kromme Models to be conducted from twenty-one or more to thirty-six. However, this increase is not unmanageable and the traditional approach can still be applied relatively easily.

The major disadvantage of the traditional sensitivity analysis method, that only the effects of individual parameter perturbations of small magnitude are considered, has already been mentioned. It is a well established fact that a combination of parameter changes can result in substantial changes in model outputs (Vermeulen & De Jongh 1976, Hearne 1985), particularly if perturbations in parameter values of 10% and greater are allowed. This fact has led to the development of the parameter combination and the objective function sensitivity analysis methods (Hearne 1985, 1986). These methods indicate the most sensitive direction in parameter space, that is, the direction in which a perturbation of the parameter vectors maximises the displacement of the solution trajectory or the objective function, respectively. As such, they are more appropriately considered for application following the implementation of the management evaluation component of the model and the determination of the state variables of most relevance to management (Swart 1987). Further sensitivity analysis methods include the dynamical systems approach, which analyses the equilibrium configuration rather than transient behaviour and considers parameter variations across the full range of possible values. Such techniques have been applied to models with up to ten state variables and a fair number of parameters, but some scaling and smoothing of functions is required (Van Coller 1995). However, when either exogenous or endogenous seasonal forcing is present, such dynamical analysis techniques barely cope with models with analytical solutions (Hastings *et al.* 1993, Gragnani & Rinaldi 1995) and certainly are not appropriate for an investigation of a model such as the estuarine systems model which exhibits cyclical forcing on both a seasonal and tidal scale (L van Coller *pers. comm.*).

The approach adopted in the sensitivity analysis of the physical dynamics component of the model, therefore, comprised the traditional sensitivity analysis method of individually perturbing the endogenous parameters and included a consideration of the effects of perturbations of individual endogenous function parameters, exogenous function parameters and the initial values of the state variables.

4.4.2 **The standard trajectories of the Great Brak and Kromme models**

Before evaluating the sensitivity of the Great Brak and Kromme models to parameter perturbations, it is necessary to determine a standard trajectory for use as the reference path from which the deviations in the values of the state and auxiliary variables will be measured.

It is common practice to use the earliest historical records or recorded antecedent state of an aquatic system as the reference state against which the severity of changes may be assessed. In this way, the natural or undisturbed state of the aquatic ecosystem is approximated and this provides the reference point against which comparisons are made.

A similar approach was followed in generating standard trajectories for the Great Brak and Kromme Estuaries. The exogenous functions were specified with the long term average situation of the undeveloped catchments in mind. Thus the inflow functions, IF(t), were parameterised as seasonally varying base flows with annual average volumes equivalent to the natural mean annual run-offs of 34 x 10^6 m^3 and 120 x 10^6 m^3 for the Great Brak and Kromme catchments, respectively. No floods were included. The marine forcing comprised tidal variations with the semi-diurnal and spring-neap magnitudes and time scales characteristic of the Mossel Bay and Port Elizabeth areas, respectively. Seasonal variations in wave heights were included, but no extreme tidal levels or high wave events were allowed (i.e. MF(t) = 1 & WF(t) = 1). The results of five year simulations of the Great Brak and Kromme Estuaries under these average natural run-off conditions are presented in Figures 4.4a and 4.4b and are described below. Each simulation commenced on 1 October, the start of the hydrological year.

In the case of the small Great Brak Estuary, the maximum tidal exchange is about 12 $m^3.s^{-1}$ through the mouth of the estuary. Water levels in the estuary basin vary from 0.65 to 0.95 m to MSL (Fig 4.4a), exhibiting elevated low water levels over spring tides and during spring and summer when the sill at the mouth decreases slightly in height (Fig 4.3p) and tidal exchange is enhanced. The mean salinities in the estuary vary from nearly zero on ebb tides during spring and summer to 26 $kg.m^{-3}$ on the flood tides during winter (Fig 4.4a). The axial difference in salinity remains at 35 $kg.m^{-3}$ throughout the simulation period. The stratification index varies from less than 0.005 to 0.03, showing the most variation in response to differences in tidal forcing over the low flow winter period (Fig 4.4a). The circulation index varies from a minimum of about 0.02 during winter to a maximum of approximately 0.06 in summer (Fig 4.4a). The estuary basin, therefore, remains in a partially mixed state throughout the simulation (refer to Fig 3.4a). The fresh water flushing and the tidal flushing variables vary from 3 to 11.5 and 150 to 530 yr^{-1}, respectively (Fig 4.4a). Both exhibit minima in winter, indicating that the extent of tidal flushing is strongly related to the fresh water inflow and the associated mouth condition under the natural run-off scenario. Finally, the height of the sill at the mouth of the estuary varies from a minimum of 0.584 m to MSL in the winter months of the first year of the simulation to a maximum of about 0.61 m to MSL in the summer months of years three to five of the simulation. A repetitive cycle of alternate maxima of 0.61 m to MSL in winter and minima of about 0.59 m to MSL in autumn manifests in years three to five of the simulation (Fig 4.3p) indicating that a dynamic equilibrium is only achieved at this stage of the simulation. All state variables exhibit this stable periodic behaviour in years three to five of the simulation, indicating that transient behaviour has died out by this stage eg. the influence of the initial values selected for the various state variables. The last two years of the five year simulation period, therefore, are an appropriate choice of time period over which to evaluate the effects of parameter perturbations on the output of the Great Brak model.

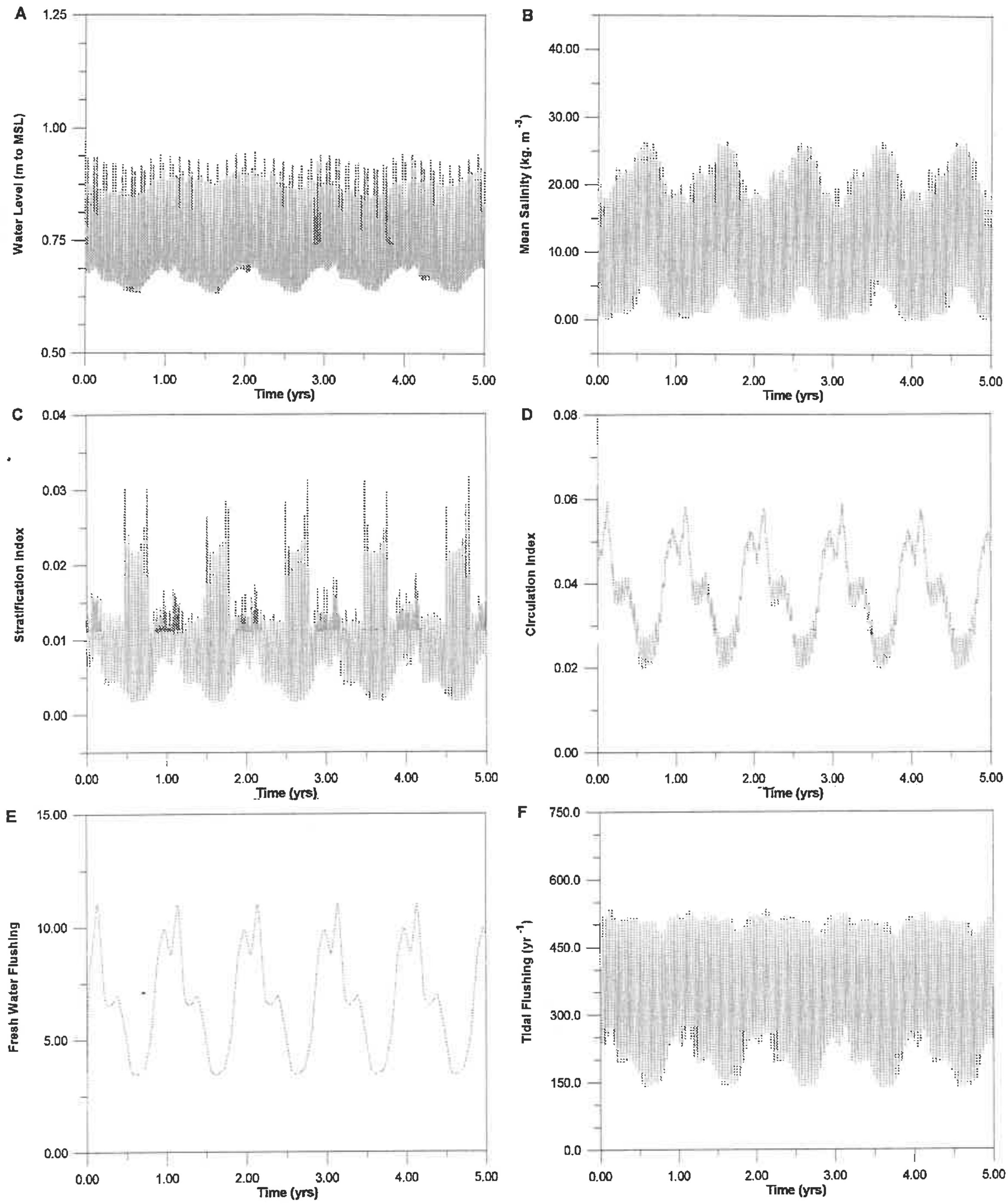

Figure 4.4a The simulated water levels (A), mean salinities (B), stratification-circulation state (C & D) and freshwater and tidal flushing (E & F) of the Great Brak Estuary over five years under a seasonal freshwater flow amounting to 34 x 10^6 m^3 per annum and seasonal wave conditions. The corresponding variation in sill height is depicted in Fig 4.3p.

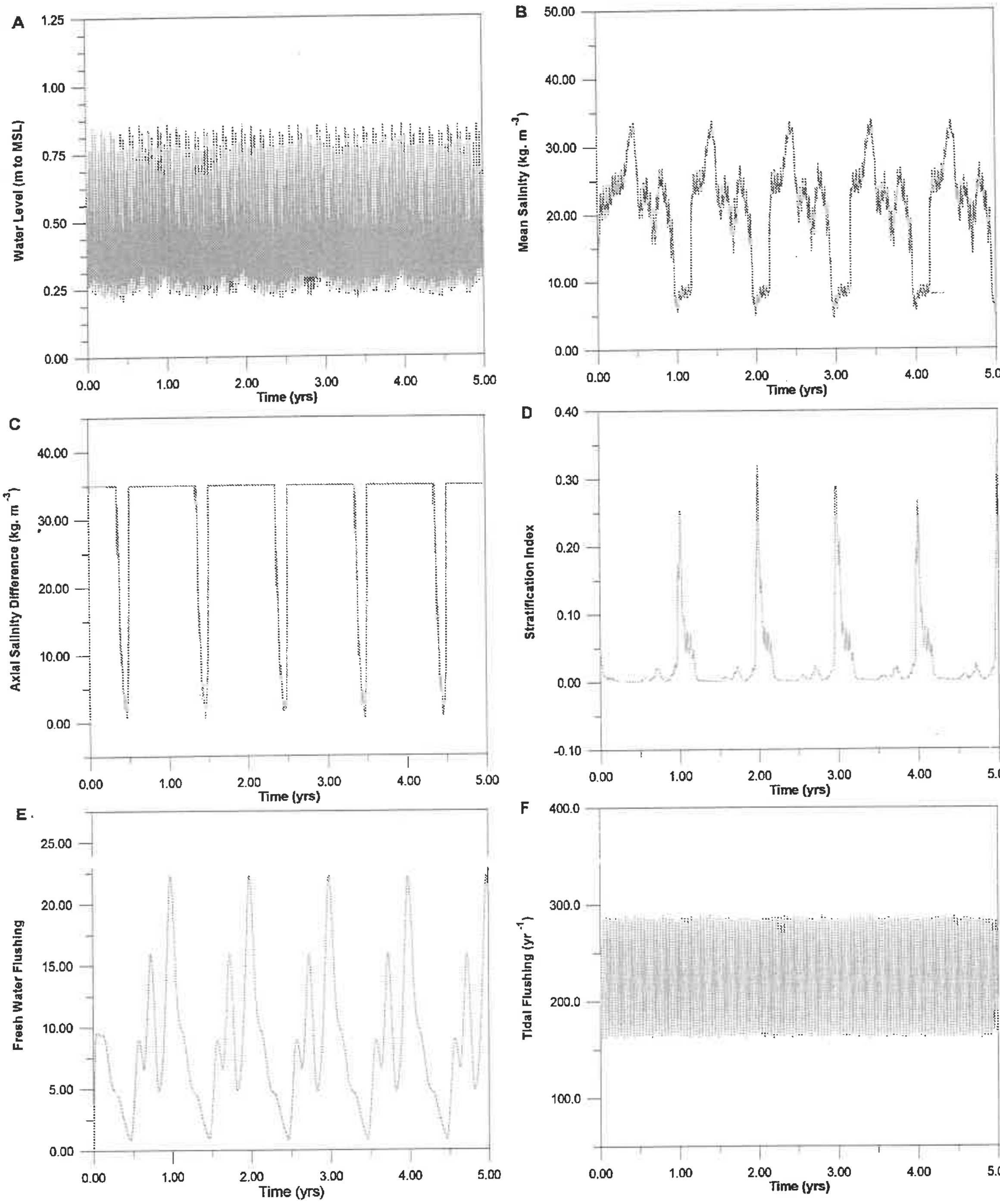

Figure 4.4b The simulated water levels (A), mean salinities (B), axial salinity differences (C), stratification state (D) and freshwater and tidal flushing (E & F) of the Kromme Estuary over five years under a seasonal freshwater flow amounting to 120 x 10^6 m^3 per annum and seasonal wave conditions.

The tidal flux of the larger Kromme Estuary achieves a maximum of 165 $m^3.s^{-1}$ on some spring flood tides and the greatest variation in water levels, from a minimum of 0.19 m to MSL to a maximum of 0.86 m to MSL, occurs on spring tides in autumn (Fig 4.4b). The sill height at the mouth of the estuary remains approximately 0.09 m to MSL throughout the five year simulation.

In contrast, the mean salinities in the estuary vary from a minimum of 6 kg.m^{-3} during September and October to a maximum of 33 kg.m^{-3} in March when the fresh water inflow to the estuary is lowest and the differences in salinity between the head and the mouth reaches of the estuary exhibit a dramatic decline from 35 kg.m^{-3} to nearly zero in the month of March every year (Fig 4.4b). It is evident that even under the average natural run-off scenario, the salinities of the Kromme Estuary are sensitive to decreases in fresh water inflow. The circulation index of the Kromme Estuary remains unity throughout, but the stratification index exhibits annual October maxima of about 0.35, indicating that although some vertical differencing in salinity does exist, the Kromme Estuary is a well mixed system under natural run-off conditions. Fresh water flushing varies from a March minimum of 1 to 22.5, while the tidal flushing varies from 160 to 290 yr^{-1} (Fig 4.4b). A periodic dynamic equilibrium is attained in years one to two of the Kromme model simulation, indicating that these years are an appropriate time period over which to evaluate the effects of parameter perturbations on the model output.

4.4.3 **Sensitivity analysis results**

The maximum and the average magnitude of the normalised sensitivity co-efficients arising from one percent perturbations in thirty-seven parameters were determined for years three to five in the case of the Great Brak Estuary and years one to two for the Kromme Estuary. The thirty-seven parameters comprised the eleven endogenous parameters (L, B, w, l, t_1, t_2, t_3, x_{11n}, DIAM, Ps & BSF), eleven parameters from the endogenous functions (SA(x_1), WLF(x_1), SCF(WL), VF(TWL,WL), GCF(x_4), SEM(x_5,x_6), SEF(x_3,x_4), DF(x_2/x_1)), the seven initial values of the state variables and eight parameters (BN, RN, EN, GN, MSL, HLA, SNA & WF) from the exogenous functions.

The Great Brak Estuary

The endogenous parameters and functions are ranked in Table 4.4a according to the maximum magnitude of the normalised sensitivity co-efficients of any of the affected state variables of the Great Brak model during the simulation years three to five. All endogenous parameter perturbations which caused the normalized sensitivity co-efficients for some of the state variables to exhibit maximum magnitudes or average magnitudes greater than the arbitrarily selected values of 0.040 or 0.015, respectively, are included. The associated state variables and the average magnitudes of the normalised sensitivity values over the selected period are also listed. In Table 4.4b, the parameters of the exogenous functions with a similar degree of influence are ranked in a similar manner.

Table 4.4a The individual endogenous parameters and functions exerting most influence on the Great Brak model results as evidenced by the maximum normalised sensitivity co-efficients and the average absolute normalised sensitivity values of the affected state variables.

PARAMETER OR FUNCTION	MAXIMUM N_{jk}	AVERAGE $\lvert N_{jk} \rvert$	STATE VARIABLE
DIAM	1.039	0.043	x_2
	-0.200	0.025	x_3
	0.097	0.049	x_6
	-0.044	0.020	x_1
	-0.040	0.021	x_4
	-0.024	0.022	x_7
w	0.685	0.026	x_2
	-0.174	0.018	x_3
	0.053	0.031	x_6
BSF	-0.502	0.018	x_2
	0.170	0.013	x_3
	-0.038	0.021	x_6
SCF(WL), GCF(x_4), VF(TWL,WL)	-0.262	0.009	x_2
	0.157	0.007	x_3
L	0.251	0.009	x_2
	-0.073	0.006	x_3
l	-0.135	0.006	x_2
	0.115	0.005	x_3
WLF(x_1)	0.119	0.005	x_3
	-0.091	0.003	x_2
SEM(x_5,x_6), SEF(x_3,x_4)	-0.059	0.002	x_2
DF(x_2/x_1)	-0.059	0.0002	x_2
B	0.041	0.0002	x_2

The majority of the state variables show limited sensitivity to endogenous parameter perturbations as all but three of the maximum normalised sensitivity co-efficients have magnitudes of less than 0.5 in response to one percent parameter perturbations and the average normalised sensitivity co-efficients are less than 0.05 for every parameter perturbation. This indicates that the physical dynamics component of the Great Brak model is reasonably robust to parameter uncertainty, although care should be taken in the specification of the most influential parameters.

The three largest deviations in the normalized sensitivity co-efficients, namely 1.039, 0.685, and -0.502, arise from perturbations of the parameters DIAM, w and BSF. These parameters all relate to the condition of the mouth of the Great Brak Estuary as they determine the sediment mobility in the mouth area, the width over which deposition and tidal flow occurs and the enhanced transport of sediment owing to wave stirring. The fourth and fifth largest deviations (-0.262 and 0.251) are associated with the functions affecting the tidal flux and the parameter L, which determines the distance over which the tidal head characteristically acts. Yet, the state variable most affected by these parameter perturbations is the salt content, x_2, rather than the sill height or the water volume. This highlights a structural feature of the model, in that the import and export of salt is directly dependent on the tidal flux through the mouth, and reflects the sensitivity of salt content as an indicator of the degree of tidal exchange and renewal of water in a small estuary.

The stratification index also shows consistent sensitivity to parameter perturbations. The maximum normalised sensitivity co-efficients ranging from 0.200 to 0.119 in magnitude relate to this state variable. Again the three largest of these deviations arise from perturbations of the parameters DIAM, w and BSF, while the other two of the five values in this range arise from perturbations of the functions affecting the tidal flux and the water level function. This finding emphasises the responsiveness of the Great Brak Estuary to the condition of the mouth and the tidal flux through the mouth. Some sensitivity to perturbations of the parameters DIAM and w is also exhibited by the state variable for tidal flushing, x_6, re-iterating the importance of tidal exchange to the physical state of the estuary.

The sensitivity of the Great Brak model to one percent perturbations in the exogenous parameters is fair with all but two of the maximum normalised sensitivity co-efficients having magnitudes of less than 0.2 and no average normalised sensitivity co-efficient exceeding 0.025. The largest deviation of -0.502 arises from an increase in the wave function and relates to the state variable, x_2, emphasising once again the sensitivity of the salt content of the estuary to the mouth condition. The second largest deviation arises when an increase in the base flow causes a change in the stratification index. Thus, no undue sensitivity to exogenous forcing is exhibited by the model, enhancing the confidence which may be placed in the reliability of the output.

One percent perturbations of the initial values of the state variables had no noticeable effect on the model results and all maximum normalised sensitivity co-efficients were less than 0.01.

Table 4.4b The parameters of exogenous functions exerting most influence on the Great Brak model results as evidenced by the maximum normalised sensitivity co-efficients and the average absolute normalised sensitivity values of the affected state variables.

EXOGENOUS FUNCTION	ASSOCIATED PARAMETER	MAXIMUM N_{jk}	AVERAGE $\lvert N_{jk} \rvert$	STATE VARIABLE
WF(t)	-	-0.502	0.018	x_2
		0.170	0.013	x_3
		-0.038	0.021	x_6
BF(t)	BN	-0.272	0.010	x_3
		-0.072	0.013	x_2
TF(t)	HLA	0.140	0.022	x_2
		-0.085	0.010	x_3
	SNA	-0.112	0.004	x_2
		0.111	0.004	x_3

The Kromme Estuary

The endogenous parameters and functions are ranked in Table 4.4c according to the maximum magnitude of the normalised sensitivity co-efficient of any of the affected state variables of the Kromme model during the simulation years one to two. All endogenous parameter or function value perturbations which caused the normalized sensitivity co-efficients for some of the state variables to exhibit maximum magnitudes or average magnitudes greater than the arbitrarily selected values of 0.02 or 0.005, respectively, are included. The associated state variables and the average magnitudes of the normalised sensitivity values over the simulation period are also listed. The parameters of the exogenous functions with a similar degree of influence are ranked in Table 4.4d in a similar manner.

The sensitivity of the Kromme model output to endogenous parameter and function perturbations is fair with all but two of the maximum normalised sensitivity co-efficients having magnitudes less than 0.20 and all average normalised sensitivity co-efficients being 0.03 or less. The most influential function is the reduced salinity function, which affects the import of salt to the estuary. This function affects the state variables for salt content and stratification most significantly. The second most influential functions are those relating to the export of salt from the estuary. Again perturbations in these functions affect the state variables for salt content and stratification most significantly. It is noteworthy that, in contrast to the Great Brak Estuary, the endogenous parameters to which the system is most sensitive relate directly to the salt content and not to the state of the mouth. This is reflective of the the larger size of the estuary, the larger tidal prism and the robustness of the mouth of the Kromme Estuary to alterations in inflow and sea conditions. A further point to note is that the magnitudes of the deviations exhibited by the Kromme model are generally lower than those of the Great Brak model, indicating that the Kromme model results are less sensitive to parameter perturbation than the Great Brak model outputs.

Table 4.4c The individual endogenous parameters and functions exerting most influence on the Kromme model results as evidenced by the maximum normalised sensitivity co-efficients and the average absolute normalised sensitivity values of the affected state variables.

PARAMETER OR FUNCTION	MAXIMUM N_{jk}	AVERAGE $\lvert N_{jk} \rvert$	STATE VARIABLE
$RS(x_3,x_4)$	0.342	0.012	x_2
	0.084	0.030	x_3
$SEM(x_5,x_6)$, $SEF(x_3,x_4)$	-0.241	0.012	x_2
	0.067	0.029	x_3
B	0.032	0.0002	x_2
	0.022	0.009	x_3
L	0.030	0.007	x_3
SCF(WL), $GCF(x_4)$, VF(TWL,WL)	0.026	0.001	x_2
	0.021	0.008	x_3
$DF(x_2/x_1)$	0.022	0.009	x_3

The perturbation of certain of the exogenous parameters of the base flow function and the tidal function exerted some influence on the state variables for salt content and stratification in the Kromme Estuary. However, the degree of sensitivity exhibited was again fair with all maximum normalised sensitivity co-efficients having magnitudes less than 0.01 and average magnitudes less than 0.02. Thus the Kromme model output is reasonably robust to small changes in exogenous parameters.

As in the case of the Great Brak Estuary, one percent perturbations of the initial values of the state variables had no discernible effect on the Kromme model results and all maximum normalised sensitivity co-efficients arising from these perturbations were less than 0.01.

Table 4.4d The parameters of exogenous functions exerting most influence on the Kromme model results as evidenced by the maximum normalised sensitivity co-efficients and the average absolute normalised sensitivity values of the affected state variables.

EXOGENOUS FUNCTION	ASSOCIATED PARAMETER	MAXIMUM N_{jk}	AVERAGE $\|N_{jk}\|$	STATE VARIABLE
BF(t)	BN	0.073	0.017	x_3
		-0.041	0.003	x_2
TF(t)	MSL	-0.026	0.005	x_3
	HLA	-0.021	0.004	x_3

The parameter sensitivity analysis of the Great Brak and Kromme models has indicated that the models are reasonably stable with regard to small parameter uncertainties. However, the Great Brak model results exhibited a fair degree of sensitivity to the state of the mouth and the sediment grain size parameter, in particular. This argues that attention should be focussed on accurately specifying or measuring the parameters related to the mouth condition during field work and indicates the importance of concurrent time series of water levels, salinities and sill heights for the accurate calibration of the model. Thus, although a fair degree of confidence may be placed in the reliability of the outputs of the physical dynamics component of the estuarine systems model, the confidence placed in these and future case study results could be enhanced by the collection of appropriate monitoring data.

4.5 Generic Estuarine Behaviour

The final stage in the validation of the physical dynamics component of the Estuarine Systems Model lies in establishing whether common behavioural modes of estuaries are simulated by the model and, if so, to what extent. The aspects of estuarine behaviour which have not yet been addressed, but which are well documented or commonly observed features of estuaries and so require investigation, include:

- the equilibrium flow area and inlet stability relationships due originally to O'Brien (1931, 1969) and Bruun and Gerritsen (1961), and
- the attenuation of high frequency tidal signals and the effect of low frequency signals on water levels in shallow estuaries (Aubrey & Speer 1985, Brundrit *et al.* 1988, MacKay & Schumann 1991).

Both features deal with the interaction between the mouth and the water levels and tidal exchanges in an estuary, that is, the water volume sector of the model.

4.5.1 Equilibrium flow area and inlet stability

Over the last sixty-five years some interest has centered on the stability of tidal inlets situated on exposed sandy coasts with negligible or no river discharge (O'Brien 1931, 1969, Bruun & Gerritsen 1960, Bruun 1978, Jarrett 1976, Hume 1991, Hume & Herdendorf 1993). Most of these investigations have focussed on the development of empirical throat area to tidal prism relationships. Whereas O'Brien (1931, 1969) proposed a linear relationship between the cross-sectional area below MSL at the inlet throat and the tidal prism of the system, Jarrett (1976) found slightly non linear relationships for North American inlets. However, O'Brien's relationship:

$$A_{MSL} = 6.56 \times 10^{-5}\ m^{-1}.\ P$$

where A_{MSL} = mouth cross sectional area below MSL (m^2)
P = tidal prism (m^3)

was found most applicable in other areas of the world (Hume 1991, Hume and Herdendorf 1993). Bruun and Gerritsen (1960) and Bruun (1978), on the other hand, characterised the stability of tidal inlets in terms of the ratio between opening forces and closing forces. The erosive action of the outflowing water was considered to comprise the opening force, encapsulated in the tidal prism volume, whereas the depositional action of longshore drift on an open coast was viewed as the closing force. Although, recent investigations have indicated that cross-shore rather than longshore sediment movement is responsible for the shallowing and closure of the mouths of South African estuaries (CSIR 1992a, 1992c, Huizinga 1994), the concept of competition between erosive and depositional forces in determining the state of the mouth is considered correct, as is the view that the stability of a tidal inlet can alter in relation to seasonal alterations in the depositional forcing (Bruun 1978). The idea of dynamic change is certainly more applicable to South African inlets, which have the additional complication of alterations in fresh water inflow and hence in the tidal flux through the mouth, rather than the static equilibrium envisaged by O'Brien and his adherents. Although neither concept is directly applicable to South African systems, both are addressed qualitatively by the Estuarine Systems Model. The formulation of the sediment sector of the Estuarine Systems Model has as a basis the competition between erosional and depositional forces in the determination of the mouth state. Additionally, because the cross sectional areas below MSL of the inlets of the Great Brak and Kromme Estuaries are frequently zero, they are inherently unstable according to O'Brien's formula. However, if we assume linear proportionality between the spring tidal prism and the mouth cross-sectional area as given by O'Brien and note the finding of Gerritsen & De Jong (1985) that the constant is less where littoral drift is less relevant, we may estimate the maximum cross-sectional area at the mouth of these systems. For the Great Brak and Kromme Estuaries these values are calculated as 8.5 m^2 ($P = 1.29 \times 10^5$ m^2) and 114.8 m^2 ($P = 1.75 \times 10^6$ m^2), respectively. If we then take the width of the mouths of these estuaries as 18 m and 72.5 m, respectively, the corresponding estimates of the maximum depths of the mouth are 0.47 m and 1.58 m. These estimates are of the correct order of magnitude for each of the systems, indicating that while O'Brien's relationship cannot be considered realistic nor applicable to South African systems, the relationship between mouth cross sectional area and tidal prism in the Estuarine Systems Model still does not deviate substantially from the O'Brien formula.

4.5.2 **Tidal and long period water level variations**

Considerable distortion of the offshore tidal signal is a feature commonly observed within many shallow estuaries and bays (Aubrey & Speer 1985, MacKay & Schumann 1991, Black *et al.* 1993). Modelling investigations of the response of shallow systems to forcing of different frequencies from the adjacent coastal ocean have indicated that the estuary mouth acts as a low pass filter to first order (Speer and Aubrey 1985, Brundrit *et al.* 1988, Scott 1994). As such, the mean water level within a basin is modulated by longer period forcing commonly associated with synoptic scale weather events or the spring neap tidal cycle, while higher frequency forcing is subjected to strong frictional attenuation and non linearities cause the generation of overtides (Aubrey & Speer 1985, MacKay & Schumann 1991, Hill 1994).

The strong attenuation of the semi-diurnal tidal signal and the elevation of low water levels over spring tides are features of both the Great Brak and Kromme Estuaries which were demonstrated in the site-specific calibration of the Estuarine Systems Model (Section 4.3.1). However, the generation of overtides within the estuaries and the influence of weather events on water levels have not been investigated. Rather, the tidal exchange through the mouth of the estuary and hence the degree of modulation of the estuarine water levels owing to marine tidal and synoptic forcing was formulated without specific reference to these aspects (Section 3.2.4). An investigation of the ability of the Estuarine Systems Model to generate the documented estuarine behaviour, therefore, will provide a stringent test of the validity of the treatment of the tidal flux rate.

Accordingly, the influence of the mouth (as modelled in the Estuarine Systems Model) on tidal fluxes and water levels within the Great Brak and Kromme Estuary basins was investigated by:

- eliminating seasonality from the wave conditions and applying constant fresh water inflow rates of 34×10^6 m^3 and 120×10^6 m^3, respectively,
- initially applying marine forcing in the form of a tidal function with the semi-diurnal, diurnal and spring-neap influences characteristic of the Mossel Bay and Port Elizabeth regions, respectively, and simulating the water level variations in the estuaries,

- subsequently incorporating synoptic modulations of the mean sea level with periods of 5 and 10 days in the input signal and generating estuarine water levels,
- conducting spectral analyses of the resulting estuarine water levels to establish whether attenuation, amplification, overtide generation or modulation of low water levels occurred, and
- finally adjusting the depths of the estuarine basins relative to mean sea level and observing the altered distortion of the tidal signals.

The tidal forcing functions applied to each system and the resultant water level variations in the estuarine basins are depicted in Figure 4.5a. The tidal functions comprised the semi-diurnal M_2 tide and the spring neap tidal variation characteristic of the Mossel Bay and Port Elizabeth regions (Section 4.3.1), with a diurnal modulation of amplitude 0.1 in both cases. Thus the amplitudes of variation of the spring tidal forcing were 1.75 m and 1.61m for the Great Brak and the Kromme models, respectively, and those at neap tide were 0.57 m and 0.51 m, respectively. The amplitude of the synoptic event was chosen as 0.40 m to correspond with amplitudes commonly associated with the coastally trapped waves which occur along the south east Cape coast (Schumann & Brink 1990, MacKay & Schumann 1991). The spectral densities, which were calculated per 8,4 Hz bandwidth using a Parzen window (IMSL 1991), are presented in Figure 4.5b in response to purely tidal forcing (TF) and tidal forcing with continous synoptic events of 10-day (TF-S10) and 5-day periods (TF-S5).

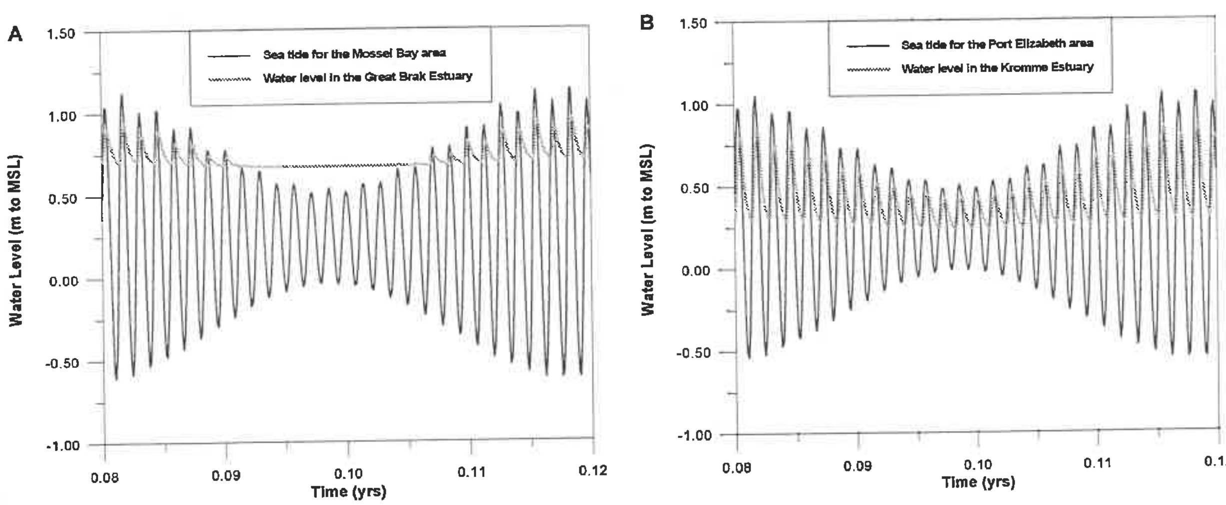

Figure 4.5a The variations in the water levels of the Great Brak (A) and Kromme Estuaries (B) owing to sea tidal forcing from the adjacent coastal ocean.

The strong attenuation of high water levels in the Great Brak Estuary and the influence of the spring neap tidal variation on low water levels which is evident in Figs 4.5a, 4.3d and 4.3e., is confirmed in the spectra of Figure 4.5b. The majority of the spectral energy density is located in the lower frequencies, indicating considerable damping of the higher frequency semi-diurnal signal and responsiveness to forcing of the low frequencies associated with the spring neap tidal cycle and the passage of synoptic weather systems or coastally trapped waves. The generation of an overtide at double the frequency of the semi-diurnal signal is highly encouraging as this six-hourly component of variation corresponds with the M_4 overtide noted by Aubrey & Speer (1985) and MacKay & Schumann (1991). A similar feature is observed in the spectrum from the Kromme Estuary. The generation of overtides owing to non-linear frictional effects in shallow water systems has been associated with flood dominance (Aubrey & Speer 1985), a notable aspect of both the Kromme and the Great Brak Estuaries. Considerable attenuation of the semi-diurnal tidal signal also occurs in the Kromme Estuary, although this is not as severe as in the Great Brak system. Modulation of the low water levels in the system due to the influence of synoptic events or the spring neap tidal cycle are substantial, yet less effective than in the Great Brak system. These observations concur with the finding by Brundrit *et al.* (1988) that with increasing protection (shallower, narrower mouth), the marine influence on estuarine water levels shifts from a tidal to a synoptic weather dominance.

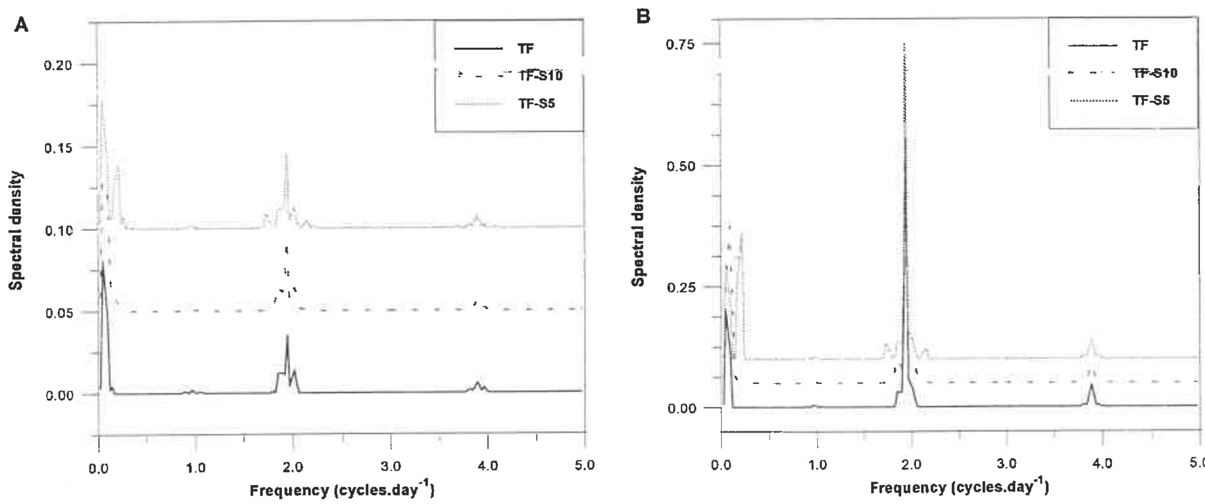

Figure 4.5b The spectral densities for the Great Brak (A) and Kromme Estuaries (B) in response to purely tidal forcing (TF), and tidal forcing with continous synoptic events of 10-day (TF-S10) and 5-day periods (TF-S5). The zeros of the TF-S10 and TF-S5 spectra are offset by 0.05 and 0.10, respectively.

A comparison of the water levels for estuary basins with the morphology of the Kromme system, but 0,5 m and 1 m deeper relative to MSL, indicates that the influence of the semi-diurnal tide propagates more strongly within the estuarine basin with increasing depth (decreasing protection of the mouth) and the degree of modulation of the water levels by spring neap tidal influences decreases with an increase in the depth. These findings confirm that the estuarine systems model simulates the enhanced low pass filtering action of the mouth with increasing constriction particularly well.

In summary, two non-linear features of estuarine behaviour, namely, the generation of overtides and the modulation of the mean water level within a shallow estuary by low frequency marine forcing, are well simulated by the Estuarine Systems Model although they were not explicitly considered in the formulation of the tidal exchange through the mouth. This enhances the confidence which may be placed in the ability of the model to generate the realistic physical dynamics of South African estuaries and completes the calibration and validation of this aspect of the model.

5. MANAGEMENT APPLICATIONS AND POLICY EVALUATION

The range of management applications of the Estuarine Systems Model is diverse, because the issues to be addressed in the management of South African estuaries are diverse (refer to Sections 1.2, 3.7 & 4.2). The management questions encompass both site specific concerns such as the allocation of freshwater to an estuary as well as regional concerns such as deciding which systems should be preferred targets for water abstraction. In each case the factors to be considered include the physical environmental response of the estuary to alterations in freshwater flows and mouth management actions, as well as the biological health of specific estuaries and of the regional estuarine resources eg. fish stocks, prawn populations, intertidal and salt marsh vegetation. To demonstrate the utility of the Estuarine Systems Modelling approach in addressing the management issues and yet highlight the inadequacies inherent in the use of any particular model, the implementation of the management evaluation component of the model will be tackled in three stages.

First, the ***site specific application*** of the model in predicting the consequences of reduced freshwater flow to the Great Brak and Kromme Estuaries and in evaluating various management options, will be described. This will highlight the predictive capability of the Estuarine Systems Model by demonstrating how a picture of the dynamic physical state of an estuary in response to management policies involving abstraction levels, water releases and even mouth breachings, is derived. In the case of the Great Brak Estuary, the average long term dynamic states and the response of the system to episodic events will be described, whereas in the Kromme case study, the choice of one management policy above others based on the resulting salinity regime will be demonstrated. Secondly, the utility of the model in a more holistic predictive approach, which incorporates simulating the effects of alterations in the physical environment on selected estuarine biota, will be investigated. Studies undertaken in collaboration with members of the Consortium for Estuarine Research and Management to explore the construction and use of a linked modelling approach with the Estuarine Systems Model as a basic building block (modelling the physical environment) will be described. The utility of this ***integrated modelling approach*** as well as limitations of the approach and the role of the Estuarine Systems Model in such a use will be highlighted. Finally, the potential ***strategic management use*** of the model in water resource development planning will be explored.

5.1 Site Specific Management Applications: The Great Brak Estuary

The first case study selected for application of the Estuarine Systems Model was the Great Brak Estuary. One of the criteria in this selection was the type of management problem associated with the estuary (Section 4.2), which is impounded 3 km upstream of the head of tidal influence and presently has fresh water inflows substantially less than under natural (undeveloped) conditions. An understanding of the effects of the reduced freshwater flow on the character and functioning of the systems is required, while strategies for the management of freshwater releases to benefit both the biotic and abiotic environment, while still addressing water demands in the Mossel Bay area, are a necessity.

5.1.1 Management performance indices

Key issues of concern identified in studies of the small, intermittently closed Great Brak Estuary are the state of the estuary mouth (open, closed, constricted) and the water quality of the system (old saline bottom water, faecal coliform counts etc.), which has been linked to the efficacy of tidal flushing and the stratification state of the system (CSIR 1990a, Taljaard & Slinger 1993, Slinger *et al.* 1994). Clearly the management performance indices chosen for the system must reflect these concerns. Thus the indices selected for the evaluation of the efficacy of management policies for the Great Brak Estuary are:

- the management performance index for mouth condition,
- the management performance index for flushing, and
- the management performance index for stratification state.

No critical time period or duration of mouth deepening is prescribed, but a critical sill height of 2 m to MSL is specified in the determination of the mouth condition under a management policy. This figure was chosen to exceed 1.95 m to MSL, the level at which natural breaching was likely to have occurred, to ensure that the mouth condition index is positive under all policies so facilitating comparison of management performance indices. No critical tidal flushing rate, time period or duration is assigned for the determination of the flushing state under a management policy, nor is a critical stratification index, time period or duration specified. The long term average natural run-off scenario (the standard trajectory of Section 4.4.2) is considered a reasonable choice of reference policy, because the management performance indices will then measure the deviations of mouth condition, tidal flushing and stratification state from the average natural or undeveloped state of the estuary. The purpose in applying the Estuarine Systems Model to the Great Brak Estuary, which is to derive an understanding of the consequences of reduced fresh water flow to the sytem and to evaluate the efficacy of management policies in maintaining the character and functioning of the system, will thus be addressed. The management evaluation will be undertaken over the simulation time interval from years three to five, because the average natural run-off scenario exhibited a repetitive dynamic state over this time period. Thus the management evaluation sector of the Great Brak model comprises:

$$MC^P = 2.0 - x_7^P$$
$$SS^P = x_3^P$$
$$FS^P = x_6^P$$

$$PI_1 = \int_T MC^P\,dt \;/\; \int_T MC^R\,dt \quad \text{for } T = [3,5]$$

$$PI_3 = \int_T SS^P\,dt \;/\; \int_T SS^R\,dt \quad \text{for } T = [3,5]$$

$$PI_4 = \int_T FS^P\,dt \;/\; \int_T FS^R\,dt \quad \text{for } T = [3,5]$$

where MC^P = mouth condition under policy P (m)
x_7^P = sill height under policy P (m)
SS^P = stratification state under policy P (unitless)
x_3^P = stratification index under policy P (unitless)
FS^P = flushing state under policy P (yr^{-1})
x_6^P = tidal flushing under policy P (yr^{-1})
PI_1 = management performance index 1 (unitless)
T = time interval over which management performance is evaluated (yr)
MC^R = mouth condition under the reference policy R (m)
PI_3 = management performance index 3 (unitless)
SS^R = stratification state under the reference policy R (unitless)
PI_4 = management performance index 4 (unitless)
FS^R = flushing state under the reference policy R (yr^{-1})

These indices provide a means of quantitatively comparing the effects of different management policies on the mouth condition, tidal flushing and stratification state of the estuary in relation to the reference policy. All three indices are necessary to the evaluation of management policies for the Great Brak system because it is conceivable that two policies may yield the same management performance index for mouth condition, yet rate very differently in terms of tidal flushing and stratification state. For instance, a shallow mouth which is open for longer periods could achieve a management performance index equivalent to a deep mouth open for a shorter period, yet the tidal flushing in the latter case would probably far exceed that in the former case. Similarly, slight differences in the pattern of freshwater inflows may not cause substantial differences in the mouth condition or tidal flushing, but could manifest in significant differences in the stratification state owing to the non-linear nature of the response of the water column to freshwater flows. Thus the policies simultaneously maximising mouth condition and tidal flushing, while maintaining a stratification-circulation state characteristic of the Great Brak Estuary are considered most beneficial.

5.1.2 Management policies

The catchment of the small Great Brak Estuary underwent development from forestry, agriculture and human settlement to the extent that the mean annual run-off to the estuary was reduced from 34 x 10^6 m^3 (the natural MAR) to 24 x 10^6 m^3 prior to the closure of the Wolwedans Dam in 1989. The guaranteed allocation to the estuary subsequent to this event is 2 x 10^6 m^3 per annum, although an annual allocation of 1 x 10^6 m^3 has been mooted for the full demand situation anticipated after the year 2000. The management policies chosen for testing in the Great Brak case study must address this future situation of increased water demand in the Mossel Bay area as well as the effective use of the present allocation against the background of the effects that past development of the catchment have already had. Consequently, a management policy will comprise the following aspects in the case of the Great Brak Estuary:

- the volume of freshwater supplied to the estuary per annum or alternatively the percentage freshwater abstracted compared with the mean natural run-off to the system,
- the distribution of the annual freshwater volume over time, that is as base flows or flood-simulating water releases,
- a number of mouth closure events of varying duration and severity, and
- a breaching strategy including specification of the water level above which the mouth may be breached and whether this will be assisted by water releases or not.

Clearly, the term '*management policy*' has been interpreted in its broadest sense for the Great Brak case study and incorporates overarching decisions such as the freshwater allocation to the estuary as well as details of water releases and mouth breaching activities.

Eight management policies ranging from the undeveloped situation (100% of natural MAR) to that of the minimum assured allocation of 1 x 10^6 m^3 per annum (about 3% of natural MAR) were selected for testing to demonstrate the long term effects of increased freshwater abstraction and examine options for water releases and mouth breachings in the post-dam situation. These management policies are described below, while the effects of episodic events such as floods or high waves on the long term dynamic equilibria occasioned by the management policies are considered later (Section 5.1.4).

1. ***Natural run-off scenarios***

The mean annual run-off of the undeveloped catchment was 34 x 10^6 m^3 with the seasonal variation characteristic of the Mossel Bay region (Table 4.3b). Even in the natural situation the mouth of the estuary was known to close, although it rarely remained closed for longer than a month before breaching occurred. Breaching usually occurred naturally at a level of approximately 1.95 m to MSL or higher (CSIR 1990a), but was sometimes expedited by local residents. The most likely month for mouth closure was June, owing to lower than average inflows and the high wave conditions of winter. The management policies selected as representative of the natural run-off situation, therefore, comprise:

A the long term average situation of no freshwater abstraction in which the mouth remains open throughout (the standard trajectory), although modulation of the sill height occurs owing to seasonality in wave conditions, and

B no freshwater abstraction, but closure of the mouth in June (wave heights, WH, of 4,45 m for 3 days) and subsequent breaching at high water levels of 1.95 m to MSL after fifteen days at a breaching rate of -1870 $m.yr^{-1}$ for two hours.

2. ***Pre-dam run-off scenario***

The mean annual run-off to the estuary was 24 x 10^6 m^3 (Bruwer 1987). The major influence of catchment development was on low flows, so that the seasonality of the inflow altered slightly (Bruwer 1987) and the mouth of the estuary closed more frequently and for longer periods (CSIR 1990a). On average, the mouth closed between two and three times per year, but rarely remained closed for four or five months. The most probable times of mouth closure were June/July, May and December/January. The mouth no longer breached naturally, but was usually breached artificially by local residents at levels of 1.6 to 1.85 m to MSL. The management policy considered representative of the pre-dam run-off scenario, therefore, comprises the long term average situation of 29.4% freshwater abstraction with a mouth closure event in May (WH = 4,25 m for 3 days) and again in December (WH = 4,05 m for 3 days) and associated mechanical breachings in June and December, respectively (at levels of 1.85 and 1,62 m to MSL, respectively, and a breaching rate of -1820 $m.yr^{-1}$ for two hours).

3. ***Post-dam run-off scenarios***

According to Bruwer (1987), the long term average run-off to the estuary is 10 x 10^6 m^3 per annum. The majority of this inflow derives from the small area of catchment below the dam and overflow from the dam during wet periods. Clearly, the seasonality of the inflow differs somewhat from the natural situation and the mouth will close even more frequently and for longer periods. It was estimated that the mouth would be open 40% of the time on average, but could close for long periods if not managed appropriately (CSIR 1990a). During dry periods the assured supply to the estuary is only 2 x 10^6 $m^3.yr^{-1}$ until the year 2000 and possibly half of this thereafter. The management policies considered representative of the post-dam run-off situation therefore include:

A A freshwater inflow of 10 x 10^6 m^3 per annum distributed seasonally according to Bruwer

(1987). Four mouth closure events occur in the year during November/December, February, May/June and August/September (WH = 4,00 m for 3 days). Mechanical breaching is undertaken at water levels of 1.62 m to MSL at a rate of -1820 $m.yr^{-1}$ for two hours.

B A continuous base flow amounting to 2 x 10^6 m^3 per annum and incorporating seasonality, is included to represent the dry period situation to the year 2000. Four mouth closure events occur in the year during November/December, February, May/June and August/September (WH = 4,00 m for 3 days). Mechanical breaching is undertaken four times (in accord with scenario 5A) to ensure that the estuary remains open for a reasonable proportion of the year.

C Three freshwater releases totalling 1.5 x 10^6 m^3 per annum timed to co-incide with mouth breachings during November/December, February and August/September (following mouth closure events associated with WH = 4,00 m for 3 days), and base flow totalling 0.5 x 10^6 m^3 per annum. An additional mouth breaching unassisted by a water release occurs in June. This policy is considered representative of the present management of the estuary during a dry period (CSIR 1990a, CSIR 1994b).

D A continuous base flow amounting to 1 x 10^6 m^3 per annum incorporating seasonality is included to represent a possible dry period after the year 2000. Four mouth closure events occur in the year during November/December, February, May/June and August/September (WH = 4,00 m for 3 days). Mechanical breaching is undertaken four times (in accord with scenario 5A) to ensure that the estuary remains open for a reasonable proportion of the year.

E Three freshwater releases totalling 1 x 10^6 m^3 per annum timed to co-incide with the November/December, February and August/September mouth breachings, and no base flow. An additional mouth breaching occurs during May/June, but this is purely mechanical and is unassisted by a water release. This policy represents possible management of the estuary during a dry period under the full water demand situation after the year 2000 (CSIR 1990a, CSIR 1994b).

While these management policies are by no means inclusive of all management options for the Great Brak Estuary, they are considered to provide a realistic background against which the average long term effects of freshwater abstraction can be assessed (Mr P Huizinga *pers. comm.*). Alternative water management strategies and associated mouth breaching options are assessed for the present and possible future situations because information can usefully contribute to current decision-making, but are omitted for the natural and pre-dam situations.

5.1.3 **Simulation results**

Natural run-off scenarios

The exogenously specified inflow functions, which comprise the upstream boundary conditions for the natural run-off scenarios 1A and 1B, are depicted in Figure 5.1a. The seasonality in rainfall on the southern Cape coast means that the run-off to the Great Brak Estuary is least during the month of June (about 50% of the MAR of 34 x 10^6 $m^3.yr^{-1}$), while peaks in flow occur in the higher rainfall months of September, November and March (between 133% and 151% of the MAR). The most likely month for flooding is October, even though the average flow is somewhat less than in September and November. However, the effects of floods will not be addressed at this stage, rather the implications for the estuary of the long term average natural run-off situations, as exemplified by the repetitive dynamic equilibrium attained in simulation years three to five, will be explored.

The simulated response of the Great Brak Estuary to the natural run-off scenario 1A is descibed briefly in Section 4.4.2 and depicted graphically in Figures 4.3p and 4.4a for a five year period. The response of the estuary to the natural run-off scenario 1B for years three to five of the simulation period is described below and depicted in Figure 5.1b.

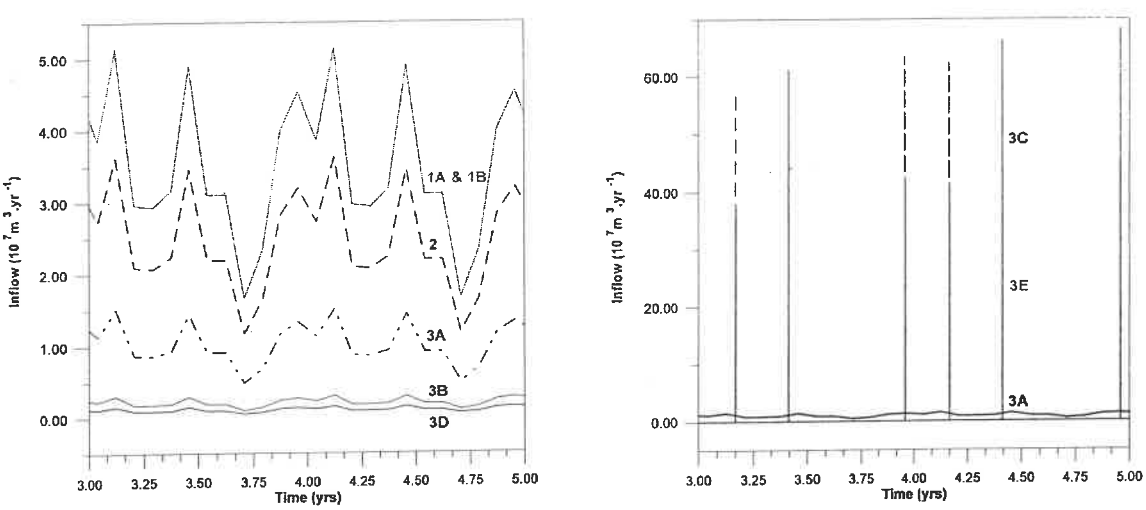

Figure 5.1a Inflow to the Great Brak Estuary under the different management policies

Under scenario 1A, the sill height varied from a maximum of 0.61 m to MSL in summer to a minimum of 0.59 m to MSL in winter in response to the seasonal variation in wave height, which comprised the downstream boundary condition of scenario 1A as no extreme tides or high wave events were included. In contrast, the sill height under scenario 1B remained at a level of approximately 0.6 m to MSL for the majority of the time, but rose to about 1.95 m to MSL when the mouth closed during the low flow month of June (WH = 4,45 m for 3 days; an additional downstream boundary condition compared with scenario 1A). Simulated water levels in the estuary varied from about 0,66 m to MSL to 0,94 m to MSL when the mouth was open, but gradually increased over 15 days after the mouth closed in June. Once a level of 1.95 m to MSL was attained, breaching of the mouth occurred (breaching rate -1870 $m.yr^{-1}$ for two hours) and water levels again varied as in the tidal, open mouth phase since the sill height returned to very near its original level. Water levels in the estuary corresponded closely under scenarios 1A and 1B apart from during the mouth closure event. In both cases, a strong spring-neap effect was evident with tidal variation occurring in the estuary water levels during spring tide, but not during neap tides when high tide levels at sea did not exceed the sill height at the mouth and so tidal penetration of the estuary did not occur. Thus over neap tides, the mouth of the estuary was open, yet closed to tidal influence. The effect of higher freshwater flows was also evident in the low water levels in the estuarine basin. Elevated low water levels occurred during the months of November, March and September when more freshwater flow entered the system than during the other months of the year. Under scenario 1A, the lowest low water levels in the estuary occurred in June owing to the combined effect of low freshwater flows and limited tidal influence due to seasonal shallowing of the sill at the mouth. However, under scenario 1B, the low water levels achieved a minimum in January, the month with the lowest freshwater inflows apart from June when the mouth was closed and the effect of the seasonal shallowing of the mouth of the estuary was overridden by the closure event.

Under both scenarios 1A and 1B, spring tidal fluxes through the mouth varied between about -9 $m^3.s^{-1}$ on the ebb and 12 $m^3.s^{-1}$ on the flood (Figures 4.4a & 5.1b). The higher magnitude of the flood tidal fluxes reflects the flood tidal dominance of this small, shallow estuary even under natural run-off conditions. Over neap tides no penetration of sea water occurred, only a slow drainage of estuarine water through the shallow mouth. No tidal exchange occurred when the mouth was closed under scenario 1B and mean salinities decreased from about 22.5 $kg.m^{-3}$ at the time of mouth closure to less than 2.5 $kg.m^{-3}$ immediately prior to mouth breaching, since only fresh water entered the system. When the mouth opened, strong tidal intrusion occurred and mean salinities attained an overall maximum of 25 $kg.m^{-3}$. Mean salinities generally exhibited maxima of between 18 and 25 $kg.m^{-3}$ over spring tides and decreased to near zero following the drainage of water on the neap tides. The maximum mean salinities increased during the months with slightly lower fresh water flows eg. January. This trend was very evident under scenario 1A when a maximum mean salinity of 26 $kg.m^{-3}$ occurred during the low flow month of June and even the minimum mean salinity exceeded 4.5 kg.m^{-3} at this time. However, axial salinity differences remained at 35 kg.m^{-3} throughout the simulation period for both scenarios 1A and 1B, owing to the continuous strong base flow.

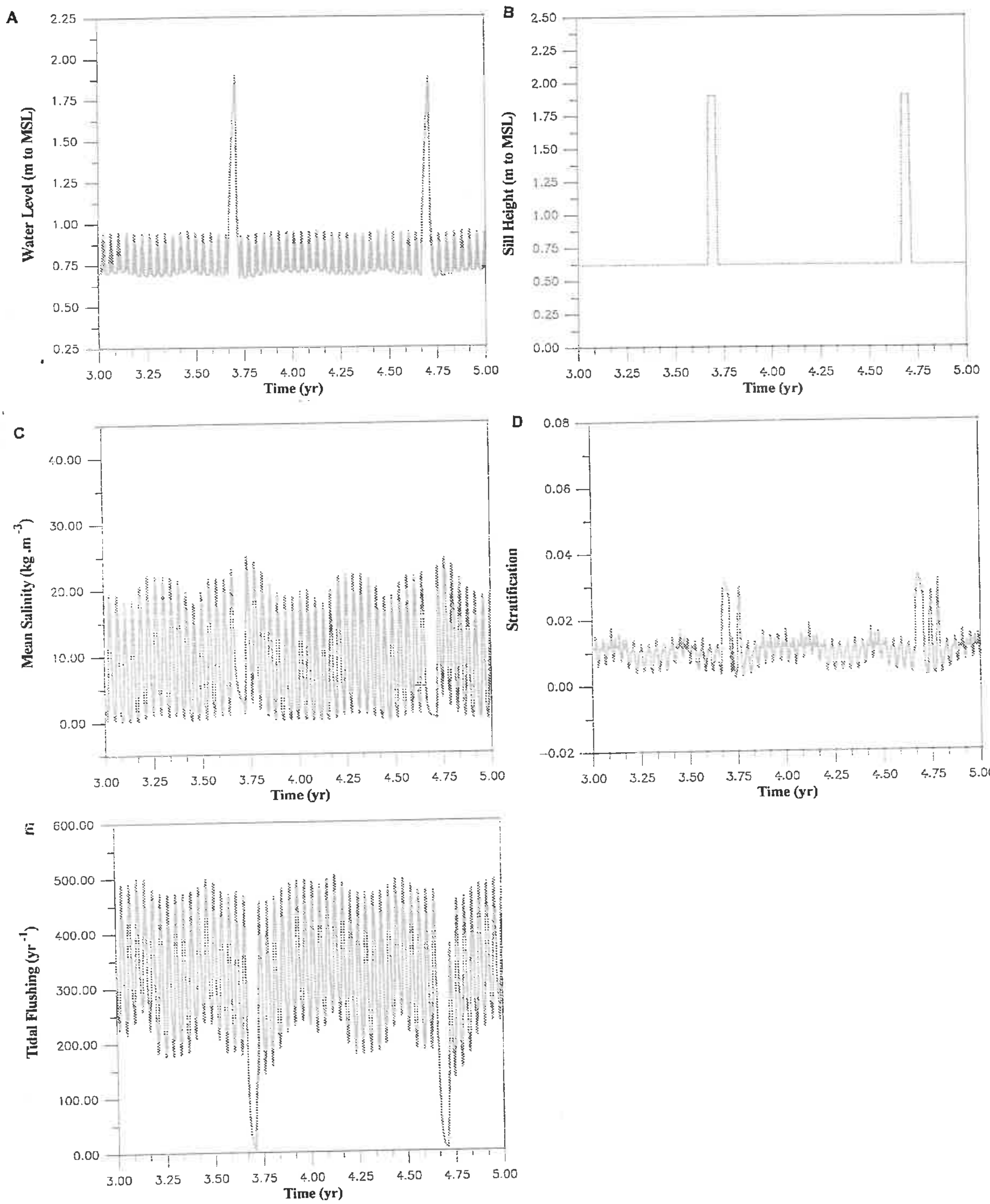

Figure 5.1b The simulated water levels (A), sill heights (B), mean salinities (C), stratification (D) and tidal flushing (E) of the Great Brak Estuary from simulation years three to five under the natural run-off scenario 1B.

The values of the stratification index varied from 0.004 to about 0.03 under both scenarios 1A and 1B and the circulation index varied from 0.02 to 0.06, indicating that the Great Brak Estuary was partially mixed under the natural run-off scenario (refer to Fig 3.4a). The stratification index exhibited sensitivity to both freshwater inflows and tidal influences. This is evident from the increase in the monthly mean values with stronger freshwater inflows (eg. during November and March under scenarios 1A and 1B) and the enhanced variation which accompanied the transition from spring to neap tide (reduced tidal influence) during low freshwater inflows (eg. during June under scenario 1A). The spring-neap tidal variation exerted substantial influence throughout the year with bi-monthly maxima and minima in the stratification index occurring nearly four days after neap and spring tides, respectively. The more vigorous tidal mixing associated with spring tide acted to de-stratify the water column, whereas the neap tide was ineffective in this sense as little or no tidal influence was felt in the estuary at this time. Consequently, the inflowing freshwater acted to intensify stratification of the water column over neap tides. The role of freshwater in intensifying stratification was most evident when the mouth closed in June under scenario 1B and the estuary gradually filled with freshwater over the more saline bottom water. This was well reflected by the increase in the stratification index from below 0.02 to just above 0.03. Following the breaching of the mouth, the stratification index still exhibited values of this order after neap tides indicating the effects of freshwater still lingering in the system. Thereafter, the stratification index decreased below 0.02, exhibiting values characteristic of enhanced tidal influence and low freshwater flows. Thus the estuary remained in a partially mixed state throughout.

The freshwater flushing under scenarios 1A and 1B corresponded exactly as this state variable is dependent on freshwater flows to the system. The annual maximum of 11.5 in late November and the peaks late in March and September reflect the accumulated influence of sustained slightly higher freshwater flows, while the annual minimum of 3.5 in late June/early July is indicative of low inflows over the preceding month (Fig 4.4a). The tidal flushing, on the other hand, differs from scenario 1A to 1B, because tidal flushing is substantially reduced during the period of mouth closure under scenario 1B. In this case, the tidal flushing variable exhibits a near zero minimum in June and a maximum of about 500 yr^{-1} in November and March when the mouth is open and higher freshwater flows occur. In contrast, scenario 1A exhibits a minimum of about 150 yr^{-1} in June.

The management performance index for flushing for the natural run-off scenario 1B reflects this difference as it attains a value of 0.963 (as opposed to unity for the natural run-off scenario 1A - the reference scenario). The management performance index for the mouth condition is 0.958 for scenario 1B. However, the management performance index for stratification state attains a value of 1.060, reflecting the effect that a mouth closure event exerts on the stratification state of the estuary.

Pre-dam run-off scenario

The externally determined inflow for the pre-dam run-off scenario comprises a total annual discharge of 24 x 10^6 m^3 following a seasonal pattern typical of the southern Cape, but with a reduced amplitude of variation compared with the natural run-off scenarios (Fig 5.1a). The simulated response of the estuary to this run-off (about 70% of the average natural situation) and the management strategy of breaching the mouth at water levels of 1.62 to 1.82 m to MSL is depicted in Figure 5.1c for years three to five of the simulation period and described below.

The mouth of the estuary closed twice per annum in May and December (additional downstream boundary conditions; WH = 4,25 m and 4,05 m, respectively, for 3 days). Following breaching of the mouth at a level of 1.62 m to MSL in December and a level of 1.85 m to MSL in June, the sill height at the mouth of the estuary decreased to approximately 0.6 m to MSL in both cases. Because no high freshwater flows, extreme tidal events or other wave events were specified, the sill height remained at about this level for the rest of the year.

When the mouth was open, simulated water levels in the estuary varied from a minimum of about 0,67 m to MSL over neap tides to a maximum of 0,94 m to MSL over spring tides. Super elevation of the low water levels was a noticeable feature of the Great Brak Estuary over spring tides, whereas the slow outflow of water through the constricted mouth ensured that low water levels declined progressively during neap tides. When the mouth closed in December, tidal action ceased and water levels gradually increased to 1.63 m to MSL. The mouth was breached ten days after closure, reflecting the common practice of residents owing to concerns for recreation, inundation of property and vegetation (CSIR 1990a). In contrast, when the mouth closed late in May (the off-season for recreation and a time of reduced vegetation growth), water levels rose to 1.82 m to MSL and remained at this level or just below for 15 days before breaching was undertaken. During both mouth closure events, mean salinities in the system decreased to below 2 $kg.m^{-3}$, but rose to maxima of 27 $kg.m^{-3}$ on the spring tide subsequent to mouth breaching. The spring-neap tidal variation exerted a strong influence on salinities when the mouth was open, with bi-monthly maxima reflecting the intrusion of highly saline water over spring tide and the bi-monthly minima representing the continuous inflow of freshwater and slow drainage of estuarine water over neap tides. The axial salinity gradient remained 35 $kg.m^{-3}$ throughout the simulation period.

The stratification and circulation indices varied from 0.003 to 0.032 and 0.003 to 0.044, respectively, indicating that the Great Brak Estuary was partially mixed under the pre-dam run-off scenario. However, in comparison with the natural run-off scenarios, the stratification index exhibited increased sensitivity to tidal influence during periods of reduced freshwater flow. This is particularly noticeable in April/May, prior to closure of the estuary mouth, when the stratification index varied from minima of 0.003 after spring tides to maxima of 0.031 after neap tides. Similar variations occur following mouth breachings, but these are ascribed to a combination of two effects. First, the stratifying effect of freshwater entering the closed system has not dissipated entirely, but lingers on after the mouth is breached and second, the lower freshwater inflows in the months of January/February and July/August mean that the sensitivity to tidal influence is enhanced. Thus, the variability in the stratification state of the estuary under the pre-dam run-off scenario increases over periods of lower flow compared with the natural run-off scenario, while the monthly average value decreases over periods of higher flow eg. November, March and September. The circulation index also showed a general decrease in value compared with the average natural situation, particularly during the closed mouth periods. This stratification-circulation response indicates that the buffering capacity of the water body to ameliorate changes in the riverine and marine forcing to the estuary is reduced compared with the natural run-off scenarios.

The freshwater flushing of the system, which is dependent primarily on freshwater inflow, is substantially reduced compared with the natural run-off scenarios and shows less variability, exhibiting a minimum of about 3 and a maximum of 8. The tidal flushing of the estuary is also reduced in comparison with the natural run-off scenario 1B, attaining a maximum of about 475 yr^{-1} in November and March compared with 500 yr^{-1} and a minimum near zero twice a year when the mouth is closed compared with once a year. This is borne out by the management performance index for flushing which assumes a value of 0.808 compared with the value of 0.963 for scenario 1B and unity for scenario 1A. The management performance index for mouth condition similarly reflects the increased frequency and duration of mouth closure compared with the natural run-off scenarios by decreasing to 0.901. In contrast, the stratifying influence of freshwater entering the estuary when the mouth is closed and the enhanced sensitivity of the stratification state to alterations in freshwater flow and tidal forcing are reflected in an increase in the management performance index for stratification state, which attains a value of 1.181.

Post-dam run-off scenarios

The exogenous inflow functions for the post-dam run-off scenarios 3A, 3B and 3D are specified as continuous base flows with the seasonal variation of the southern Cape coast (Fig 5.1a). However, a large proportion of the inflow delivered to the estuary under the post-dam run-off scenarios 3C and 3E arises from three water releases from the Wolwedans Dam per annum. These are specified using flood functions with start times of 29 November, 27 February and 15 September, annual repeat intervals, ten hour durations and total volumes of 5×10^5 m^3 and 3.3×10^5 m^3 for scenarios 3C and 3E, respectively. The responses of the Great Brak Estuary to the post-dam run-off scenarios are described below and depicted graphically in Figures 5.1d to 5.1h for the simulation years three to five.

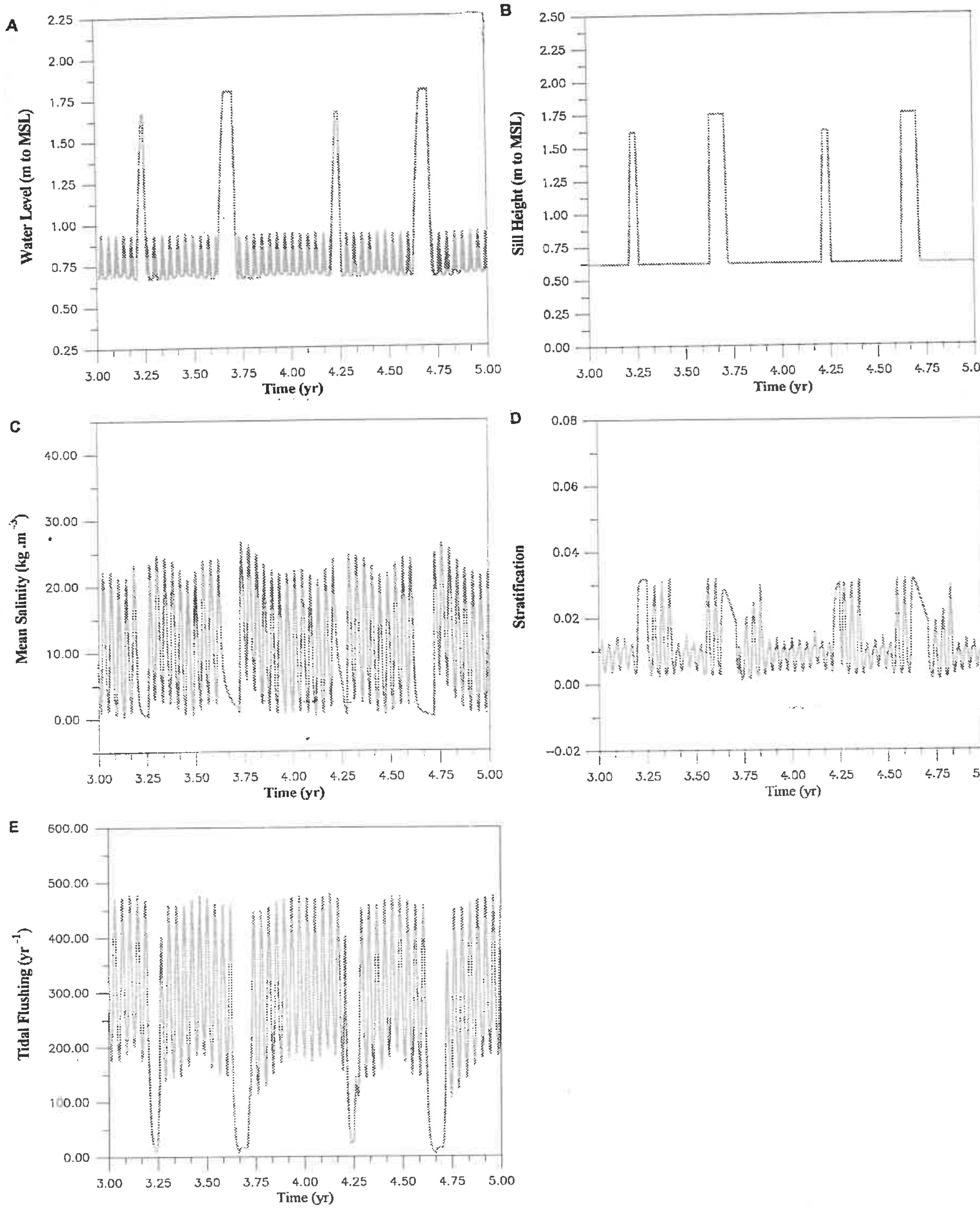

Figure 5.1c The simulated water levels (A), sill heights (B), mean salinities (C), stratification (D) and tidal flushing (E) of the Great Brak Estuary from simulation years three to five under the pre-dam run-off scenario 2

In each of the post-dam run-off scenarios, the mouth of the estuary closed four times per year in early November and February, in late April and in mid July (WH = 4,00 m for 3 days in each case). The sill height attained values in excess of 1.6 m to MSL at these times and remained at a high level until artificial breaching of the mouth was undertaken. A breaching rate of -1820 m.yr^{-1} was applied for 3,5 hours when purely mechanical breaching occurred and for 2 hours when a breaching was assisted by a water release eg. in Scenarios 3C and 3E. During the subsequent tidal phases, the sill height minima attained were very close to 0.6 m to MSL for all the post-dam run-off scenarios. The water levels in the estuary consequently exhibited very similar behaviour for all the scenarios when the mouth was open, ranging from a minimum of 0.67 m to MSL over neap tides to a maximum of 0.92 m to MSL over spring tides. However, marked differences occurred when the mouth was closed. Under the post-dam run-off scenario 3A, the water level rose rapidly to 1.65 m to MSL and remained at this level with slow outflow until the mouth was breached mechanically. Note that the height of the berm was such that an unstable outflow channel could form, but the inflow to the estuary was insufficient to cause the mouth to breach naturally. Under scenarios 3B and 3D, the water levels in the estuary increased gradually after the mouth closed, attaining overall maxima of 1.37 m to MSL and 1.12 m to MSL, respectively. In each case, the inflows to the system were insufficient to cause extensive inundation of the salt marsh areas, overflow of the berm, or natural breaching. In contrast, the water releases of scenarios 3C and 3E caused water levels to rise to 1.85 m to MSL and 1.50 m to MSL, respectively, on occasions. Thus the necessary inundation of salt marshes occurred, but mouth breaching was undertaken before submergence could cause die back of the vegetation. Differences are discernible in the water levels attained in the simulation years three and four during the mouth closure events for the scenarios 3B, 3C, 3D and 3E. These arise because mouth closures recur at the same time every year, but the spring-neap tidal cycle is slightly out of phase with the calendar year. Thus in February of simulation year three, the mouth closes on a spring tide, but mouth closure occurs on a neap tide in February of the following year and the maximum water levels attained differ. Under the higher freshwater flows of scenario 3A these effects exert very little influence on water levels. Despite water level differences between scenarios and between simulation years, the tidal fluxes under all of the post-dam run-off scenarios are very similar, ranging from a flood tidal maximum of about 12 m^3.s^{-1} to an ebb tidal minimum of -8 m^3.s^{-1}.

Under scenario 3A, the mean salinities in the estuary exhibited maxima of 28 kg.m^{-3} over spring tide when the mouth was open and minima of between 6 and 12.5 kg.m^{-3}, depending on the freshwater inflow at the time. Salinities generally decreased to less than 3 kg.m^{-3} when the mouth was closed. The exception occurred in February of simulation year three as mouth closure commenced at spring tide when salinities were high and was not prolonged. Under the reduced freshwater flows of scenarios 3B and 3D, mean salinities were significantly higher, achieving maxima of about 32 and 33 kg.m^{-3}, respectively, over spring tide when the mouth was open and minima of about 24 and 29 kg.m^{-3}, respectively, over neap tides. Additionally, the limited freshwater flows caused mean salinities to rarely decline below 15 and 20 kg.m^{-3} under scenarios 3B and 3D, respectively, when the mouth was closed. In contrast, the freshwater releases of scenario 3C, timed to co-incide with periods of mouth closure, cause the mean salinities to decrease below 10 $kg.m^{-3}$ three times per annum. However, during tidal phases the mean salinities in the estuary are generally higher than at the corresponding time under scenario 3B, owing to lower base flows. Similarly, under run-off scenario 3E the mean salinities decrease below 15 $kg.m^{-3}$ three times per annum, but are higher when the mouth is open than at the corresponding time under scenario 3D. These effects are very clear when one considers the axial salinity differences arising from the various management scenarios. Under scenario 3A, the freshwater inflow is sufficient to maintain an axial salinity gradient of 35 $kg.m^{-3}$ for the majority of the year. Only on spring tides at times of reduced freshwater inflow eg. December/January, April and July, do the head to mouth differences in salinity decrease below 20 $kg.m^{-3}$. Under scenario 3B, however, the head to mouth salinity differences never exceed 20 $kg.m^{-3}$ and under scenario 3D, they are always below 10 $kg.m^{-3}$. However, the freshwater releases of scenarios 3C and 3E cause the head to mouth salinity differences to rise to 35 $kg.m^{-3}$ three times per annum from levels of less than 5 and 1 $kg.m^{-3}$, respectively, throughout the rest of the year. Thus the management strategy of flood releases as opposed to continuous base flows is very effective in enhancing salinity gradients in the Great Brak Estuary given a limited allocation of freshwater.

Comparison of the stratification-circulation states of the Great Brak Estuary under scenarios 3A, 3B and 3D indicates that when the limited freshwater allocation is delivered to the estuary as continuous base flow, there is a strong reduction in the amplitudes of variation and the average values of both the stratification and circulation indices with increasing freshwater abstraction. Under scenarios 3A, 3B and 3D, the maxima exhibited by the circulation index decline from 0.026 to 0.012 and 0.007, respectively, while the stratification indices attained maxima of 0.024, 0.0035 and 0.002, respectively. However, when flood releases are included as a management strategy as in scenarios 3C and 3E, the stratification indices tri-annually attain values in excess of 0.07 and 0.035, respectively, whereas the circulation indices tri-annually exceed 0.065 and 0.06, respectively. Between flood events, the stratification in the system is minimal (stratification index values less than 0.005) and the circulation index is always less than 0.005. These figures indicate that the estuary remained partially mixed throughout and the flood events of scenarios 3C and 3E were not sufficient to thrust the Great Brak Estuary into a highly stratified state. However, the contribution of the density-driven circulation to the landward transport of salt did increase during the flood events (ν decreased in Fig 4.3a). Thus, the management strategy of water releases as opposed to continuous base flow releases, causes the estuary to exhibit partially mixed states more typical of the natural run-off scenario interspersed with periods in which the estuary remains partially mixed but the water column is fairly uniform. The stratification-circulation state under comparable total annual allocations released as base flows, is also partially mixed but with less variability and more stratification of the water column than during the intervals between floods. Thus the management strategy of flood releases intensifies stratification at times, but these effects are short-lived.

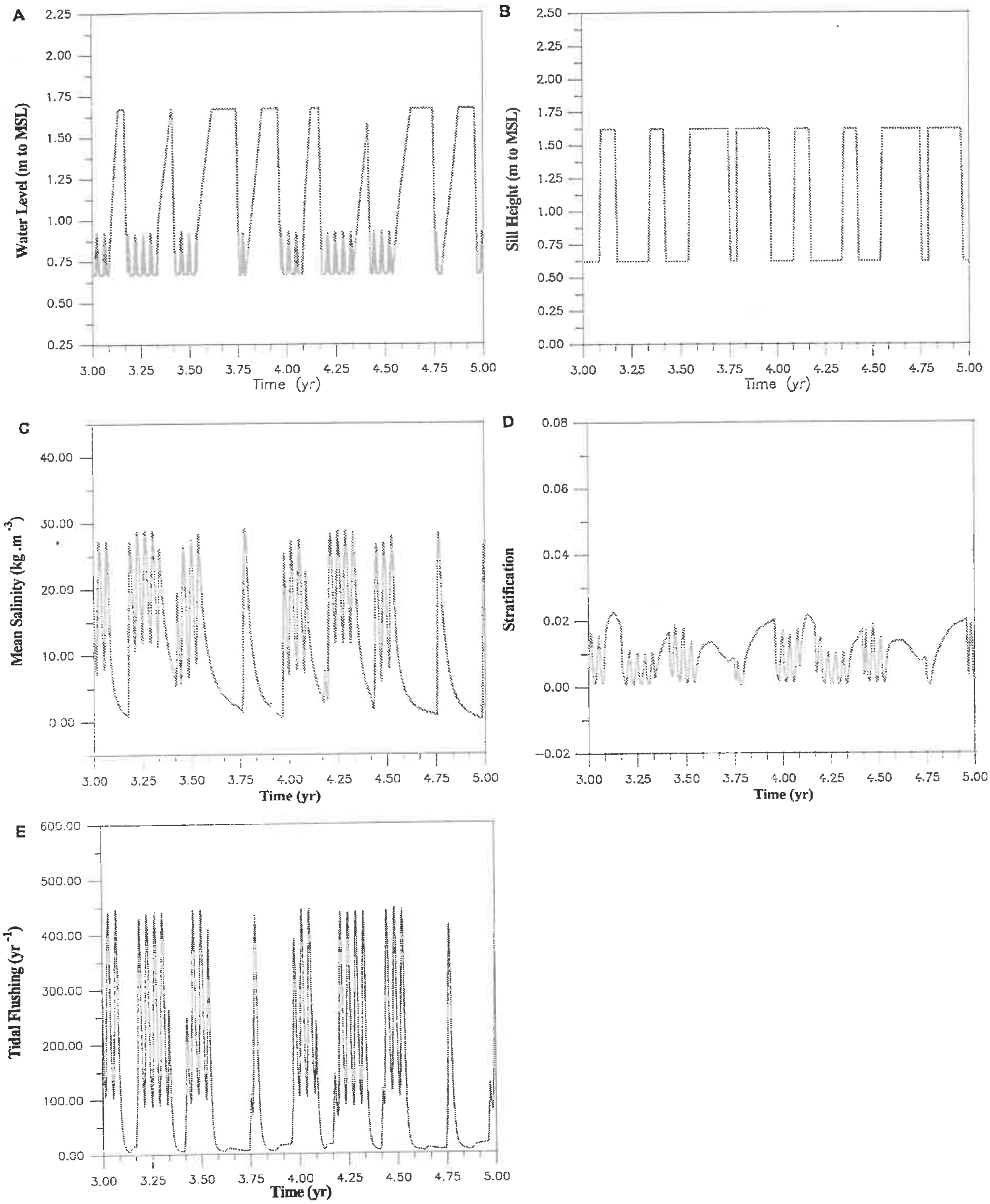

Figure 5.1d The simulated water levels (A), sill heights (B), mean salinities (C), stratification (D) and tidal flushing (E) of the Great Brak Estuary from simulation years three to five under the post-dam run-off scenario 3A.

The freshwater flushing under scenario 3A exhibits a substantial reduction in the amplitude of variation and the average value compared with the pre-dam run-off scenario and the natural run-off scenario. This trend of sharp decline with decreasing freshwater supply is even more marked when one compares the freshwater flushing under scenarios 3A, 3B and 3D. Scenarios 3B and 3D exhibit average values approximately 20% and 10%, respectively, of those under scenario 3A.

These effects are significant because they indicate a severe reduction in flushing of older water from the Great Brak Estuary by scouring owing to high velocity freshwater flows. The management scenario of water releases, however, attempts to artificially introduce these effects. Under scenarios 3C, the freshwater flushing tri-annually attains the levels characteristic of the pre-dam run-off scenario, but exhibits little freshwater flushing at times other than during and immediately after a water release. Under scenario 3E, the freshwater flushing tri-annually attains values in excess of those of scenario 3A and equivalent to the times of low freshwater influence under scenario 2 eg. during January, June and July. Clearly, the freshwater flushing of the estuary is highly impacted by the reduction in freshwater flow to the system and only during the water releases of scenario 3C does anything approaching the pre-dam situation recur.

The tidal flushing of the Great Brak Estuary under scenario 3A exhibits maxima of about 450 yr^{-1} (compared with 475 yr^{-1} under scenario 2) and minima of zero during periods of mouth closure. During tidal phases the tidal flushing minima are of the order of 100 yr^{-1}. Thus there is a substantial reduction in the efficacy of tidal flushing between the pre-dam situation and the post dam run-off scenario 3A. Scenarios 3B, 3C, 3D and 3E bear this out, exhibiting maxima of 430 yr^{-1} and minima of 50 yr^{1} when the mouth is open. Thus the tidal flushing of the Great Brak Estuary is severely impacted by the reduction in freshwater flow to the system and water releases do not appear to exert an ameliorating effect.

These effects are summarised by the management performance indices listed in Table 5.1a. Clearly, there is a substantial reduction in mouth condition and tidal flushing associated with the reduction in freshwater flow from the pre-dam to the post-dam situation. The management performance indices for mouth condition decline by 33% compared with the pre-dam value, while the management performance indices for flushing decrease by a minimum of 55% from the pre-dam value, which is in itself is a 20% reduction from that of the reference natural situation. However, the most remarkable effects are manifested by the management performance indices for stratification state. Although the post-dam run-off scenario 3A exhibits little change from the natural situation (less than 6% variation), the post-dam run-off scenarios 3B, 3C, 3D and 3E exhibit reductions of between 83% and 95% in their management performance indices for stratification state compared with the pre-dam value. Thus the management performance indices for mouth condition and tidal flushing indicate the effects of various freshwater abstraction levels on the character and functioning of the estuary, but are not particularly sensitive to differences in the management of the total annual allocations for a given abstraction level. In contrast, the stratification state provides a valuable and sensitive indication of the efficacy of different uses of the freshwater allocation at high levels of freshwater abstraction. Thus the three management performance indices provide a complementary suite of values to assist effective decision making at the national water authority level as well as at the local or regional water management level. An amalgamation of these indices into one index would involve the loss of information about the aspects of the estuary identified as relevant in its management by both the public and authorities of the town of Great Brak River and was thus avoided.

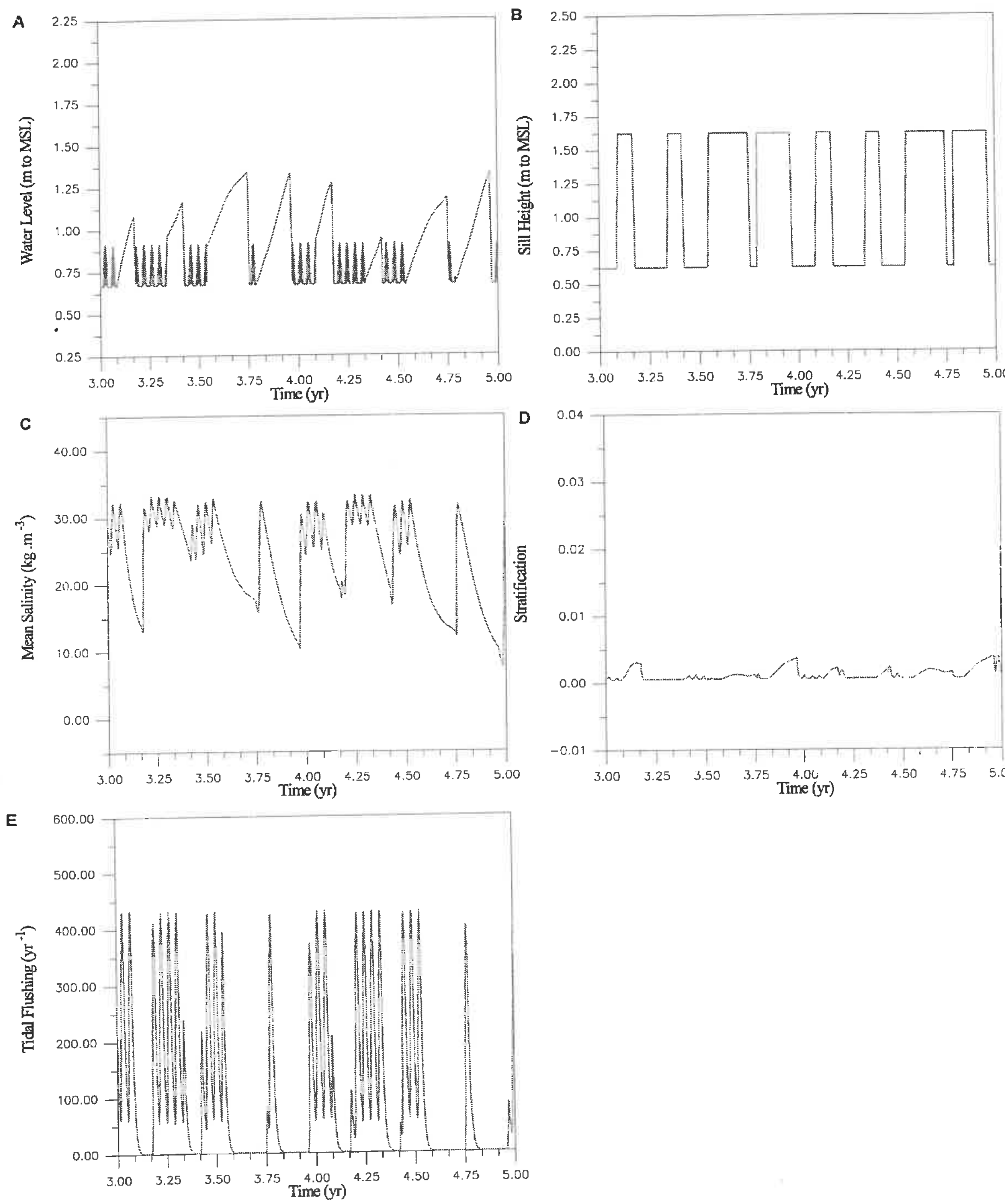

Figure 5.1e The simulated water levels (A), sill heights (B), mean salinities (C), stratification (D) and tidal flushing (E) of the Great Brak Estuary from simulation years three to five under the post-dam run-off scenario 3B.

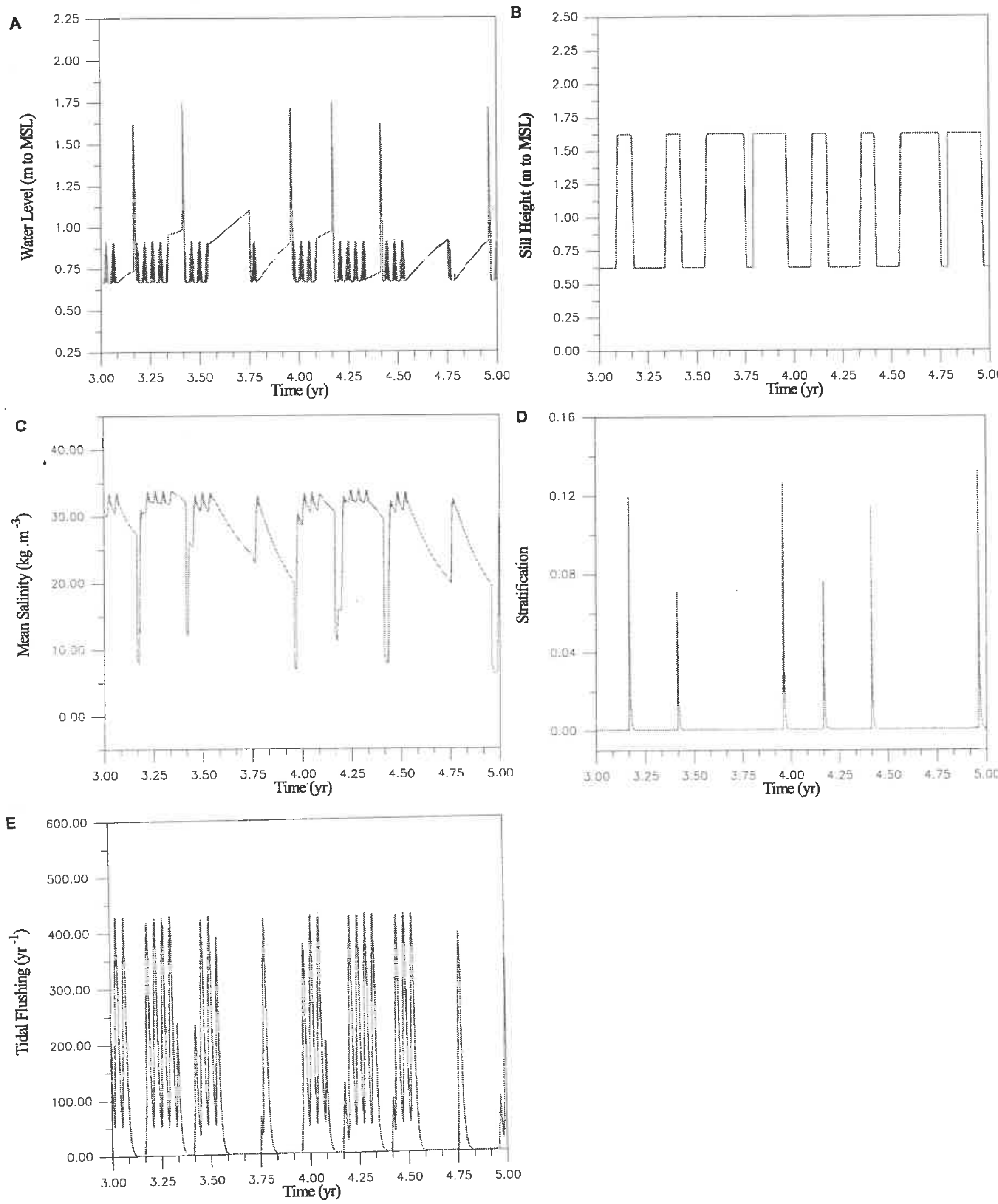

Figure 5.1f The simulated water levels (A), sill heights (B), mean salinities (C), stratification (D) and tidal flushing (E) of the Great Brak Estuary from simulation years three to five under the post-dam run-off scenario 3C.

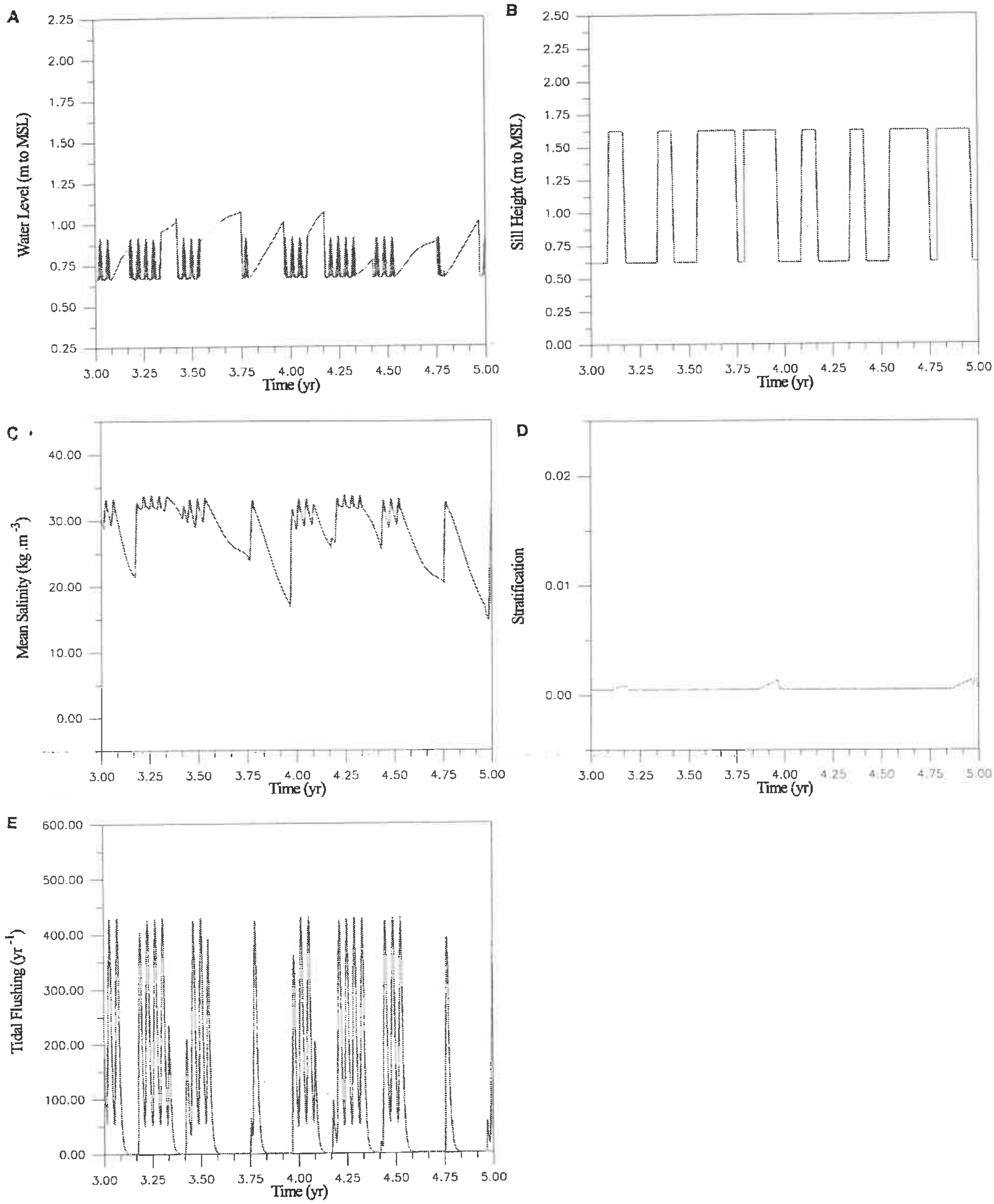

Figure 5.1g The simulated water levels (A), sill heights (B), mean salinities (C), stratification (D) and tidal flushing (E) of the Great Brak Estuary from simulation years three to five under the post-dam run-off scenario 3D.

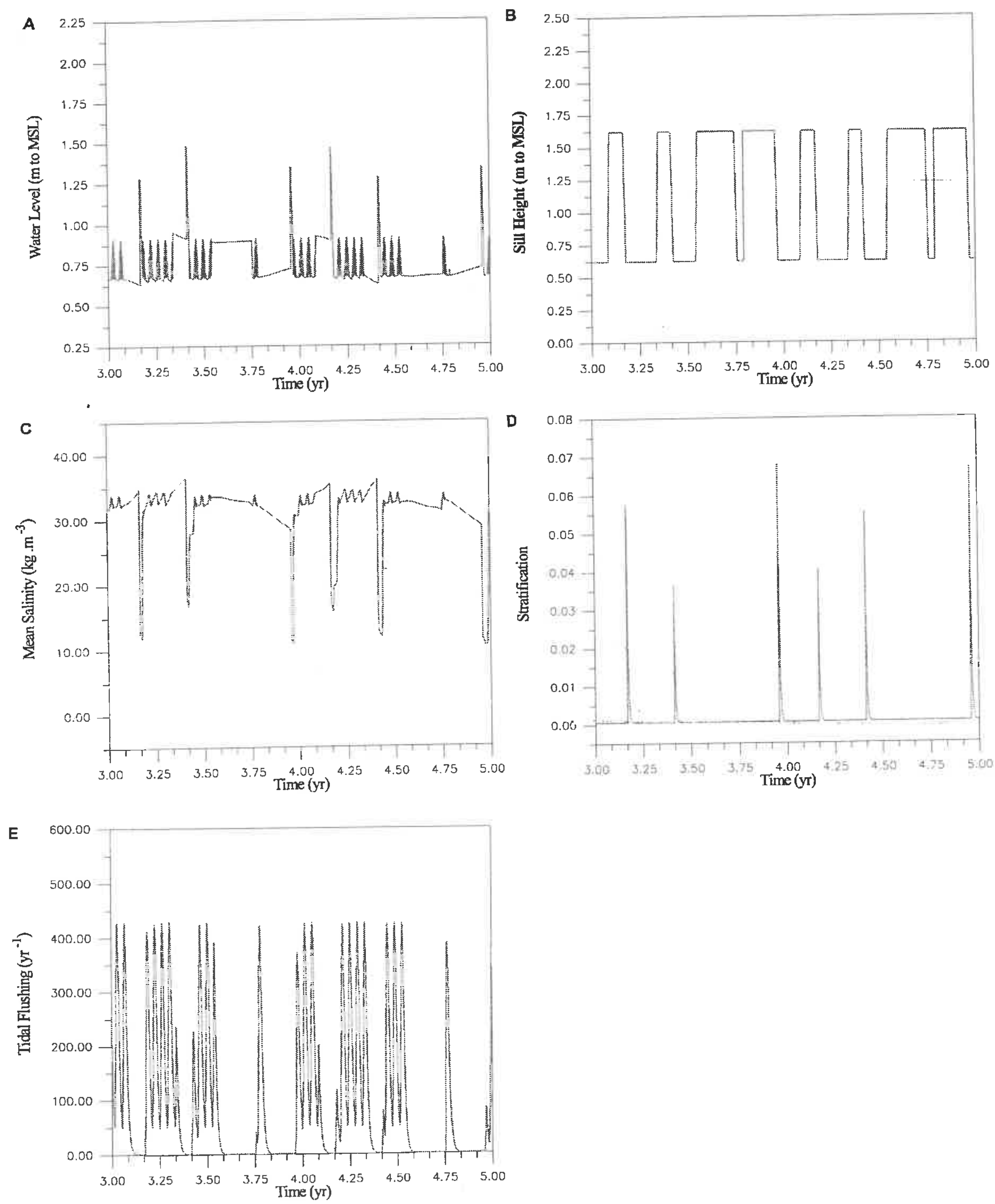

Figure 5.1h The simulated water levels (A), sill heights (B), mean salinities (C), stratification (D) and tidal flushing (E) of the Great Brak Estuary from simulation years three to five under the post-dam run-off scenario 3E.

From these simulations, it is clear that the construction of the Wolwedans Dam has had a major impact on the Great Brak Estuary. First, the mouth closes more frequently and for longer periods than under the pre-dam or natural situations, higher water levels are consequently experienced more regularly, the mean salinities in the system are generally higher during open mouth phases, the axial salinity differences are lower during dry periods and may not exceed 20 $kg.m^{-3}$ as would almost always have been the case prior to dam construction, the stratification-circulation state is still partially mixed, but vertical differences in salinity have decreased, the flushing of the system by freshwater is substantially reduced and tidal flushing is severely affected. Some amelioration of these effects may be achieved by imposing a policy of intermittent freshwater releases as opposed to continuous base flows, but the major effect of these flood-simulating releases is to imitate the pre-dam and natural run-off situations in the structure of the water column for brief periods. Thereafter, the estuary returns to the semi-permanent, fairly uniform state of little top to bottom difference in salinity and little density-driven landward transport of salt.

Table 5.1a Values of the management performance indices (relative to the reference policy - scenario 1A) for each of the run-off scenarios applied to the Great Brak Estuary

RUN-OFF SCENARIOS	MANAGEMENT PERFORMANCE INDICES		
	Mouth Condition	Flushing	Stratification State
1B	0.958	0.963	1.060
2	0.901	0.808	1.181
3A	0.610	0.367	0.999
3B	0.610	0.312	0.115
3C	0.610	0.310	0.199
3D	0.610	0.303	0.055
3E	0.610	0.305	0.124

5.1.4 **Episodic Events**

While the run-off scenarios simulated thus far provide an indication of the long term dynamic equilibrium states of the estuary in response to various freshwater abstraction rates, mouth closures and management stategies encompassing water releases and mouth breachings, they do not specifically address the response or sensitivity of the system to episodic events. For instance, how would a large flood affect the system and how rapidly would the pre-flood situation re-establish itself, if ever? Is the vulnerability of the estuary mouth to closure during high wave events enhanced by reductions in the freshwater flow and to what extent? In order to answer these and other similar questions, the dynamic behaviour of the estuary under the the different management policies was investigated. Insights into this dynamic behaviour were obtained by examining the effects of different episodic events or disturbances, namely: high waves and a large flood.

High wave events

The effects of high wave events of varying magnitude on the mouth condition of the Great Brak Estuary have been alluded to in the preceding discussion of the simulation results. For instance, the closure and subsequent breaching of the mouth under the natural run-off scenario 1B was associated with WH = 4,45 m for 3 days, whereas the mouth closed with WH = 4,05 m and 4,25 m under the pre-dam run-off scenario and WH = 4,00 m under the post-dam run-off scenarios. In each case different maximum sill heights were attained. These ranged from 1,9 m to MSL under scenario 1B to 1,62 and 1,85 m to MSL in Scenario 2 and 1,62 m to MSL in the post-dam run-off situations. Clearly, the change in the height of the sill at the mouth of the Great Brak Estuary differs in response to different wave events and freshwater flow rates. The behaviour of the estuary mouth and its sensitivity to closure from the enhanced sediment suspended by high waves and subsequently deposited in the mouth channel were investigated for seven wave events under freshwater flow rates characteristic of the natural, pre-dam and post-dam situations.

Constant freshwater inflow rates of 34×10^6, 24×10^6, 10×10^6, 2×10^6 and 1×10^6 $m^3.yr^{-1}$ were applied and a dynamic equilibrium state (with sill heights approximately 0,6 m to MSL) was attained within the first five years of the simulation period in each case. The high wave events commenced on 1 June in the seventh year of each simulation and persisted for three days. The behaviour of the mouth in response to the wave events was characterised as closed, open or self breaching as listed in Table 5.1b.

Table 5.1b The response of the mouth of the Great Brak Estuary under various freshwater inflows to high wave events of different magnitudes and three day duration.

WAVE HEIGHT (WH in m)	FRESHWATER FLOW ($m^3.yr^{-1}$)				
	34 x 10^6	24 x 10^6	10 x 10^6	2 x 10^6	1 x 10^6
3,82	open	open	open	open	open
3,97	open	open	closed	closed	closed
4,00	self breaching	closed	closed	closed	closed
4,05	self breaching	closed	closed	closed	closed
4,25	self breaching	closed	closed	closed	closed
4,45	self breaching	closed	closed	closed	closed
5,00	closed	closed	closed	closed	closed

The mouth was considered closed if tidal exchange ceased entirely following the high wave event, but was deemed open if tidal exchange still occurred immediately after the event albeit much reduced in magnitude. The term self breaching describes the situation in which tidal exchange ceased for some time after the event while the water level rose to the height of the berm and beyond and natural breaching of the mouth then occurred. This could happen within days of the onset of the wave event or could occur two to three months thereafter.

It is noticeable that only under the freshwater flows characteristic of the natural run-off scenario does the self breaching capability of the mouth manifest itself. In fact, under a freshwater inflow of 34 x 10^6 $m^3.yr^{-1}$, the mouth only closes completely for a wave height of 5,00 m or more. By contrast, the mouth is less resistant to closure under the pre-dam and post-dam situations and the transition between open and closed phases occurs in the range of wave heights between 3,97 m and 4,00 m in the former case and from a wave height of 3,82 m to 3,97 m in the latter case. Clearly, the sensitivity of the mouth to closure is enhanced when freshwater flows are reduced and the mouth then closes at slightly lower wave heights. Additionally, the resilience of the mouth is reduced when freshwater flows are reduced. This may be seen from an analysis of the situations in which the mouth remained open. For instance, the recovery time (the time it takes for the sill height at the mouth to return to within 10% of the pre-disturbance dynamic state) from a wave event with wave heights of 3,82 m is 14 years for the natural run-off scenario, 15 years for the pre-dam situation and more than 24 years for the post-dam situations with an inflow of 10 x 10^6 $m^3.yr^{-1}$ inflow, while recovery of the mouth to within 10% of the pre-disturbance state does not occur under the post-dam situation with low flows of 2 x 10^6 $m^3.yr^{-1}$ and 1 x 10^6 $m^3.yr^{-1}$. Instead, a new state of dynamic equilibrium is achieved within twelve years with a sill height of about 0,83 m to MSL in both cases. For a wave event with wave heights of 3,97 m, the recovery of the mouth takes 15 and 20 years under constant freshwater flows of 24 x 10^6 $m^3.yr^{-1}$ and 34 x 10^6 $m^3.yr^{-1}$, respectively, but under a freshwater flow of 10 x 10^6 $m^3.yr^{-1}$ and lower, the mouth closes completely.

Thus, in the absence of artificial breaching activities and/or freshwater floods, closure of the mouth of the Great Brak Estuary could occur more frequently and for longer periods, even indefinitely, under severely reduced freshwater flows. The understanding derived from this investigation of the response of the mouth to episodic wave events was used in formulating the mouth closure component of the management policies specified in Section 5.1.2 and in quantifying the mechanical assistance necessary for an artificial breaching. It is noteworthy that under the management policies the response of the system to the prescribed wave events remained consistent throughout the simulation period, because the sill height was approximately 0,6 m to MSL during the open mouth phases (that is the conditions at the onset of the wave events were similar throughout). The response of the system to different magnitude wave events following the occurrence of a freshwater flood (altered conditions at the onset of the wave events) will be investigated after the effects of the 1 in 50 year floods are addressed.

The 1 in 50 year flood

The predicted magnitude and maximum flow rate of the 1 in 50 year flood differs for the natural, pre-dam and post-dam run-off scenarios (Bruwer 1987) owing to the development of the catchment, particularly the construction of impoundments. The total volumes discharged under the natural, pre-dam and post-dam run-off scenarios are 1.476 x 10^7 m^3, 1.44 x 10^7 m^3 and 1.235 x 10^7 m^3, respectively, while the maximum flow rates are 820 $m^3.s^{-1}$, 800 $m^3.s^{-1}$ and 686 $m^3.s^{-1}$, respectively. Each flood occurred on 1 October at the start of simulation year seven and was of 10 hour duration. No artificial breachings of the mouth were specified, allowing water levels to rise and the mouth to scour naturally. The effects on the estuary of the 1 in 50 year floods are presented in Table 5.1c for seasonal freshwater flow rates typical of the natural, pre-dam and post-dam situations with seasonality in the wave conditions, but no subsequent wave events.

Table 5.1c The response of the Great Brak Estuary to the 1 in 50 year floods characteristic of the natural, pre-dam and post-dam run-off situations.

RESPONSE TO 1 IN 50 YEAR FLOOD	FRESHWATER FLOW ($m^3.yr^{-1}$)				
	34 x 10^6	24 x 10^6	10 x 10^6	2 x 10^6	1 x 10^6
Min. Sill Height (m to MSL)	- 0.12	- 0.11	- 0.02	0.00	0.00
Recovery Time (yrs)	new state	new state	17	13	12

The response of the estuary to freshwater flooding is described in terms of the behaviour of the sill at the mouth and the corresponding water level variations in the estuary. The scour at the mouth of the estuary owing to the occurrence of the 1 in 50 year flood is greatest for the natural run-off scenario and least for the post-dam scenario with 1 x 10^6 $m^3.yr^{-1}$ freshwater flow. However, the recovery of the mouth is fastest under the lower freshwater flows, because there is an inability to sustain the mouth at depth and the sill height gradually builds up to its previous level. Under the natural and pre-dam run-off situations the estuary attains new states of dynamic equilibrium with sill heights averaging - 0,11 m to MSL and enhanced tidal variations within the estuary of 1,19 m and 1,18 m, respectively. Thus the degree of variation exhibited by the estuarine system is greatest under the higher freshwater flows.

These investigations are highly theoretical. In practice, wave events would disrupt the new dynamic states achieved under the higher run-offs and perturb the slow recoveries of the post-dam scenarios. In order to ascertain whether the response of the estuary would be similar to that exhibited prior to the large freshwater flood (which occurred on 1 October at the start of simulation year seven) or whether the system would develop more or less sensitivity to marine forcing, the effects of eight different wave events commencing on 1 June in the seventh simulation year and lasting for three days were explored.

Freshwater flooding followed by high wave events

The results of applying different wave heights to the estuary under various freshwater inflow rates following the 1 in 50 year flood are listed in Table 5.1d.

Table 5.1d The response of the mouth of the Great Brak Estuary to high wave events of different magnitudes and three day duration, occurring eight months after the 1 in 50 year floods characteristic of different run-off scenarios

WAVE HEIGHT (WH in m)	FRESHWATER FLOW ($m^3.yr^{-1}$)				
	34 x 10^6	24 x 10^6	10 x 10^6	2 x 10^6	1 x 10^6
3,82	open	open	open	open	open
3,97	open	open	open	open	open
4,00	self breaching	self breaching	open	closed	closed
4,05	self breaching	self breaching	closed	closed	closed
4,25	self breaching	self breaching	closed	closed	closed
4,45	self breaching	closed	closed	closed	closed
5,00	self breaching	closed	closed	closed	closed

In contrast to the pre-flood situation, the self breaching mode of behaviour is now manifested under both the natural and pre-dam run-off situations rather than only under the natural run-off scenario. Additionally, after the flood event, mouth closure only occurred at wave heights of 6,00 m and 4,45 m under the natural and pre-dam situations, respectively, whereas previously this occurred at 5,00 m and 4,00 m, respectively. Thus, under inflows of 34 x 10^6 $m^3.yr^{-1}$ and 24 x 10^6 $m^3.yr^{-1}$, the occurrence of a large flood placed the estuary in a new state of dynamic equilibrium which was more resistant to closure of the mouth in both cases. Similarly, under the post-dam run-off scenarios the resistance of the mouth to closure increased following the flood, with closure then occurring at 4,00 m or 4,05 m wave heights as opposed to the 3,97 m wave heights which caused closure previously. The recovery time of the mouth following a wave event of 3,82 m is also reduced in comparison with the case in which no flood occurred. For instance, under freshwater flows of 34 x 10^6 $m^3.yr^{-1}$ and 24 x 10^6 $m^3.yr^{-1}$, the recovery times after the flood are 10 and 12 years, respectively, whereas previously they were 14 and 15 years, respectively. Thus the occurrence of a large freshwater flood undoubtedly alters the response of the estuary to subsequent wave events, enhancing its resistance to mouth closure and reducing the time taken to recover from an episodic wave event of magnitude insufficient to cause closure. However, when closure of the mouth does occur, artificial breaching of the estuary (as specified in the management policies of section 5.1.2) is required to re-set the system to its pre-flood state.

The effects of a flood disturbance on the long term dynamic states

By considering the effects of a 1 in 50 year flood timed to occur at the start of the seventh simulation year for each the management policies with seasonal base flows (i.e. scenarios 1A, 1B, 2, 3A, 3B and 3D), our understanding of the dynamic response of the Great Brak Estuary to the various management policies is enhanced. For instance, the imposition of the 1 in 50 year flood under Scenario 1A induced a new state of dynamic equilibrium with a lower sill height and enhanced tidal variation (Table 5.1c). In the absence of extreme tides or high wave events, this situation persisted. In contrast, under Scenario 1B, the mouth closed temporarily nine months after the flood owing to a high wave event of magnitude WH = 4,45 m and three day duration. Although the system had the capability to breach itself under these conditions, this would not have occurred within 15 days and would also have occurred at levels higher than 1,95 m to MSL. Consequently, the management option of artificially breaching the mouth after 15 days at a rate of -1870 $m.yr^{-1}$ for two hours was applied. Thus the only deviation from the long term average run-off situation occurred in the nine month period between the flood and the mouth closure event. In this period, tidal variations were 1,19 m in the estuary and the sill height averaged about - 0,11m to MSL.

Similarly under the pre-dam run-off scenario, a new dynamic state prevailed after the 1 in 50 year flood with the sill at the mouth averaging - 0,1m to MSL and tidal variations of 1,18 m occurring in the estuary. However, this state only endured for three months before the mouth closed in December in response to a wave event with wave heights of 4,05 m. Although the mouth could breach itself, it was breached artificially at a level of 1,62 m to prevent the occurrence of the higher water levels at which self breaching would have occurred. Thus, following closure of the mouth in December and the artificial breaching of the mouth, the system re-set to the long term average state characteristic of the pre-dam situation.

Despite the occurrence of the large flood in October, the mouth closed in November (WH = 4,00 m for 3 days) and was breached in December in accord with the artificial breaching policies of the post-dam run-off scenarios with base flows of 2×10^6 $m^3.yr^{-1}$ and 1×10^6 $m^3.yr^{-1}$. In the case of Scenario 3A, the mouth did not close in November under wave heights of 4,00 m for 3 days, but instead closed three months later in February. By this time the sill height at the mouth had increased sufficiently for a wave event of 4,00 m to cause closure. Thus for the post-dam run-off scenarios, the deviation from the long term average situation is greatest in the case of the 10×10^6 $m^3.yr^{-1}$ base flow situation and the estuary is in a state more reminiscent of the natural or pre-dam situations from October to February. In the case of the other post-dam scenarios (3B and 3D), the deviations from the long term average dynamic states only occur during October and November. Although this is a short time period, these effects are highly significant because the estuary experiences good freshwater flow followed by extensive tidal flushing during this time.

In summary, consideration of the effects of the occurrence of a 1 in 50 year flood on the long term average dynamic states occasioned by the imposition of management policies 1A, 1B, 2, 3A, 3B and 3D has highlighted characteristic variability in the physical environment of the Great Brak Estuary. Under the high freshwater flows of the natural run-off situation and the occurrence of wave events, the system exhibits strong pulsing, moving from a state with a deep open mouth with strong tidal variation to that of a closed mouth within nine months. This type of variability characterised the Great Brak Estuary prior to extensive development of the catchment. The reduction in the magnitude and duration of such pulsing under reduced freshwater flows is clearly illustrated by the shorter time period during which the deep open mouth prevailed under run-off scenario 2 (three months, sill height about -0,1 m to MSL) and the reduced amplitude of response under run-off scenario 3A (five months, sill height about 0,0 m to MSL). This trend is highlighted in the responses of Scenarios 3B and 3D which exhibit altered variability for a time period of 2 months, mimicing the natural or pre-dam situation for this time, before returning to the semi-permanent, fairly uniform state characteristic of freshwater starvation. The system, therefore, exhibits increased "flashiness" under increased abstraction, losing the strong, long pulsing characteristic of the system under high freshwater flows.

5.1.5 **Synthesis of findings**

The Great Brak Estuary is subject to intermittent closure. From the preceding investigations, it is clear that as freshwater flow to the estuary is reduced the sensitivity of the mouth to closure increases and the prospects of recovery within weeks to months are slim without human intervention. Even where the management options of water releases and mouth breachings are applied, the long term dynamic state of the estuary under low freshwater flows is characterised by reduced variability, a more uniform water column and persistently higher mean salinities. Thus, the response trajectory of the system tends towards increased marine dominance.

Analyses of the effects of large floods on the estuary lead to similar conclusions. Under low freshwater flows the system exhibits a more constrained response of lesser magnitude and duration than under reasonably high base flows (10×10^6 $m^3.yr^{-1}$ and greater). Thus, the responsiveness of the system to freshwater pulsing is reduced and becomes more flashy, highlighting the fact that the estuary moves in the direction of enhanced marine influence with decreasing freshwater flow. The undeveloped system would have had more sustained and intense responses to freshwater floods, exhibiting a greater range of states for longer periods, greater resistance to sea events and quicker recovery from them. In view of the loss of behavioural modes eg. the ability of the mouth to breach itself, and the reduction in biotic niches eg. more limited range of states, and the clear direction of change occasioned by increasing abstraction levels, what conclusions can we draw about the acceptability and efficacy of the management policies tested?

Whereas under the natural and pre-dam situations, axial salinity gradients consistently exceed 20 $kg.m^{-3}$ and the stratification-circulation state of the estuary remains partially mixed with vertical differencing in salinities over the water column, this is not the case for the post-dam run-off scenarios. In wet years, the axial salinity gradient rarely reduces below 20 $kg.m^{-3}$, but in dry years head to mouth differences in salinity never attain 20 $kg.m^{-3}$ unless a flood simulating water release is undertaken. Similarly, although the stratification-circulation state of the estuary remains partially mixed, the water column is fairly uniform in dry years apart from during a water release. It is clear that there are substantial detrimental effects on the abiotic character of the Great Brak Estuary in moving from a wet to a dry post-dam run-off scenario. Whereas the wet post-dam situation retains much of the character of the pre-dam situation, this is not the case in dry years. Water releases have been shown to temporarily alleviate the dry post-dam situations and to benefit the estuary more than a policy of a continuous very low base flow by briefly emulating the pre-dam or natural situation. Thus for a given abstraction level in the post-dam situation, policies involving water releases and mouth breachings are preferred above those of low base flows only.

Further, this investigation has highlighted that a guaranteed total allocation of 2 x 10^6 $m^3.yr^{-1}$ alters the character of the abiotic environment of the estuary. In view of this stress situation under the existing policy for dry years, a guaranteed allocation of 1 x 10^6 $m^3.yr^{-1}$ is unacceptable. Indeed, it is anticipated that the existing policy will affect the biotic functioning of the estuary should a number of dry years occur in sequence. This is borne out by observations of the population and size classes of the estuarine mud prawn, *Upogebia africana*, in the Great Brak Estuary over a dry three year period (Wooldridge 1994). Populations in the estuary showed a skewed distribution towards large size classes owing to prolonged inlet closure and an overall reduction in numbers. There is little doubt that if an investigation as to the acceptability of constructing a similar dam on a similar system were commissioned today, the guaranteed allocation deemed acceptable would have been nearer the 10 x 10^6 $m^3.yr^{-1}$ mark than the present allocation to the estuary. In justifying such recommendations, the results of simulations of the estuarine systems model would prove very useful. Indeed, we can conclude that for the Great Brak Estuary, the present policy of flood simulating water releases in dry years is more effective than that of a continuous low base flow and that the abiotic character and functioning of the estuary will be well maintained over wet years.

5.2 Site Specific Management Applications: The Kromme Estuary

5.2.1 Management performance indices

Issues of significance in the permanently open Kromme Estuary relate to severe flood attenuation and reduced ecological productivity arising from reduced nutrient inputs and salinity effects such as minimal axial salinity gradients and the occurrence of hypersalinities. As morphological changes in any but the mouth area were excluded from consideration in the formulation of the Estuarine Systems Model, the effects of reduced flooding on sedimentation in the Kromme Estuary cannot be tackled. However, the success (failure) of fish recruitment to estuaries has been related to the presence (absence) of head to mouth salinity differences of 20 $kg.m^{-3}$ or greater, while the presence of a brackish component in an estuary is strongly linked to the diversity of the estuarine floral community and the productivity attributable to phytoplankton. Salinity distributions therefore provide the critical link between the effects on the abiotic environment and the responses of the biota to reduced freshwater flows. Thus the index most relevant to an assessment of the maintenance of the character and functioning of the Kromme Estuary is the management performance index related to salinity distributions. No critical time period or duration is specified in defining the salinity index on which the performance index is based, nor is a critical axial salinity difference prescribed. The occurrence of hypersalinities will tend to decrease the salinity index, as desired, because the axial salinity difference will be zero or near zero at these times.

The major aims in undertaking the Kromme case study were to examine alternative uses of the annual freshwater allocation to the estuary and possibly provide grounds for an increase in this annual allocation by demonstrating the detrimental consequences of substantial freshwater abstraction from the system. The standard trajectory (Section 4.4.2), therefore, provides an ideal reference state against which to measure the effects of freshwater abstraction in terms of salinity distributions. The time interval selected for evaluation of the management performance was the simulation period one to two years, because the average natural run-off scenario had achieved an annually periodic dynamic state by then (Figure 4.4b). The management evaluation component of the Kromme model thus comprises:

$$SI^P = ASD^P$$

$$PI_2 = \int_T SI^P\,dt \Big/ \int_T SI^R\,dt \quad \text{and } T = [1,2]$$

where SI^P = salinity index under policy P ($kg.m^{-3}$)
ASD^P = axial salinity difference under policy P ($kg.m^{-3}$)
PI_2 = management performance index 2 (unitless)
T = time interval over which management performance is evaluated (yr)
SI^R = salinity index under the reference policy R ($kg.m^{-3}$)

5.2.2 Management policies

The Kromme Estuary presently receives less than 2%, on average, of the run-off it would have received under natural conditions. The optimal use of the annual allocation of 2 x 10^6 m^3 is a matter of immediate concern following criticism by ecologists of the efficacy of the present procedure of monthly releases of equal volumes (DWAF 1993b). Thus alternative strategies for use of the present allocation must be included in the management policies considered in the Kromme case study. To address this aspect of the management of the Kromme Estuary alone, would be to focus on immediate management requirements and neglect slightly longer term requirements and the overall estuarine management objective of maintaining the character and functioning of a system. There is a need to make a case for an increase in the environmental allocation to the estuary by demonstrating the detrimental effects of reduced freshwater flow. Therefore, management policies which lead to the development of an understanding of the changes in state of a permanently open estuary owing to alterations in inflow from the natural run-off condition to a situation with little or no freshwater input, must be included.

Consequently, ten management policies were selected for evaluation in the Kromme Case Study. These fall into six total annual allocation categories, with four alternative management strategies considered under the present annual allocation category. The management policies, which are described in detail below, were formulated in association with members of the Consortium for Estuarine Research and Management (primarily Mr P Huizinga of the CSIR) in order to ensure that they were as reflective of present management alternatives and pragmatic uses of allocations as possible.

1. ***Natural run-off scenario (the standard trajectory)***

The natural MAR of the Kromme Estuary is approximately 120 x 10^6 $m^3.yr^{-1}$. The estimated natural (undeveloped) condition is included as a run-off scenario, despite a lack of quantitative data, because it provides a standard against which the change in estuarine condition can be evaluated as a consequence of changes in run-off to the estuary. No high flow events were included in this scenario. The inflow was parametrized as a seasonal base flow amounting to the natural MAR, according to the seasonality in rainfall for the Port Elizabeth area (Table 4.3b). Simulation results from this scenario, therefore, provide information on the average seasonal water levels, salinity distributions, stratification-circulation state and mouth condition in the estuary prior to the construction of the Mpofu Dam.

2. ***Intermediate run-off scenario: 40% of natural MAR***

The majority of the annual allocation of 48 x 10^6 m^3 enters the estuary as a continuous base flow of 37.39 x 10^6 $m^3.yr^{-1}$ or 1.186 $m^3.s^{-1}$. An annual flood equivalent to the 1 in 2 year flood (8.61 x 10^6 $m^3.yr^{-1}$ (Jezewski & Roberts 1986)) is set to occur in early summer (October/November), while a late summer flood of volume 2 x 10^6 m^3 occurs four months after the first flood. The maximum discharge rates of the flood releases occur at the maximum release rate from the Mpofu Dam, which is between 20 and 22 $m^3.s^{-1}$.

3. ***Intermediate run-off scenario: 20% of natural MAR***

Nearly half of the annual allocation of 24 x 10^6 m^3 enters the estuary as a continuous base flow of 13.39 x 10^6 $m^3.yr^{-1}$ or 0.425 $m^3.s^{-1}$. An annual flood equivalent to the 1 in 2 year flood (8.610 x 10^6 $m^3.yr^{-1}$) is set to occur in early summer (October/November), while a late summer flood of volume 2 x 10^6 m^3 occurs four months after the first flood. The maximum discharge rates of the flood releases occur at the maximum release rate from the dam, which is about 21 $m^3.s^{-1}$. This inflow scenario has a total annual allocation half that of the intermediate run-off scenario 2.

4. ***Intermediate run-off scenario: 10% of natural MAR***

The base flow to the estuary is taken as the evaporative requirement of 2.372 x 10^6 $m^3.yr^{-1}$ specified for the Kromme Estuary by Jezewski and Roberts (1986). An annual flood equivalent to the 1 in 2 year flood (8.610 x 10^6 $m^3.yr^{-1}$) is set to occur in early summer (October/November)

A freshette of volume 1x 10^6 m^3 is set to occur in late summer, that is about 4 months after the early summer flood. The maximum discharge rates of the flood releases occur at the maximum release rate from the dam, which is about 21 $m^3.s^{-1}$. The total annual allocation of this scenario is half of the intermediate run-off scenario 3 and a quarter of the intermediate run-off scenario 2.

5. ***Present run-off scenarios: 2% of the natural MAR***

The total annual allocated volume is 2 x 10^6 m^3. The present release policy of monthly releases of one twelfth of the annual volume was formulated originally with the aim of preventing the occurrence of hypersalinities. The evidence indicates that this management policy has been unsuccessful in this endeavour (DWAF 1993b) and that other strategies to reduce the occurrence of hypersalinities and increase axial salinity gradients need to be considered. The alternative management strategies to be evaluated, therefore, comprise:

A The present release policy of monthly releases of one twelfth of the annual volume,

B One flood release of 2 x 10^6 m^3 in early summer,

C Two releases of equal volume (1 x 10^6 m^3) in early summer and six months later,

D Two releases of equal volume (1 x 10^6 m^3) in early summer and four months later,

E Three releases per year, one release of 1 x 10^6 m^3 in early summer and two smaller releases of equal volume later, but not in winter.

6. ***No freshwater releases***

This scenario is included so that the detrimental effects of freshwater deprivation can be indicated.

5.2.3 Simulation results

All model simulations were conducted for a two year period, commencing on 1 October for each of the run-off scenarios. Only the results of the second year of the simulation are presented as any influence that the choice of initial conditions might have imposed on the model results would then have dissipated. The initial conditions include the assumption that the estuary has a mean salinity of 32 $kg.m^{-3}$ on 1 October, that the sill height at the mouth is 0.09 m to MSL and that the system is well mixed.

Unlike the Great Brak Estuary, where the major determinant of the physical dynamics is the state of the mouth, that is, whether the mouth is closed or open and, if open, to what extent and with what frequency, the mouth of the Kromme Estuary is permanently open. Furthermore, the volume of water entering and leaving the estuary on ebb and flood tides is of sufficient magnitude to maintain the mouth in a permanently open state, provided the volumes of sediment and the rates of delivery to the estuary are comparable with present day quantities. In this study, we are specifically assuming that the discharge velocity of freshwater does not exceed the maximum release rate of the upstream impoundment, that is, 22 $m^3.s^{-1}$. Flow of this magnitude is insufficient to cause substantial movement or delivery of sediment to the Kromme Estuary. Thus, the implications of the inflow scenarios for the sediment balance of the Kromme Estuary are not addressed in this study. Rather, the effects of the different inflow scenarios were observed primarily in the different salinity regimes and the discussion of model results concentrates on these.

There were no significant changes in tidal flux, water levels, current speeds and the height of the sill at the mouth with the different run-off scenarios. The water levels in the Kromme Estuary typically varied between 0,2 m to MSL and 1, 0 m to MSL, while the maximum flood tidal flux is of the order of 150 $m^3.s^{-1}$ under all of the run-off scenarios.

The natural run-off scenario

The inflow to the estuary under the natural run-off scenario was simulated by imposing the seasonality of the Eastern Cape region on the average annual run-off (Figure 5.2a). This means that no large floods were simulated, nor low flow periods. Rather, the long term average situation was simulated assuming the rainfall and evaporation figures representative of the region. Similarly, the spring-neap and semi-diurnal tidal variation typical of the Port Elizabeth region was assumed. The mean salinity in the estuary (Figure 5.2b) exhibited strong seasonal variation with low mean salinities of less than 10 $kg.m^{-3}$ in the high freshwater flow season (September to mid December) and high mean salinities of greater than 28 $kg.m^{-3}$ in the dry season (mid February to mid April). Only in the dry season did the head to mouth salinity difference decrease below 20 $kg.m^{-3}$. Superimposed on the seasonality of the mean salinity is a higher frequency spring-neap tidal influence, which as expected can also be detected in the stratification index. The stratification index was less than 0.08 for the majority of the year, only exceeding this value from early September to mid October. Even then, the maximum value attained was only 0.34, indicating that under the highest average freshwater flows of the natural situation, the Kromme Estuary did not attain a highly stratified state. Instead, the stratification-circulation state of the estuary may be classified as partially mixed during the high flow period and as well mixed for most of the year.

The intermediate run-off scenario (40% of the natural MAR)

The differences between the freshwater inflow to the estuary under this scenario and under the previous scenario (Figure 5.2a) lie in the occurrence of two floods, the first on 15 November and the second four months later, and a reduced base flow (about 31% of that under the natural run-off scenario). The effect of reducing the base flow is evident in the enhanced mean salinities which occur in the estuary (Figure 5.2c). Mean salinities exceed 30 $kg.m^{-3}$ for about seven months of the year as opposed to a period of one and a half months under the natural run-off scenario. Mean salinities only decrease below 20 $kg.m^{-3}$ when the 1 in 2 year flood volume is released and for about 25 days thereafter. The minimum mean salinities attained are about 5 $kg.m^{-3}$. In contrast, the release of 2 x 10^6 m^3 in March has an effect on mean salinities similar to the increase in base flow in June and September/October, which causes mean salinities to decrease below 25 $kg.m^{-3}$. Despite the lower base flow, strong seasonal variation is still exhibited in the mean salinities and the spring-neap tidal effect is also evident.

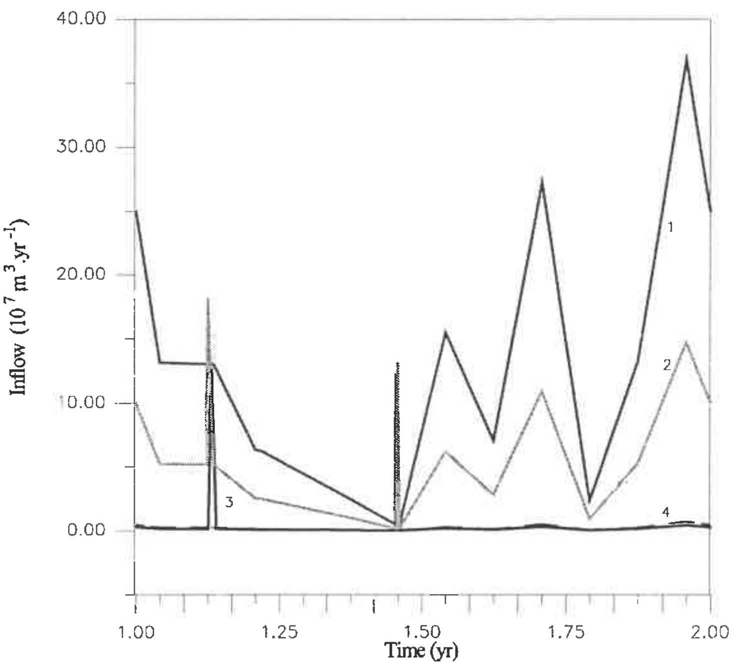

Figure 5.2a Inflow to the Kromme Estuary under the the run-off scenarios 1, 2, 3 and 4

Axial salinity differences of 35 $kg.m^{-3}$ are predicted for about 5 months of the year under this run-off scenario (Figure 5.2c), indicating that a brackish area probably is maintained in the upper reaches of the estuary. The 1 in 2 year flood release is important in that a longitudinal gradient of 35 $kg.m^{-3}$ is maintained for more than a month over the mid summer period when it might otherwise have declined below 20 $kg.m^{-3}$. Similarly, the second flood causes the head to mouth differences in salinity to increase sharply from near zero to 35 $kg.m^{-3}$ and remain at this level for over a week in early autumn. The stratification index only exceeds 0.01 during the 1 in 2 year flood release, attaining a maximum value of 0.197 in mid November. The stratification-circulation state of the estuary, therefore, is well mixed under this run-off scenario apart from a partially mixed phase during the summer flood release.

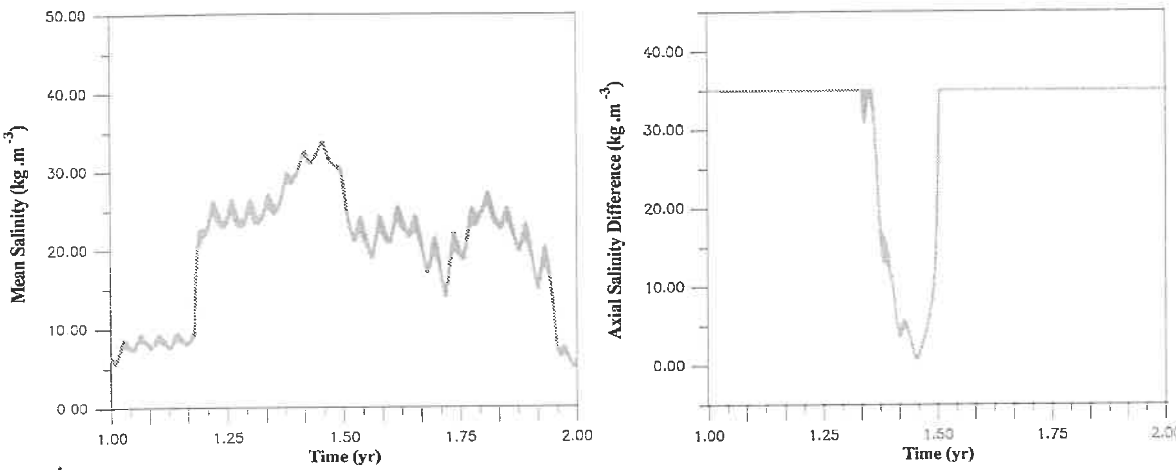

Figure 5.2b The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the natural run-off scenario

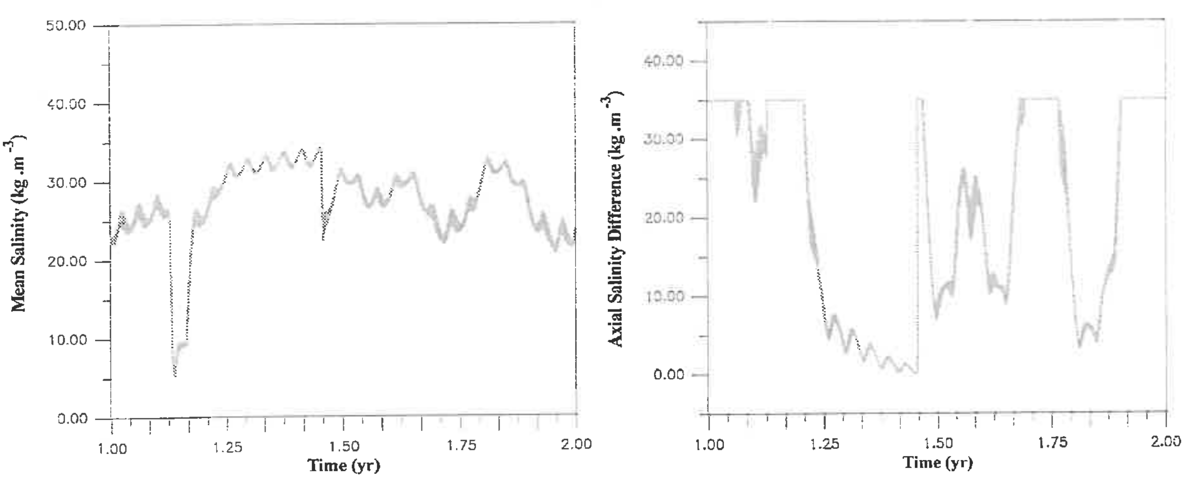

Figure 5.2c The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the intermediate run-off scenario (40% MAR)

Despite the two flood releases the overall adverse effects of reducing freshwater flow to the system are captured in the management performance index for salinity distributions, which decreases from unity for the natural run-off situation to 0,657 when only 40% of the natural MAR enters the estuary.

The intermediate run-off scenario (20% of the natural MAR)

The differences in inflow between this and the preceding scenario lie in the volume of base flow received by the estuary (Figure 5.2a). Both scenarios include a release equivalent in volume to the 1 in 2 year flood timed to occur on 15 November and a freshette of 2 x 10^6 m^3 four months later. Consequently, the mean salinities in the estuary are higher than under the previous two scenarios and the effects of the flood releases are more noticeable (Figure 5.2d). Mean salinities generally exceed 30 $kg.m^{-3}$, exhibiting a strong spring-neap influence and a much reduced seasonal influence, and only decrease below 25 $kg.m^{-3}$ under the influence of the water releases.

The 1 in 2 year flood release causes the axial salinity difference in the Kromme Estuary to rise from less than 5 $kg.m^{-3}$ to 35 $kg.m^{-3}$ and remain at this level for more than a month (Figure 5.2d). The late summer flood also achieves an axial salinity difference of 35 $kg.m^{-3}$ in the estuary and this is maintained for about a week. Although the higher base flows during the months of September and October are able to sustain a longitudinal salinity gradient greater than 10 $kg.m^{-3}$, it is doubtful whether this together with the effects of the flood releases is sufficient to maintain a brackish region in the estuary.

An analysis of the stratification-circulation state confirms this, in that the maximum value which the stratification index attains is 0.132. This value reflects the enhanced buoyancy input associated with the 1 in 2 year flood release compared with the usual situation in which the stratification index is consistently less than 0.005. The estuary is therefore in a well mixed state for the majority of the time.

The management performance index attains a value of 0,219 for this run-off scenario, indicating that the reduction in base flow compared with the previous scenario (management performance index 0,657) has a strong detrimental effect on salinity distributions.

The intermediate run-off scenario (10% of the natural MAR)

The major differences in the inflow received by the estuary under this run-off scenario and the preceding two scenarios are the substantial decrease in base flow to only 2.372 x 10^6 $m^3.yr^{-1}$ and the reduction in the volume of the freshette in March by half (Figure 5.2a). Consequently, the mean salinity in the system lies between 32 and 34 $kg.m^{-3}$ for most of the time, showing a strong

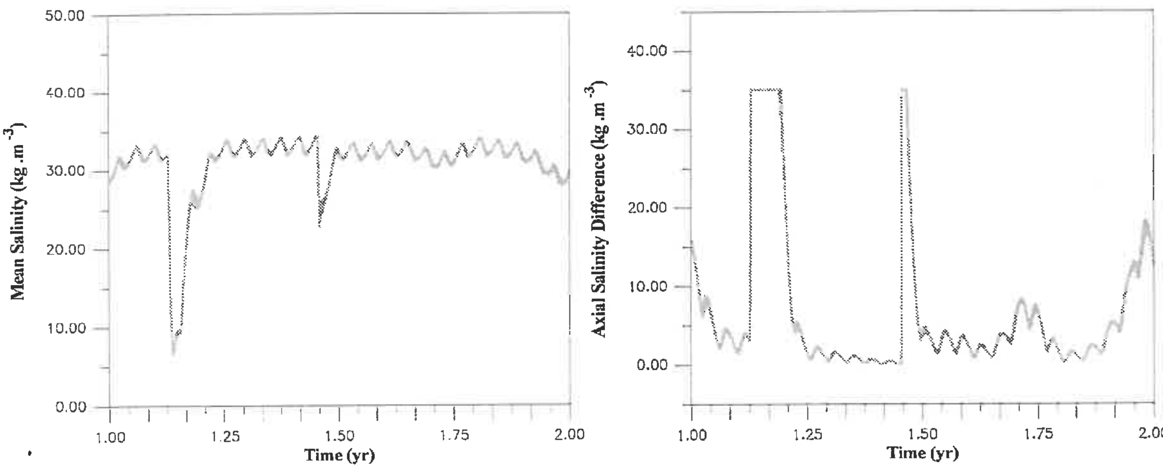

Figure 5.2d The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the intermediate run-off scenario (20% MAR)

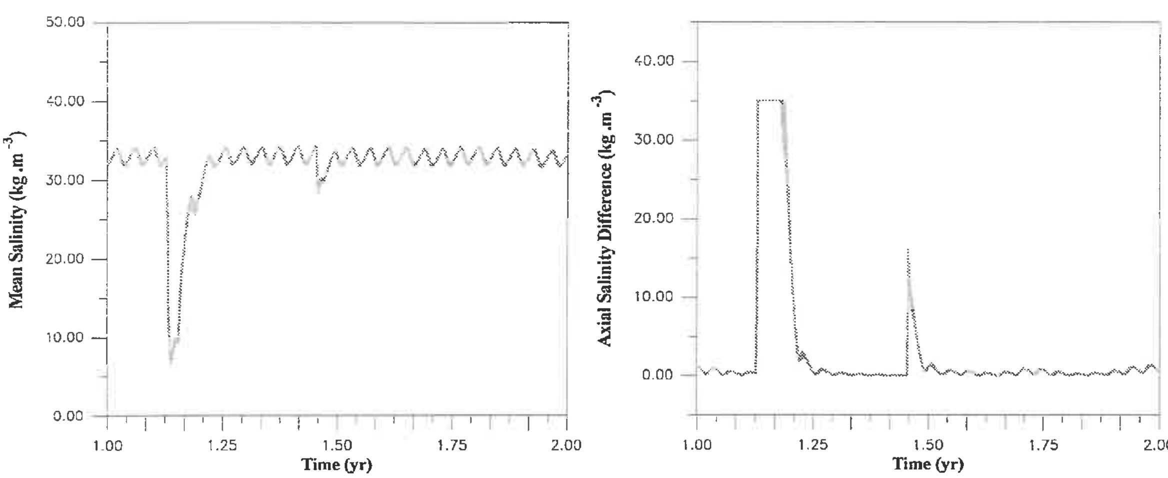

Figure 5.2e The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the intermediate run-off scenario (10% MAR)

spring-neap variation and no seasonal variation (Figure 5.2e). Only during the two flood events do mean salinities decrease below 30 $kg.m^{-3}$. The 1 in 2 year flood, which enters the estuary on 15 November, causes the mean salinity to drop to about 6,5 $kg.m^{-3}$. A condition of mean salinity less than 10 $kg.m^{-3}$ is maintained for about 15 days, but the effect of the flood on mean salinities has dissipated substantially by mid December and entirely by the end of December. The associated axial salinity differences are 35 $kg.m^{-3}$ at the height of the flood and for a month thereafter, but decline to less than 3 $kg.m^{-3}$ by the end of December. The effect of the freshette is shortlived with mean salinities returning to their usual values of between 32 and 34 $kg.m^{-3}$ within a month and axial salinity differences also decreasing from a maximum of 18 $kg.m^{-3}$ to less than 3 $kg.m^{-3}$ within a month. This contrasts with the effect of the freshette of 2 x 10^6 m^3 released in the previous two scenarios, which caused head to mouth salinity differences to rise well above 20 $kg.m^{-3}$, to 35 $kg.m^{-3}$ for about a week.

The stratification index briefly rises above 0,08 under the influence of the 1 in 2 year flood, indicating that a partially mixed state may prevail under these flood conditions. The estuary remains in a well mixed state otherwise.

The management performance index for this scenario is only 9,9% of that for the natural run-off situation, emphasising the severe effects of reduced freshwater flow on the maintenance of salinity gradients in the estuary.

Present run-off scenarios: 2% of the natural MAR

A Monthly releases

The mean salinity of the estuary is barely affected by the monthly freshwater releases currently undertaken, exhibiting slight decreases in value at the time of the releases but remaining consistently above 31,5 $kg.m^{-3}$, varying with the spring neap tidal cycle (Figure 5.2f). The head to mouth salinity differences are also consistently less than 2,5 $kg.m^{-3}$, indicating that the estuary is virtually a sheltered extension of the sea under the present run-off scenario. Stratification is absent and the estuary is well mixed. The management performance index is only 0,8 % of that under the natural run-off scenario.

B One release per year (in early summer)

Under the release of the full annual allocation in mid November, the mean salinities decline from values between 32 and 34 $kg.m^{-3}$ to 22,5 $kg.m^{-3}$ (Figure 5.2g). Lower mean salinities are exhibited for slightly less than a month and the dominant effect on salinities for the rest of the year is exerted by the spring neap tidal cycle. However, a head to mouth axial salinity gradient of 35 $kg.m^{-3}$ is achieved for about a week in November, but dissipates fairly rapidly and all effect on the axial salinities has disappeared within a month. Stratification is minimal and the system

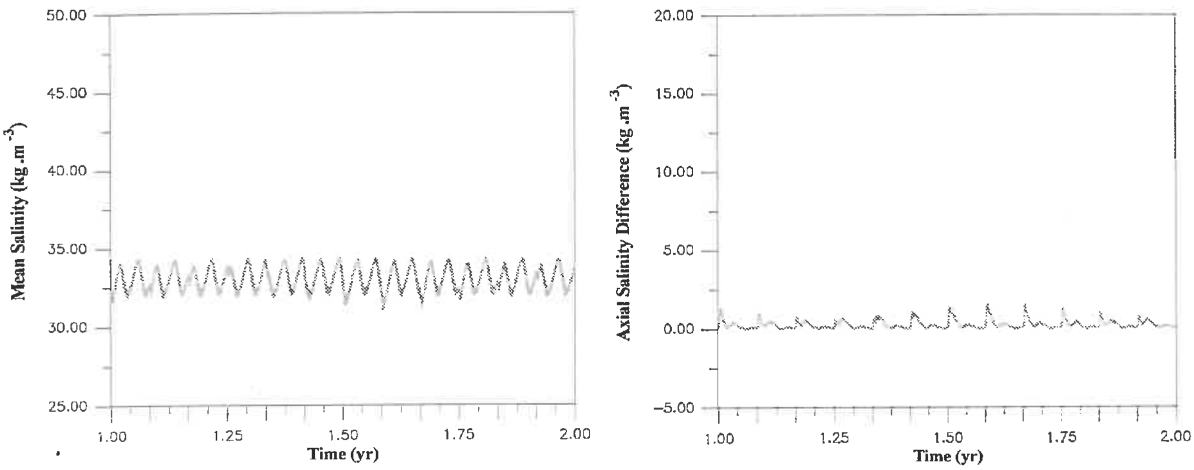

Figure 5.2f The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the present run-off scenario 5A

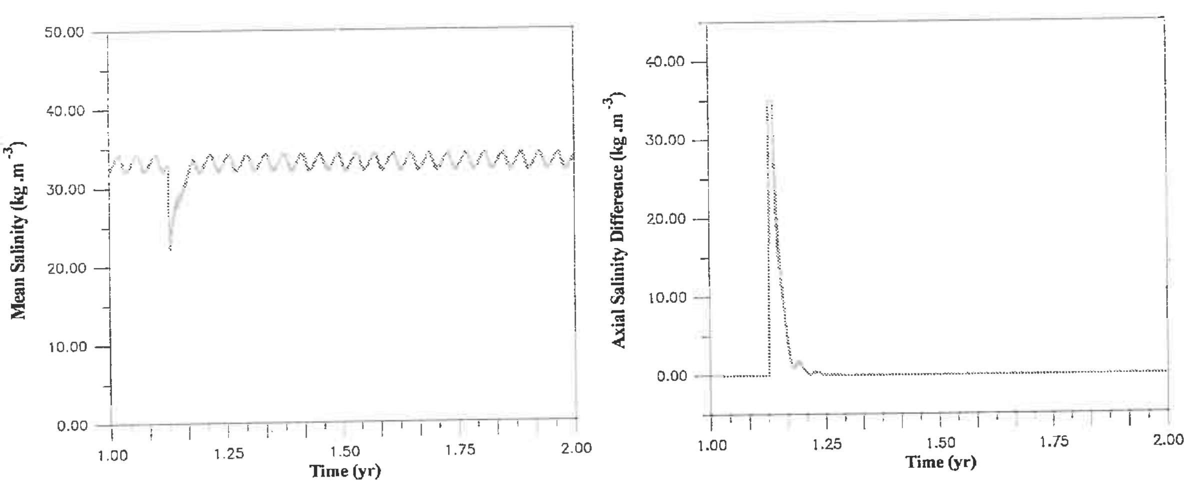

Figure 5.2g The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the run-off scenario 5B

is well mixed apart from during the flood release when a partially mixed state is temporarily attained. A management performance index of 0,027 is attained. Although this is a severe reduction compared with the reference natural scenario, it is an improvement in terms of salinity distributions on the present policy of monthly releases.

C Two releases of equal volume per year (early summer and six months later)

The effect of the freshwater releases on the mean salinity in the estuary varies slightly according to the stage of the spring neap tidal cycle at the time of the release. On the 15 November, the mean salinity of the estuary was higher than on 15 May. The minimum mean salinity achieved was thus lower under the second release than under the first, but both are in the region of 27 $kg.m^{-3}$ (Figure 5.2h). The effects of the releases on mean salinities dissipated completely within a month in both cases. The axial salinity differences achieved were between 18 and 21 $kg.m^{-3}$, but these effects were shortlived and the estuary remained in a partially mixed state throughout. The management performance index is 0,017, indicating that this policy is more beneficial for salinity distributions in the estuary than the policy of monthly releases and yet not as helpful as one large relaese in maintaining axial salinity gradients as high as possible for as long as possible.

D Two releases of equal volume per year (early summer and four months later)

Both releases caused the mean salinities to decrease to about 27 $kg.m^{-3}$ and their effects dissipated within a month (Figure 5.2i). The axial salinity differences resulting from the releases were between 17 and 18 $kg.m^{-3}$, but these effects were temporary and the head to mouth differences in salinity soon declined to less than 2 $kg.m^{-3}$. The stratification-circulation state of the estuary was well mixed. The management performance index is 0,016, almost equivalent to that of the previous policy.

E Three releases per year (larger release in early summer, 2 smaller releases later, but not in winter)

The effect of the first release is the same as described in the previous two scenarios. The effects of the two smaller releases on mean salinities are very slight and mean salinities do not decline below 30 $kg.m^{-3}$ as a result of the releases (Figure 5.2j). This is borne out by the axial salinity differences which remain below 5 $kg.m^{-3}$ at the time of the two smaller releases. The estuary was well mixed throughout the simulation and the management performance index was 0,013. This indicates that three smaller releases are not as effective in maintaining head to mouth salinity differences as the one large release or even two releases of equivalent total volume per annum.

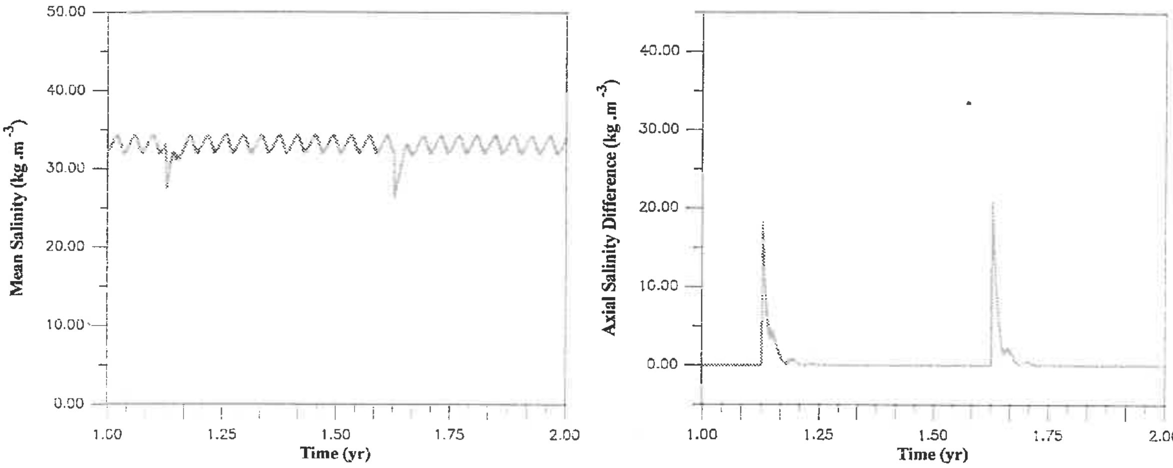

Figure 5.2h The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the run-off scenario 5C

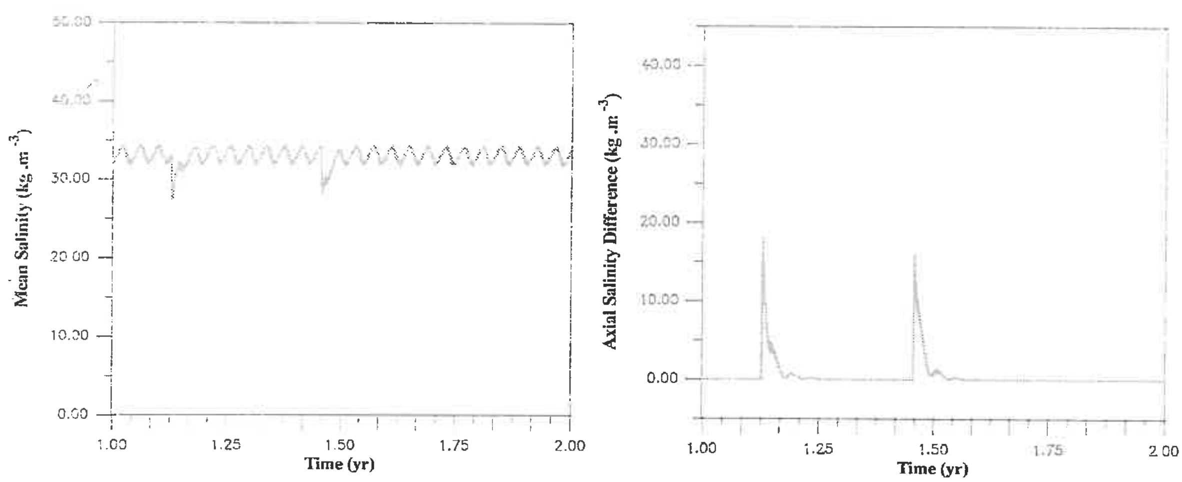

Figure 5.2i The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the run-off scenario 5D

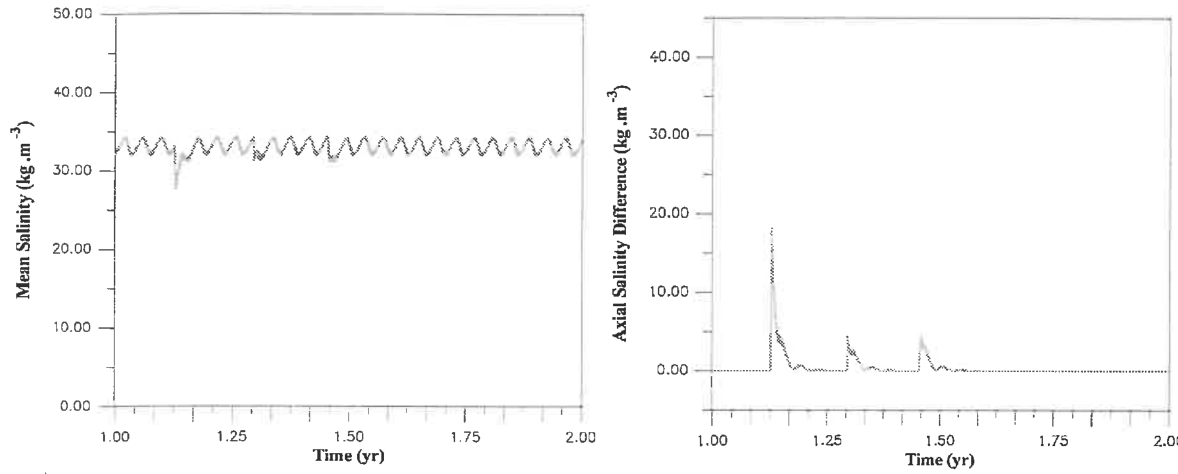

Figure 5.2j The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the run-off scenario 5E

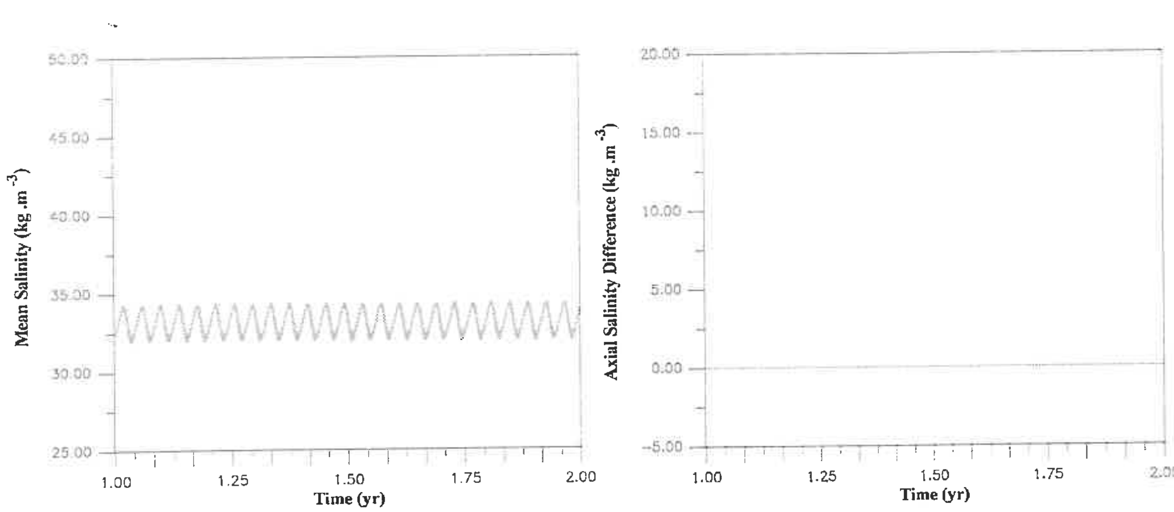

Figure 5.2k The simulated mean salinities (left) and axial salinity differences (right) of the Kromme Estuary during the second year of the simulation under the no freshwater release scenario 6

No freshwater releases

Mean salinities are consistently higher than 32,4 $kg.m^{-3}$ throughout this simulation and vary strongly with the spring neap tidal cycle, but do not exceed 35 ppt (Figure 5.2k). Although, the Estuarine Systems Model does not accommodate the prediction of hypersalinities, it is clear that the propensity for hypersaline conditions to occur under this scenario is very high and the management performance index is very near zero.

5.2.4 **Disturbance scenarios**

The preceding analysis highlights the long term effects of reduced freshwater flow to the Kromme Estuary and the uniform, well mixed nature of the water body which results from severe abstraction levels such as occur at present. The character of the Kromme Estuary has clearly been impacted by catchment developments and the system presently exists as an arm of the sea, intermittently exhibiting a more estuarine character when a pulse of freshwater reaches the system. Whereas an annual allocation of 12 x 10^6 $m^3.yr^{-1}$ to 24 x 10^6 $m^3.yr^{-1}$ to the estuary allows for the maintenance of aspects of the character and functioning of the undisturbed system, the present allocation of 2 x 10^6 $m^3.yr^{-1}$ does not permit the maintenance of characteristic salinity gradients, indicating that the system is stressed in comparison with its natural state. The degree to which the system is stressed may be investigated further by considering disturbances or perturbations to the dynamic states arising from the imposition of the management policies. The most likely perturbation to an annual water allocation policy in the semi-arid Eastern Cape region is a drought situation. Consequently, the disturbance scenario selected for testing comprised ***no freshwater flow to the estuary for a period of one year***.

The response of the Kromme Estuary to such a disturbance (commencing at the beginning of simulation year three and persisting for one year) varies depending on the amount of freshwater usually reaching the estuary. In the case of scenario 6, there is no disturbance and the normal estuarine state persists. For the present run-off scenarios, this disturbance causes a less than 10% deviation in the salinity values in each case and the normal dynamic estuarine states are completely restored once the first water releases commence in the fourth year of the simulation. For the run-off scenarios ranging from 10% of the MAR to the natural situation, the extent of deviation from the prevailing estuarine states increases with increasing freshwater flow and the recovery time is fastest under the highest freshwater flows. The natural run-off situation is re-established within 18 days of the drought breaking (i.e. the cessation of the disturbance), whereas the scenarios 2, 3, and 4 take between 28 and 30 days before normality is re-established. Thus, the natural scenario exhibits the most intense reaction and the strongest resilience. As the freshwater supply to the estuary decreases, the system becomes more inert to drought disturbances and the strong pulsing behaviour associated with the natural system is reduced in intensity until it is manifested no longer.

5.2.5 **Optimisation**

In view of the fact that the current allocation of water to the Kromme Estuary is 2×10^6 $m^3.yr^{-1}$, one of the management questions most urgently requiring an answer is the optimal use of this freshwater allocation. This question is partially addressed earlier in that alternative uses were investigated. These ranged from monthly releases of one twelfth of the annual volume to one large release of the whole allocation in mid November, two releases of half of the annual allocation in November and May or in November and March, and three smaller releases. The simulation results indicate that the policy of releasing the whole allocation in one flood in November is more favourable for the maintenance of axial salinity differences than any of the other options. The management performance indices confirm this with policy 5B scoring slightly more than the other policies with this total allocation. However, given that a policy of one large release benefits the system slightly more than the other policies with the same annual allocation, the question as to the optimal timing and duration of such a release remains. As the volume of the release and the maximum rate at which the water can be released are fixed, the question becomes an optimisation problem of the form:

$$\max f(st_1, dur_1) \qquad \text{subject to } 0 \leq st_1 \leq 1 \text{ and } 0 \leq dur_1 \leq 1$$

where $f : \mathbf{R}^2 \rightarrow \mathbf{R}$ and is the management performance index to be optimised, while st_1 is the inception time of the flood and dur_1 is the duration of the release (i.e. a maximisation problem with simple bounds).

Such an optimisation was undertaken using the finite-difference gradient method of the IMSL MATH/LIBRARY known as DBCONF (IMSL 1991). The reasons for the selection of this optimisation routine and the non-routine parameter selections involved in implementing the routine on the Kromme Estuary are described in Appendix B. The results indicate that the release should optimally occur on 22 September and endure for seven days or occur on 26 September and endure for 3 days. These releases generate a management performance index of 0,030 compared with that of management policy 5B (0,027). If September and October are eliminated from the feasible region over which the optimisation is allowed to occur (in case their selection is an artefact of annual cycling), the next best option is to flood on 10 June with a short sharp burst of less than 2 days. However, this optimisation produces a management performance index of 0,029, which is less than that occasioned by a flood release in September. In reality it is doubtful whether a June flood would even have the beneficial effect indicated, because hypersalinities are not simulated by the Estuarine Systems Model but would probably occur in

February, March, April and May if the flood release were timed for June. Thus it is concluded that mid to late September is indeed the best time for the release of the present total annual allocation to the estuary as a flood of between three and seven day duration, purely on the basis of the axial salinity gradients which will result under the average rainfall and evaporation conditions.

As a further test of the robustness of this conclusion, the three endogenous functions to which the Kromme results proved most sensitive, namely: the reduced salinity function, the salinity export multiplier and the stratification export function (Section 4.4.3), were allowed to vary simultaneously within 5% either side of their nominal value. The maximum management performance index then obtained under run-off scenario 5B (flood release on 15 November) was 0,028. This represents a deviation of less than 4% from the figure attained with nominal parameter values and is less than the maximum attained with the September or June floods, indicating that the management evaluation component of the Kromme model is reasonably robust to some parameter uncertainty. A fair degree of confidence, therefore, may be placed in the management evaluation results and in the selection of the optimal flooding time and duration.

5.3 Summary of the Site Specific Management Applications

The estuarine systems model was applied to two systems with different mouth conditions: one permanently open and the other intermittently closed, but with similar management issues regarding freshwater allocations and flows. The implementation of the management evaluation component on the Kromme and Great Brak case studies has demonstrated the efficacy of the model in site specific management by:

- indicating quantitatively the consequences of reduced freshwater supply to the systems in terms of the effects on the mouth, salinities, the associated stratification-circulation states and the degree of tidal and freshwater flushing of the estuaries,
- enabling a comparative assessment of the relative merits of different management policies comprising freshwater abstraction levels, water releases and mouth closures and breachings (where appropriate), in maintaining the characteristic functioning of the estuaries,
- facilitating the selection of one management policy above another within the bounds of the present total annual allocations to the estuaries, and
- providing the background against which future decisions on allocations to the estuaries can be made.

Thus the stated purposes in site specific management applications (Sections 5.2.1 and 5.1.1) have been met in that effective decisions can be made based on model predictions and evaluations which take into account the inherent characters of the estuaries and accommodate the time varying nature of their responses.

The management evaluation has addressed both the average long term responses of the estuaries to reduced freshwater flows and the responses to episodic events or perturbations of these long term average states. The understanding developed in this way facilitates more robust decision-making and highlights the alterations in behavioural responses associated with the progression of the systems along trajectories to more uniform, marine dominated systems as freshwater supplies are reduced. These alterations include reduced intensity and duration of pulsing, an enhanced sensitivity to marine forcing, a loss of resilience in the response of the mouth (eg. loss of the self breaching capability) and reduced axial salinity differences with increasing freshwater abstraction. The results indicate that precautionary measures are necessary to ensure that this trajectory does not become the norm for estuarine systems so reducing the diversity along the South African coast.

A limitation of the Estuarine Systems Model which is clear from the study lies in its inability to predict hypersalinities. Their likelihood of occurrence can be inferred from high mean salinity values and low axial salinity gradients, but the concentrations achieved cannot be predicted. As the extent and persistence of hypersalinities is biologically significant, this is unfortunate. Similarly, the specific exclusion of major sedimentary effects and the concentration on the mouth channel and changes therein when the model was formulated, has limited the consideration of episodic flooding in the case of the Kromme Estuary. This aspect was addressed well in the Great Brak study and proved very useful in developing an understanding of the dynamic physical behaviour of the system and providing the water levels, salinities and tidal flux information necessary for biological response assessments. A particularly positive aspect of the site specific management applications lies in the finding that the management evaluation sector is reasonably robust to perturbations of the parameter values to which the Kromme model is most sensitive. This means that a fair degree of confidence can be placed in the finding that the optimal timing and duration of one large release of the present annual allocation to the Kromme Estuary is mid to late September for between three to seven days. Although this aspect was not addressed specifically for the Great Brak Estuary, the fact that three management performance indices were applied and all three concurred in their indications enhanced the reliance placed in the model results. In summary then, the estuarine systems model is an effective tool in elucidating the consequences of management policies on the physical dynamics of the small estuaries typical of the South African coast and in facilitating decision making on freshwater abstraction levels, flood simulating releases and mouth breachings.

5.4 Integrative Management Applications

In evaluating the acceptability or otherwise of the medium to long term responses of estuaries to management policies involving freshwater allocations, water releases and mouth breachings, it is not only the physical dynamics that are important. Often the effects on the ecosystem or the combined abiotic and biotic responses are deemed most significant by managers and the public. Accordingly, the concept of a more comprehensive predictive approach using the estuarine systems model as a basic building block and incorporating biological prediction arose. An investigation in this regard was initiated in September 1992 and resulted in a collaborative research project in which a number of estuarine scientists, engineers and natural resource modellers were involved (CERM 1992). The objectives of the project included:

- establishing the state of decision support in terms of existing estuarine models, their data requirements, limitations, ranges of applicability and actual applications,
- investigating possible interlinkages between different models, establishing data and software requirements for the interlinkages and undertaking linkage of the models where useful and viable,
- identifying model development requirements in terms of existing models, improved interlinkages or new model formulation, and
- using a case study approach to test the application of this decision support module.

Details of the progress of the collaborative research project and the final results are contained in Slinger (1994, 1995 & 1996). However, the development of the linked modelling system and the application of the estuarine systems model within this broad integrative approach will be described briefly hereafter and the results of the project will be summarised.

5.4.1 The linked modelling system

At the inception of the project, it was unclear whether an integrated modelling approach was feasible. However, five models applicable to the determination of the freshwater requirements of estuaries existed, namely the estuarine systems model, the Mike 11 hydrodynamic and transport-dispersion model, the Mike 11 water quality module, the plant estuarine decision support system (PEDSSys) and the estuarine ecosystem evaluation model (EEEM). None of these models had been applied in a linked fashion before and most had not even been applied to the same estuarine systems. Furthermore, the different models were in various stages of development and were applicable over different temporal and spatial scales. The predictive capabilities of these models as well as those of the dynamic vegetation model (DVM), the development of which was initiated during the project, are listed in Table 5.4a.

Table 5.4a Characteristics of the estuarine models comprising the linked modelling system

MODEL NAME	STATE VARIABLES/OUTPUTS	TEMPORAL AND SPATIAL SCALES
Estuarine systems model (ESM)	Water volume, mean water level, mean salinity, axial salinity difference, stratification, circulation, freshwater flushing, tidal flushing, tidal flux, sill height at the mouth	Medium to long term (months to years); spatially indexed - detail in the mouth channel
Mike 11 hydrodynamic and transport-dispersion model	Water volume, water levels, flow velocities, volume fluxes, vertically-averaged salinities	Short to medium term (hours to a year); one dimensional (longitudinally)
Mike 11 water quality module	Vertically-averaged temperatures, dissolved oxygen concentrations, dissolved nutrient levels, bacterial numbers	Short term (hours to a month); one dimensional (longitudinally)
Plant estuarine decision support system (PEDSSys)	Growth rate adjustment scores for submerged macrophytes, emergent macrophytes and saltmarsh macrophytes, biomass ratings for phytoplankton and benthic microalgae	Short to medium term (days to a year); spatial scale - dependent on the physical input data
Estuarine ecosystem evaluation model (EEEM)	Fish recruitment index, a biomass productivity index and size class frequency histograms for the *Upogebia africana* (mudprawn) population	Medium to long term (months to years); no spatial detail
Dynamic vegetation model (DVM)	Biomass and productivity of submerged macrophytes (*Zostera capensis*, *Ruppia cirrhosa*) and an emergent reed (*Phragmites australis*)	Medium to long term (months to years); two-dimensional (cross-sectional and longitudinal)

Various types of linkage were envisaged, the primary being a flow of appropriately summarised data from the models simulating the abiotic environment of the estuary to those simulating the biotic environment of the estuary, so facilitating biological prediction. However, from the existing applications of the estuarine systems model and the Mike 11 hydrodynamic and transport-dispersion model to the Great Brak Estuary it was clear that data flow between complementary models was also possible. For instance, the Mike 11 hydrodynamic and transport-dispersion simulations were used in calibrating the tidal flux in the application of the estuarine systems model to the Kromme Estuary (Section 4.3), while the ability of the estuarine systems model to simulate the opening and closure of the estuary mouth could potentially be used to extend the applicability of Mike 11 to estuaries experiencing intermittent closure of the mouth. This complementary nature was formalised in the conceptual design of the linked modelling system in that data links were incorporated between the Mike 11 hydrodynamic and transport-dispersion model and the estuarine systems model. In a similar fashion, the plant estuarine decision support system and the dynamic vegetation model form a complementary pair of models and data links between these models were also incorporated. No such possibility of pairing long and short term predictive tools to form a complementary set exists in the faunal prediction component, because all of the available models have time horizons of months to years and are already incorporated in the estuarine ecosystem evaluation model. The linked system of estuarine models with essential data flows, model inputs and outputs (after Slinger 1996) is depicted in Fig 5.4a.

Note that the flow of data is not necessarily only one way (abiotic to biotic), but could be two way if biologists specified critical abiotic conditions for the communities/species under consideration and wanted information on how frequently such conditions could occur and the severity of the conditions under different management policies. Clearly, however, the whole system rests primarily on the predictions of the Mike 11 modelling system and the estuarine systems model - the basic building blocks.

5.4.2 **Implementation of the linked modelling approach**

As in the implementation of the estuarine systems model, the first step in applying the linked modelling system is the selection of case studies, next the calibration of each model individually using the best available data and then the selection of appropriate management policies or run-off scenarios. The case studies selected for the application of the linked modelling system were once again the Great Brak and Kromme Estuaries, owing primarily to:

- the availability of calibration data for both the abiotic and biotic models,
- the fact that at the inception of the project the Mike 11 hydrodynamic and transport-dispersion model had already been implemented on the Kromme Estuary and the estuarine systems model had been calibrated on the Great Brak Estuary,
- the good comparative basis provided by the differences in estuarine character between a permanently open estuary and an intermittently closed system, and
- the decisions on water releases and mouth breachings presently facing managers of these systems.

Each of the models was calibrated individually by the responsible member of the Consortium for Estuarine Research and Management (CERM). The run-off scenarios selected for testing were formulated in conjunction with CERM members, but primarily in association with Mr P Huizinga (*pers. comm.*). These comprised the majority of the management policies simulated by the estuarine systems model for the Kromme and Great Brak case studies (Sections 5.1.2 and 5.2.2), although the full range of episodic events and their effects on the Great Brak Estuary were not addressed in the linked modelling approach, but only the effects of a 1 in 50 year flood and a short dry spell in which the base flow declined by 50% from March to June. In both the Kromme and the Great Brak case studies the emphasis was placed on predicting the long term responses

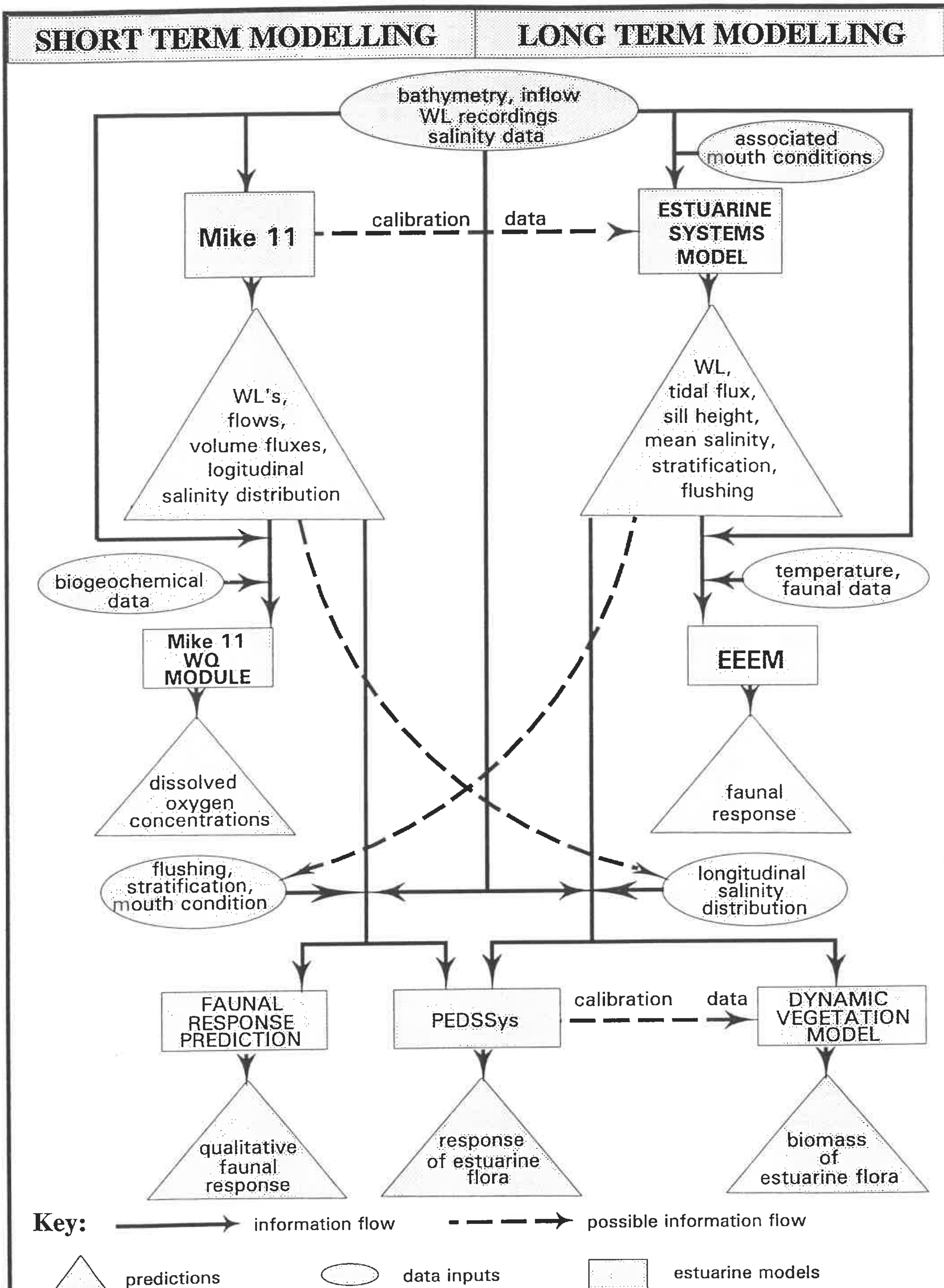

Figure 5.4a Linkages between existing estuarine models

of the abiotic environments in terms of the dynamic physical states occasioned by different management policies. The biological responses to these abiotic states were then predicted.

Data exchange

Critical to generating biotic predictions knowing the time dependent variations in the abiotic environment in response to various run-off scenarios, is the ability to sensibly summarise the abiotic data to the time scales appropriate for biotic response simulation. This was the most challenging aspect in making the linked modelling system operational as an holistic predictive tool.

A decision was taken by the researchers involved in the collaborative project not to implement software linkages in the transfer of data. Instead data exchanges were implemented by selecting appropriate temporal and spatial scales for the reduction of the abiotic data and then sending the reduced data files to the other team members over the Internet. Accordingly, the data from the estuarine systems model were summarised and transferred to the biotic models in the form of an ASCII data file containing:

- a daily inflow rate,
- a daily minimum and maximum water level,
- daily averages of the mean salinities, stratification index values, flushing rates, and sill heights at the mouth, and
- daily maximum tidal fluxes and current velocities through the mouth.

Using these data, predictions of the biotic responses to the run-off scenarios were undertaken for the Kromme and Great Brak case studies by Dr J Adams, Prof G Bate, Ms J Busse, Prof J Hearne and Mr N Quinn. Their results are summarised subsequently.

Biological predictions: the Great Brak case study

The physical data provided by Mike 11 was effectively used by PEDSSys to predict the distribution of macrophytes along the length of the estuary, whereas the information provided by the estuarine systems model was useful in predicting the effects of the mouth condition on the plants.

Under the natural and pre-dam situations, the plants responded similarly to open mouth conditions and all plants obtained maximal growth rate adjustment scores of 10 for both the long term average dynamic states and the 1 in 50 year flood disturbances. Although extremely high water levels occurred during the floods, these did not persist for long enough to cause extensive die-back. Open mouth conditions ensured good tidal flushing, which is essential to the functioning of the intertidal saltmarshes, and hence an open mouth benefitted the estuarine flora. Under the post-dam run-off scenarios, the duration and frequency of open mouth conditions were reduced and salinities in the estuary increased owing to reductions in base flow. Accordingly, PEDSSys predicted reduced vigour in the brackish species such as *Sporobolus* and *Phragmites*. The detrimental effects of high salinities under open mouth conditions were most evident under the post-dam scenario 3A and 3B dry spell disturbances (50% reduction in base flow from March to June). Water levels decreased to less than 0,7 m to MSL, causing exposure of the submerged plants and desiccation of the marsh sediments. All plants except *Sarcocornia* obtained the lowest possible growth rate adjustment scores of -10. PEDSSys predicted a score of -3,3 for *Sarcocornia*, because it can tolerate dry sediment conditions better than other estuarine plants.

When the mouth was closed under the natural scenario, the pre-dam scenario and the post-dam run-off scenario 3A, the water levels in the estuary exceeded 0,9 m to MSL and the emergent macrophytes and salt marsh plants were inundated. As prolonged inundation is harmful, particularly during the growing season, this affected the emergent macrophytes and marsh plants detrimentally. The closed mouth situation also adversely affected the *Zostera* beds, owing to high water levels and increased turbidity. Under the post-dam run-off scenario 3B, however, the water levels were lower than for the run-off scenarios 1, 2 and 3A. Less than 25% of the emergent plants were inundated and the plant growth rate adjustment scores increased. Thus PEDDSys showed that closed mouth conditions were detrimental to the estuarine flora, apart from under the post-dam run-off scenario with base flows of 2×10^6 $m^3.yr^{-1}$ when inundation alleviated the dry conditions of the substrate.

The PEDSSys predictions of the response of the estuarine flora concur broadly with the conclusions from the application of the estuarine systems model in that:

- the essential character of the estuary is maintained under the pre-dam scenario and the post-dam run-off scenario 3A, whereas this is not the case under scenario 3B,
- the 1 in 50 year flood disturbances are deemed beneficial to the system, and
- the flora are sensitive to further reductions in base flow and the associated reductions in water levels and increases in salinities (enhanced marine-dominance), yielding the lowest possible growth rate adjustment scores under these conditions.

Although PEDSSys has indicated the negative effects of prolonged closure of the mouth, it cannot assess the effects of repeated opening and closure of the mouth on the estuarine plant communities. This information is provided for three plant species by the dynamic vegetation model.

The dynamic vegetation model simulated the theoretical equilibrium biomass and the total average yearly productivity of *Zostera capensis*, *Ruppia cirrhosa* and *Phragmites australis* under the given abiotic conditions. The water level and salinity data supplied by the ESM were assumed to represent conditions in the middle reaches of the estuary. Results are presented in Table 5.4b. The first entry in Table 5.4b means that *Zostera* occupies 0% of the area modelled, *Ruppia* 27% of the space and *Phragmites* 43% of the space, where the total density of occupied space is 70%.

Table 5.4b Predictions of the equilibrium biomass density (g.m^{-2}) and the total average yearly productivity (g.m^{-2}) under the run-off scenarios 1, 2, 3A and 3B (*Zostera/ Ruppia/ Phragmites*)

Run-off Scenarios	1	2	3A	3B
Biomass Density	0/ 0.27/ 0.43	0/ 0.29/ 0.36	0/ 0.42/ 0.36	0.19/ 0.16/ 0
Productivity	0/ 5/ 121	0/ 5/ 112	0.001/ 11/ 94	4.4/ 3.5/ 0

Under the natural situation, the pre-dam situation and the post-dam run-off scenario 3A, no Zostera was present in the middle reaches of the estuary owing to the low salinities occasioned by reasonably high base flows. However, under scenario 3B, the salt tolerant *Zostera* had encroached into the middle reaches and occupied 19% of the space, while the freshwater reed, *Phragmites*, was displaced completely. *Ruppia* was present under all run-off scenarios, because it has a wide salinity tolerance range. It exhibited its highest average yearly production and equilibrium biomass density under run-off scenario 3A primarily because the prevailing salinities affected *Phragmites* adversely, yet were insufficient to support *Zostera* i.e. there was reduced competition. Additionally, there was an overall reduction in the productivity (as predicted by PEDSSys) and percentage of space occupied with reductions in freshwater flow.

The results agree with predictions produced for five positions along the length of the estuary using Mike 11 data, which indicate that under natural and pre-dam conditions the middle reaches of the estuary are dominated by brackish species. Following the construction of impoundments, stable sediment conditions and enhanced salinities have led to an encroachment of marine species into the middle reaches and a general reduction in productivity.

The response of the plants to the flood disturbances and the dry spells simulated by the ESM (but not by the Mike 11 model) reveal that the floods exert a marked effect, substantially reducing the biomass of *Ruppia* and *Phragmites* under all run-off scenarios as well as that of *Zostera* which only occurs in the middle reaches under scenario 3B. However, the productivities of the plants increase following the floods indicating that the floods act to re-set the system and agreeing with the PEDSSys predictions in this regard. The dry spell causes the biomass and the productivities of the plants to decline, exerting a detrimental effect which the system takes about a year to recover from under all run-off scenarios.

In contrast to the floral prediction component, where both the Mike 11 and the ESM simulations of the abiotic environment were used in a complimentary fashion to run both the PEDSSys and DVM models, the faunal prediction component only uses the ESM results. The production level of the estuarine mudprawn, *Upogebia africana*, is highest under the natural run-off scenario and all size classes are present throughout the eight year simulation period. The prawns reach a maximum carapace length of 25 mm. Under the pre-dam situation, the equilibrium biomass attained is 85% of that under the natural situation and some gaps appear in the size class frequency histograms reflecting periods when the mouth was closed and no recruitment was possible. Under the post-dam run-off scenarios 3A and 3B, the equilibrium biomass reduces to between 25% and 35% of the natural situation, large gaps occur in the size class frequency histograms and the prawns only attain a maximum carapace length of 22 mm.

Under the natural run-off scenario, the fish recruitment index attains a value of 96 out of a possible 100, owing to the strong head to mouth salinity differences. This situation persists under the pre-dam run-off scenario, so recruitment occurs at maximum potential apart from during mouth closure periods. A score of 89 is therefore attained. Under the post-dam scenario 3A, the mouth closes more frequently and the axial salinity differences are reduced, but still exceed 20 $kg.m^{-3}$ for reasonable time periods. A mean annual score of 43 is attained. Although closure of the mouth occurs as in scenario 3A, the severe reduction in axial salinity differences associated with the low base flows of scenario 3B causes the fish recruitment index to decline to 8, one twelfth of the value under the natural run-off scenario.

Clearly, the biological predictions, while not including many elements of the ecosystem, confirm the earlier findings on the abiotic environment (Section 5.1) and indicate that an altered estuarine state minimally representative of the natural condition of the estuary will result should the $2 \times 10^6\ m^3.yr^{-1}$ annual allocation to the estuary be the only freshwater entering the system. Indeed, the biological predictions indicate that increased marine dominance in the abiotic environment will result in an altered biotic environment with salt tolerant plant species spreading through the estuary, reduced fish abundance and diversity and fewer, smaller mud-prawns.

Biological predictions: the Kromme case study

In the case of the permanently open Kromme Estuary, PEDSSys predictions were undertaken using Mike 11 simulations with constant freshwater flows characteristic of the different inflow scenarios as the upstream boundary conditions. The results indicate that the most important physical conditions for the estuarine flora are fluctuations in salinity and the existence of axial salinity differences, rather than the current velocities or water levels. The starting condition for predictions was taken as the present situation in which the seagrass, *Zostera*, occurs throughout the estuary and there is no longitudinal distribution of macrophytes from species tolerant of high salinities in the mouth area to brackish species in the upper reaches. PEDSSys predicted a negative growth rate adjustment score for the seagrass, *Zostera,* and a positive score for the brackish reed, *Phragmites*, under the natural run-off situation and the intermediate run-off scenario (40% of MAR), that is a tendency towards restoration of a longitudinal distribution of macrophytes. In contrast, for the situation in which no freshwater reached the estuary, PEDSSys predicted a growth rate adjustment score of 10 for *Zostera*.

In contrast to the Great Brak case study, *Ruppia cirrhosa* was omitted from consideration, because it cannot survive regular exposure or strong currents such as occur in the Kromme Estuary. However, the average equilibrium biomass and the productivity of both *Zostera capensis* and *Phragmites australis* were predicted by the dynamic vegetation model in response to the abiotic conditions simulated by the estuarine systems model. The reference conditions were taken as the present situation. A marked decline in the average productivity and the average equilibrium biomass of *Zostera* was predicted for the natural situation and the intermediate run-off scenario (40% of MAR), whereas no significant differences were predicted under the run-off scenarios 3, 4, 5B, 5C, 5D, 5E and 6. These results concur with the PEDSSys predictions and indicate that Zostera would not have colonised all areas of the estuary under natural or near-natural conditions, but would have been excluded from the upper brackish reaches. *Phragmites* would have occupied these reaches and would not have been displaced from the middle reaches as is the case at present.

As the mouth of the Kromme Estuary is open throughout the year, the opportunity for faunal recruitment is always present. This is reflected in the simulation results for *Upogebia africana*, which do not show marked differences between the different run-off scenarios, although there is a general decreasing trend in the invertebrate production index values with reductions in freshwater flow (Table 5.4c). As the mudprawn is sensitive to salinities exceeding 34 $kg.m^{-3}$ and extremes of temperature, the lack of significant differences may be an artefact of the omission of temperature effects from consideration and the inability of the ESM to predict hypersalinities. In contrast, the fish recruitment index indicates strong differences in response to the abiotic conditions under the different run-off scenarios. Although there are opportunities for 100% recruitment under all run-off scenarios apart from 5A (the present situation), these do not persist for long periods under any except the natural situation. Thus fish recruitment in the Kromme Estuary is severely impacted by reductions in freshwater flow, declining by 45% when the freshwater flow to the estuary decreases by 60% and dropping by between 83% and 88% when the run-off to the estuary decreases by 90% to 98%. Thus both the invertebrate production index and the fish recruitment index concur in predicting that the response of the fauna naturally present in the Kromme Estuary to the abiotic environments resulting from reduced freshwater flows (as predicted by the ESM), is adverse.

Table 5.4c Summary of the results of the estuarine ecosystem evaluation model (EEEM)

Run-off Scenarios	Invertebrate Production Index		Fish Recruitment Index	
	mean	*maximum*	*mean*	*maximum*
1	47.66	91.58	87	100
2	43.37	83.79	48	100
3	40.91	79.46	18	100
4	40.18	78.11	15	100
5A	39.41	76.52	10	10
5B	39.47	76.68	12	100

The production of meaningful floral and faunal response predictions for the permanently open Kromme Estuary, therefore, was facilitated by output from the estuarine systems model.

5.4.3 **Summary of the linked modelling applications**

Predictions of the response of components of the ecosystems of the Great Brak and Kromme Estuaries were facilitated by the provision of appropriately reduced data on the physical dynamic states of these systems as simulated by the estuarine systems model and the Mike 11 hydrodynamic and transport-dispersion model. Although, only an indicator invertebrate species and an index of fish recruitment were used in the prediction of the faunal response, the simulation results confirm the estuarine systems model predictions of more uniform conditions with enhanced marine dominance. Fish abundance and diversity is reduced with more marine species predominating and the mudprawn stocks decline, even exhibiting reduced size in the intermittently closed Great Brak Estuary under low freshwater flows. These results are confirmed further by the floral predictions which indicate a reduction in macrophyte diversity along the length of both the Kromme and Great Brak Estuaries with decreasing freshwater supply. This has already occurred in the Kromme Estuary in response to freshwater starvation over the last ten years. The seagrass Zostera has flourished, increasing in areal cover by a factor of 1.6 (Adams & Talbot 1992). However, the total equilibrium biomass of the flora under low freshwater flows is reduced compared with the natural situation. This is also true for the faunal components, indicating a general decrease in the productivity of the estuarine ecosystem. As estuaries are amongst the most productive ecosystems in the world (Odum 1971, Clark 1974, Branch & Branch 1981, Knox 1986), the shift towards more marine dominance and reduced productivity is a very serious concern.

An interesting result of the integrated modelling application is that the faunal indices, the quantitative DVM predictions and the PEDSSYs growth rate adjustment scores for both the Great Brak and Kromme case studies show the same rank-ordering of the run-off scenarios as obtained using the management performance indices of the estuarine systems model. This concurs with the statement by Jezewski and Roberts (1986) that the ecosystem reflects a secondary response to the hydrodynamic and geomorphological character of the estuary, although the relative ratings of the effects differ between the abiotic and biotic predictions as well as between the different biotic components, owing to the non-linearity of the responses. But the coherence between all the outputs indicates that the management performance indices were effective in evaluating the consequences of the run-off scenarios on the abiotic environment, and by implication on the biotic environment.

However, the integrative application of the ESM in a linked set of models is clearly still in its infancy. Further models need to be developed to enhance the range of species and ecosystem components for which predictions can be made. Additionally, the reverse flow of data and information needs to become operational, that is the prediction of the frequency, severity and duration of physical conditions critical in determining biotic responses. This should occur naturally as the biological models and the predictive expertise of ecologists improve. However, at this stage a major contribution from this study is its revelation of the complementary use of the estuarine systems model and standard hydrodynamic modelling in predicting the physical environment of estuaries in a way that is appropriate for biological prediction. For instance, Mike 11 data were used in the simulation of the longitudinal differences in the biomass of plants, whereas ESM predictions were used to accomodate the fluctuations in mouth state and the associated effects on water levels and salinities. Without the latter information, no long term biological response predictions could have been made for an intermittently closed estuary. This is not the case for a permanently open system, where the Mike 11 system could be used for more detailed spatial prediction of the floral response. However, in the Kromme case study the ESM facilitated the prediction of the faunal response by providing suitable information on the axial salinity differences caused by different freshwater flow rates.

Besides the paucity of biological models and the present uni-directional flow of data, the linked system of models comprising the integrative management application is generically applicable to South African estuaries. The strength of the fundamental idea of using the estuarine systems model and the Mike 11 hydrodynamic and transport-dispersion models as building blocks in a linked set of models has been demonstrated in that a level of biotic response prediction not previously possible has occurred. Further developments, particularly in the biotic and water quality components of the linked modelling system, are anticipated in future.

5.5 Strategic Management Applications

The applications discussed thus far have focussed on management decision making on a site specific basis. However as mentioned in Section 1.2, some of the issues facing South African estuaries are related to regional development of water resources and/or coastal areas. Thus there is a management requirement for tools to support decision making at a regional or national level. This need was recognised in 1992 by the Consortium for Estuarine Research and Management (CERM 1992) and formed a component of the collaborative research undertaken in the subsequent three year period. This research component focussed successfully on developing a system to rate the conservation value of estuarine resources and their importance for human use as a recreational amenity, a site for coastal development and a biological resource (bait collection, protein source) (Quinn 1996). The rating system does not address decision making regarding future resource utilization although it could be used to rate the conservation value and importance for human use of a predicted future state. However, common questions facing water resource planners include "which system of a number along a particular section of coast can best be utilized for water abstraction?" or "would a combination of development options on more than one system be more beneficial for both the environment and water yield?". While neither the estuarine systems model nor the linked modelling system can supply answers to such questions directly, they can supply information on which sound decisions in this regard could be based.

In view of the fairly robust nature of the estuarine systems model with regard to uncertainties in parameter values, the ESM could potentially be used to provide an indication of likely estuarine behaviour based on readily available data of limited accuracy. In particular, the estuarine systems model can utilize basic information on:

- hydrology (percentage of the MAR reaching the estuary, seasonality of the inflow, the volumes and discharges of large floods, low flow conditions),
- mouth condition (Whitfield characterisation, characteristic mouth depth, the duration and frequency of mouth closure, artificial breaching occurrence and particulars)
- topography (average depth, width, length of the estuary)
- sea conditions (tidal range, average water level fluctuations within the estuary, wave conditions at the mouth), and
- thermohaline structure (characteristic stratification-circulation state, head to mouth differences in salinity, the occurrence and persistence of hypersalinities),

to predict the range of states the estuary can occupy, that is its characteristic dynamic behaviour and how this could alter with alterations in freshwater supply to the system. As this level of information is often readily available, the ESM could be used as a screening tool at the pre-feasibility phase of water resource developments to classify those estuaries that appear least sensitive to freshwater abstraction. These estuaries could then be selected for thorough, site specific investigation and monitoring. In this way, the estuarine systems model could facilitate strategic decision making based on the sensitivity of the abiotic environment of an estuary to reductions in freshwater supply and could direct further research efforts more cost effectively, so enhancing water resource planning and improving the conservation of our coastal resources.

An example of a potential application of the ESM at a strategic level is the planning of water resource developments in the central Natal-KwaZulu area. A number of systems in this area have been earmarked as potential sources of water to the KwaZulu-Natal area and as supplementary supplies to the Gauteng region. These include the Mkomaas River in the south, the Illovu River, the Mgeni River, the Mhloti River, Mvoti River and the Tugela River in the north (Mr K Legge *pers. comm.*). Of these systems, only the Tugela and the Mkomaas estuaries are characterised as permanently open. Although the other four systems are all characterised as temporarily open estuaries, they exhibit very different mouth behaviour. On average the Illovu is closed about 25% of the time, the Mhloti 75% of the time, the Mvoti 15% of the time and the Mgeni 45% of the time at present (Mr P Huizinga *pers. comm.*). As the permanently open systems are those with the largest natural freshwater flows, it is tempting for water resource planners to opt for development options on these systems rather than on the four smaller rivers. In fact, the location of the Mgeni River along a development corridor in the region means that it has already suffered the consequences of extensive development. However, the Natal-KwaZulu coastal sea is a nutrient-poor area and the offshore fishery and prawn stocks are maintained by supplies of turbid, nutrient-laden river water from the permanently open estuaries and pulses of these waters when the temporarily closed systems breach (Dr A Connell *pers. comm.*). To curtail or even reduce these significant supplies to the coastal ocean or the sediment supplies to the coastal beaches of northern Natal-KwaZulu could have consequences far outweighing the advantages to be gained by utilizing these seemingly better options for dam developments. In this context, the application of the estuarine systems model at a screening level could well indicate that the consequences of increasing the frequency and duration of mouth closure of some or all of the four smaller systems would have less severe effects. In fact, the Mvoti Estuary which has a protected mouth may not even suffer severe consequences if certain proposed dam development options, incorporating water releases for ecological purposes, are implemented on the system (Mr P Huizinga *pers. comm.*).
The long term use of such a screening approach is limited, because more accurate site-specific predictions incorporating biotic considerations will become possible in future as data in this regard become available. However, at present the strategic management application of the estuarine systems model may well contribute cost-effectively to the sustainable development of the water resources of South African and the maintenance of the character and diversity of the estuaries of the coastal zone.

This completes the implementation of the management evaluation component of the model.

6. DISCUSSION AND CONCLUSION

6.1 Implications for Estuarine Science and Management

The most pressing issue facing South African estuaries is the growing demand for freshwater in the semi-arid southern African environment (DWA 1986, Whitfield & Bruton 1989, CERM 1992, Whitfield & Wooldridge 1994). This is leading to increased abstraction in the catchments for agricultural, domestic and industrial uses and causing a number of detrimental effects in downstream estuaries. These effects range from geomorphological changes such as sedimentation and an increase in the incidence and duration of mouth closure (Reddering 1988, CSIR 1990a, 1992b, 1992c, 1994b), through hydrodynamic effects (altered currents, salinities and water column structure (Allanson & Read 1987, Reddering 1988, Slinger & Largier 1990, Taljaard & Slinger 1993, Slinger *et al.* 1994)) to alterations in the ecosystem structure and functioning of the estuary (Whitfield & Bruton 1989, Plumstead 1990, Baird & Heymans 1994, Adams & Bate 1994, Wooldridge 1994, Allanson & Read 1995). Associated, but largely unquantified effects in the coastal seas also result (Whitfield & Wooldridge 1994, Dr A Connell *pers. comm.*). Despite documenting the effects on estuaries and speculating on their long term consequences, very few scientists have attempted to quantify the degree of change occasioned by anthropogenic influences as opposed to the changes which would occur naturally as the estuaries of South Africa evolve from drowned river valleys to coastal lagoons over geological time. It is interesting and challenging to explore the contribution that the systems modelling approach can make in this regard.

6.1.1 Systems modelling and successionary trends in estuaries

Despite the ephemeral nature of estuaries in geological time, estuaries exhibit only a weak successionary trend from deeply incised, drowned river valleys to shallow, marine dominated, coastal lagoons. Instead of showing the marked trend of more stably cycling systems towards increasing proportions of specialised plants and animals with strong interspecific connections (Bruton 1989), estuaries naturally exhibit wide fluctuations in biotic and abiotic conditions with innumerable small scale successions superimposed on the weak overall succession. Thus they constantly move back and forth along a continuum of successionary states with the return to an earlier successionary state occurring as a result of a flood (Whitfield & Bruton 1989).

The question remains as to the degree of human influence on the natural successionary trend. Whitfield and Bruton (1989) commented extensively in this regard for South African estuaries, building on the findings of Reddering (1988) that reduced flooding owing to upstream impoundments causes enhanced sedimentation immediately within an estuary mouth, which in turn leads to increased flood tidal dominance and reduced flushing by ebb tidal currents. The weaker ebb tidal currents allow the sediments of the flood tidal delta to stabilize and become more resistant to scour by floods than would have been the case in the natural situation. This interplay of factors means that the estuary tends to shallow and expedites its path along a trajectory towards enhanced marine dominance i.e. the successionary state of a tidal lagoon. As freshwater deprivation continues this trend is enhanced with tidal flushing becoming dominant in the renewal and exchange of estuarine water in some systems rather than the natural flushing process of freshwater flooding (Slinger *et al.* 1994). Increased upstream dispersion of salt may occur (Slinger & Taljaard 1994) and hypersalinities can result (Whitfield & Bruton 1989). Biotic diversity declines with the loss of brackish water habitats such as *Phragmites* beds and the decrease in cues for fish migrating into the estuary. Whitfield and Bruton (1989) and Whitfield and Wooldridge (1994) emphasise that flooding is of major importance to the sustained functioning of estuaries and contend that it is particularly the smaller floods, those most affected by impoundments, which are required to regularly re-set the systems and stimulate biotic production. In the absence of these freshwater influxes, Whitfield and Wooldridge (1994) suggest that estuaries could be forced into extreme ecological states eg. prolonged hypersaline periods, with deleterious consequences for normal processes within the systems. While the underlying implication that the natural state was one in which water supplies were more plentiful is arguable (Allanson & Read 1995), it is unquestionable that reduced intensity and frequency of flooding enhances the progression of an estuary towards marine dominance.

The role of freshwater flooding in re-setting an estuary to an earlier successionary state was clearly demonstrated and quantified in the Great Brak case study (Section 5.1.4). Under natural or pre-dam inflow conditions, the 1 in 50 year flood caused the estuary to enter a 'new' dynamic state in which the sill at the mouth deepened and enhanced tidal exchange occurred i.e. the estuary exhibited characteristics of a deeper, drowned river valley or an earlier successionary stage. In the case of the Great Brak Estuary, the influence of man in reducing freshwater supplies to the estuary further (the post-dam situation) meant that the 1 in 50 year flood no longer had the same degree of influence. Although the sill at the mouth decreased in height and tidal flows increased, these effects where reduced in magnitude and duration. Thus the system was more resistant to the flood event and the capability to return to an earlier successionary state was reduced.

This implies that the natural range of oscillations of the Great Brak Estuary has been reduced and the full variability of the system is being lost. Whitfield and Bruton (1989) and Odum *et al.* (1995) associate this with an overall decrease in the productivity of the ecosystem and a reduction in species niches.

In general, an ecosystem is most productive when the biotic interrelationships are adapted to utilize the fluctuations of the dominant physical forcing i.e. the biotic system is tuned to the frequency of the dominant forcing (Blum 1995, Odum *et al.* 1995). Estuaries are excellent examples in this regard because the fauna and flora utilize the tidal forcing to aid their feeding, behaviour, life histories and migration (Grindley 1972, 1981, Day 1981, Hilmer & Bate 1991, Whitfield & Kok 1992, Adams & Bate 1994, Wooldridge 1994) and so increase system productivity. High frequency forcing usually is absorbed by the ecosystem, giving rise to small scale variations and microdiversity eg. microbial oscillations, and so increasing productivity (Odum *et al.* 1995). Low frequency forcing or episodic events, on the other hand, generally act to re-set the system to an earlier successionary state and often stimulate productivity (Odum *et al.* 1995).

Although the scouring response of the Kromme Estuary to large floods could not be simulated using the estuarine systems model (a limitation arising out of the assumptions of the model), it is evident that the Kromme Estuary is marine dominated at present. The high mean salinity, uniform water column, frequent occurrence of hypersalinities and the benthic basis of the food chain (Baird & Heymans 1994) are evidence of this condition. In fact, Baird and Heymans (1994) demonstrated that productivity in the Kromme Estuary declined from the 1984 pre-impoundment situation to the 1992 situation of freshwater deprivation. Thus in both the Great Brak and Kromme case studies, anthropogenic reductions in freshwater supply (reduced base flows and reductions in the frequency and intensity of flooding) cause the systems to exhibit reduced pulsing, enhanced marine dominance and reduced productivity.

The overriding contribution of the estuarine systems modelling approach, therefore, is the verification and quantification of noted effects of freshwater deprivation, namely increased mouth closure and resistance to scouring by floods, associated increased salinities and limited variation in water column structure. Additionally, the prediction and quantification of certain of the biotic responses induced by these altered abiotic conditions is facilitated. For small temporarily open systems, the quantification of the primary effects on the mouth is particularly relevant, while for permanently open systems the effects on the salinity structure are of interest. In each case, the range of states a system could occupy can be investigated. However, this is not the real issue, because a flood of sufficient magnitude could probably cause an estuary to exhibit extreme behaviour i.e. the system could still occupy its full range of states, but the magnitude of the forcing required to induce this can increase and the frequency of its occurrence decrease. The estuarine systems model has the capability to simulate these effects and allow quantification of the percentage of time spent in more usual states as opposed to extremes. For instance, the increased resistance to freshwater flooding and the enhanced responsiveness to marine forcing under lower freshwater flows, means that the natural dynamism of the Great Brak Estuary has been reduced, rather than the range of states curtailed.

So, the case study results from the estuarine systems model applications to the Great Brak and Kromme Estuaries have confirmed many of the qualitative observations of Reddering (1988), Whitfield and Bruton (1989) and Whitfield and Wooldridge (1994) of the tendency of anthropogenic reductions in freshwater flow to hasten the progression of South African estuaries towards marine dominance, a situation which Odum *et al.* (1995) associate with reduced ecosystem productivity.

Perhaps the quantifiable evidence provided by the estuarine systems model simulations will add weight to the arguments of estuarine scientists that to allow this tendency to become true of the majority of South African estuaries would be to fail to preserve biotic diversity. More thoughtful approaches to estuarine management and water resource developments are required, in which models are used to predict the stages at which a sustained loss of natural dynamism and variability will occur and a faster progression to the later successionary stage of a coastal lagoon result.

6.1.2 Systems modelling and management decisions

Whether the philosophy behind the concerns regarding the influence of man in expediting estuaries towards the later successionary state of coastal lagoons is appreciated by managers or not, the provision of the estuarine systems model, a practical, scientifically-based simulation tool, should encourage informed management decisions.

At the inception of this study, the only formal predictive capability in estuarine science in existence in South Africa was a standard one-dimensional hydrodynamic and transport-dispersion model (Huizinga 1985). The results produced using this model were useful, particularly in determining flood levels, addressing circulation problems and in predicting salinity distributions under constant flows and mouth conditions. However, as the demand for freshwater increased and more of the effects of reduced flows on geomorphology, hydrodynamics and biology were observed, the need for an alternative modelling approach specifically designed to accomodate characteristic features of South African estuaries became evident and the current study was initiated.

The first objective of the study, namely to develop a model adequately representing the physical dynamics of South African estuaries over time scales from months to years and so enabling prediction of the medium to long term consequences of alterations in the fresh water inflow on the physical functioning and character of an estuary, was achieved in the formulation and successful implementation of the estuarine systems model. The basis of the model formulation is phenomenological rather than analytical and so the model is semi-empirical with a sound basis of physical arguments. The model was implemented on the Great Brak Estuary, a temporally open system, and on the Kromme Estuary, a permanently open system, and water level variations, mean salinities, flushing, stratification-circulation state, the scour of the mouth and associated fluxes of salt and water through the mouth were simulated. The empirical Ackers-White sediment transport formula (Ackers & White 1973), which is used to calculate scour at the mouth of the estuary, was modified slightly for low flow conditions to yield results in agreement with field observations. Reference simulations of the natural conditions in the estuaries were conducted and the sensitivity of the model outputs to uncertainties in parameter values was determined. One percent alterations in individual parameters resulted in maximum normalised sensitivity values of less than 0,043 throughout, lending a fair degree of confidence to the model results.

Additionally, two known non linear features of estuarine behaviour, namely the generation of overtides and the modulation of the mean water level within a shallow estuary by low frequency marine forcing (Aubrey & Speer 1985, Brundrit *et al.* 1988, MacKay & Schumann 1991), are well simulated by the estuarine systems model, although they were not explicitly included in the formulation of the tidal exchange through the mouth. This is possibly the most encouraging aspect of the whole study, because it lends further veracity to the formulation of the physical dynamics component of the model.

The second objective, namely to enable evaluation of the efficacy of management policies in the maintenance of estuarine character and functioning, was addressed when a range of management policies involving freshwater allocations, water releases and mouth breachings were selected for testing on both the Great Brak and Kromme case studies. The medium to long term effects of reductions in freshwater supply were to cause the estuaries to exhibit less variability, more uniformity and more marine dominance. In the Great Brak Estuary, some amelioration of the effects of freshwater abstraction can be obtained in the post-dam situation by imposing a policy of intermittent freshwater releases (as opposed to base flows) accompanied by breaching of the mouth at a level of between 1,62 and 1,85 m to MSL. In the Kromme Estuary, the present annual allocation represents a severe alteration in state from the natural system. However, some amelioration of the uniform head to mouth salinity conditions can be attained by the release of the total annual allocation as one large release. An optimisation of this policy of one annual flood of 2×10^6 m^3 was undertaken and revealed that a flood in mid to late September for between 3 to 7 days is most beneficial under average rainfall and evaporation conditions.

To facilitate a more comprehensive understanding of the freshwater requirements of estuaries (an overall objective of the study), it was necessary to examine the response of the biota to predicted alterations in abiotic conditions. The formulation of the estuarine systems model and its capability in predicting the physical dynamics of estuaries, especially those experiencing intermittent closure of the mouth, acted as a catalyst in the development of a linked system of physical and biological models. The estuarine systems model was used as a building block in this more holistic predictive approach (Slinger 1994, 1995, 1996). Data exchanges were undertaken at temporal and spatial scales appropriate to biotic prediction. Results indicate the consequences of management policies for a range of subtidal, intratidal and supratidal floral communities and for components of the fauna such as the estuarine mudprawn and various categories of fish. In the temporally open system, a one-dimensional hydrodynamic and transport dispersion model was used for the prediction of spatial differences in salinity under open mouth conditions, whereas the estuarine systems model simulated the alterations in mouth condition and so could supply data on concurrent changes in water levels and salinities. This facilitated biotic prediction and the data from the estuarine systems model was used extensively. This was not the case in the permanently open system, where floral predictions were made largely on the basis of spatial outputs from the standard one-dimensional hydrodynamic and transport dispersion model. However, the faunal models were reliant on the data from the estuarine systems model, producing fish recruitment indices and invertebrate production indices in response to the abiotic conditions under various management policies. All results concurred in the rank-ordering of scenarios with those of the management performance indices selected in the management evaluation component of the estuarine systems model. Thus, the complementary use of two abiotic models has facilitated the prediction of the biological response of estuaries to alterations in freshwater flow.

Thus the direct outcome of the research project is the formulation of an estuarine systems model which can be used to simulate the physical dynamics of South African estuaries in the medium to long term and to evaluate the advantages and disadvantages of different management policies, on the basis of the effects on the abiotic and the biotic environments and its implementation on two case studies. However, the indirect outcomes include:

- a thorough understanding of the physical dynamics of the small, bar-built estuaries typical of a semi-arid environment with a micro-tidal, high wave energy coast,
- an enhanced appreciation of the effects of anthropogenic reductions in freshwater flow in expediting the tendency of South African estuaries towards marine dominance,
- the stimulation of biological modelling and prediction, and
- a general enhancement in the level of management decision support provided by estuarine science in South Africa.

6.2 Limitations and Potential Improvements

The majority of the limitations to the applicability of the approach arise from assumptions made in the formulation of the estuarine systems model and have already been mentioned. These include the inability of the model to simulate the effects of large floods on the Kromme Estuary, because morphological changes other than those in the mouth channel were specifically omitted in the formulation. This choice even played a role in the selection of a spatially-indexed modelling technique, rather than one which accommodated spatial effects. A further limitation of the model is its inability to simulate hypersaline conditions. This does limit the applicability of the model in the South African situation, because hypersalinities are often of major concern. However, the use of both the estuarine systems model and a standard one-dimensional hydrodynamic transport-dispersion model in tandem would enable the propensity of the system to exhibit hypersalinities to be predicted in the medium to long term.

Potential improvements to the model and its implementation include:

- investigating ways in which the model could be developed to address the intensity and duration of hypersaline events,
- providing further support to biological prediction, possibly by automating the data reduction and transfer process, and
- developing the strategic implementation of the model in water resource planning further and establishing confidence limits which would need to be placed on results based on parameter uncertainties.

Additionally, the prediction of the stratification-circulation state as undertaken in the estuarine systems model yields valuable information on the state of the water column in the estuary. The potential exists to add this type of prediction to the vertically-averaged salinities predicted by standard one-dimensional hydrodynamic and transport-dispersion models along the length of the estuary i.e. provide quasi-two-dimensional salinity predictions. These suggestions could well form the basis of future research projects aimed at further improving the level of decision support available to estuarine and scientists managers.

Other improvements in estuarine decision support have also become possible since the inception of this study, because estuarine monitoring data are increasingly available. For instance, a complete six year record of inflow, water level, mouth condition and observed sea state is now available for the Great Brak Estuary. Similarly, information on the inflow and associated mouth condition have recently come to light for the Mvoti Estuary. Such data also exists for a number of small estuaries on the Natal-KwaZulu coast. Statistical analyses aimed at establishing the probable state of the estuary mouth given antecedent inflow and wave conditions could therefore be undertaken. In some cases this could yield predictions of mouth state more rapidly and easily than by using the estuarine systems model and should therefore be investigated.

6.3 Conclusions

The objectives of the study have been achieved in the provision of a practical, scientifically-based estuarine systems model, the management applicability of which was demonstrated in its implementation on two case study estuaries. Analysis of the model results has led to enhanced insights and understanding of the physical dynamics of estuaries including the processes involved in the closure and breaching of an estuary mouth and the enhanced sensitivity to marine forcing which results from reduced freshwater influence. The role of human activities in speeding the succession of South African estuaries towards marine dominated states which are more resistant to freshwater flooding and exhibit reduced variability and natural oscillation has been highlighted. The fact that this situation is associated with reduced biological productivity is a serious concern.

Although the estuarine systems model was instrumental in stimulating biological prediction for estuaries, this field of science is still in its infancy in South Africa. Thus, besides providing abiotic predictions, the estuarine systems model has acted as a stepping stone to integrated decision support for estuarine management. It is anticipated that this study will continue to provide a basis from which further improvements in estuarine modelling and decision support can be made, particularly at the level of strategic water resource planning in South Africa. The estuarine systems modelling approach could also prove applicable to some of the small estuaries on the sandy, high wave energy coasts of California, Mexico, Western Australia, Malaysia and Indonesia, which are known to exhibit intermittent mouth closure, and so contribute to estuarine management both locally and globally.

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APPENDIX A

MODEL EQUATIONS

A1 PHYSICAL DYNAMICS COMPONENT

Auxiliary and Rate Equations

$x_{11} = \mathrm{IF}(t)$

$x_{12} = [\mathrm{RF}(t) - \mathrm{EF}(t)].\mathrm{SA}(x_1)$

$x_{13} = \mathrm{GF}(t)$

$\mathrm{TWL} = \mathrm{TF}(t).\mathrm{MF}(t)$

$\mathrm{WL} = \mathrm{WLF}(x_1)$

$\mathrm{HH} = \mathrm{TWL} - \mathrm{WL}$

$\mathrm{TTFLUX} = (\mathrm{HH} / \mathrm{L}).\mathrm{SCF}(\mathrm{WL}).\mathrm{VF}(\mathrm{TWL},\mathrm{WL}).\mathrm{GCF}(x_4)$

$h = \mathrm{TWL} - x_7$ if $\mathrm{TWL} > \mathrm{WL}$ and $\mathrm{TWL} > x_7$

$\quad = \mathrm{WL} - x_7$ if $\mathrm{WL} \geq \mathrm{TWL}$ and $\mathrm{WL} > x_7$

$\quad = 0$ otherwise

$\mathrm{MCS} = w.h$

$\mathrm{UC} = (g.h)^{1/2}$

$\mathrm{CONTRF} = \mathrm{UC}.\mathrm{MCS}.\mathrm{CONV} / |\mathrm{TTFLUX}|$ if $\mathrm{UC}.\mathrm{MCS}.\mathrm{CONV} < |\mathrm{TTFLUX}|$

$\quad = 1$ otherwise

$x_{14} = \mathrm{TTFLUX}.\mathrm{CONTRF}$

$x_{21} = \mathrm{RS}(x_3,x_4).x_{14}$ if $x_{14} > 0$

$\quad = 0$ otherwise

$x_{22} = (x_2/x_1).x_{14}.\mathrm{SEM}(x_5,x_6).\mathrm{SEF}(x_3,x_4)$ if $x_{14} < 0$

$\quad = 0$ otherwise

$\mathrm{Ut} = |x_{14}| / (\mathrm{MCS}.\mathrm{CONV})$ if $|x_{14}| > x_{min}$

$\quad = x_{min} / (\mathrm{MCS}.\mathrm{CONV})$ otherwise

$R_{model} = g.\mathrm{DF}(x_2/x_1).(x_{11} / (\mathrm{B}.\mathrm{CONV})) / \mathrm{Ut}^3$

$F_{model} = x_{11} / (\mathrm{B}.\mathrm{CONV}.\mathrm{WL}.[\mathrm{DF}(x_2/x_1).g.\mathrm{WL}]^{1/2})$

$\mathrm{FEX} = |x_{14}| / x_1$

$\mathrm{ASD} = \mathrm{ASF}(x_2,x_5)$

$D = \mathrm{DIAM}.[g.(s-1) / \upsilon^2]^{1/3}$

$i = |\mathrm{TWL} - \mathrm{WL}| / l$

$v_* = (g.h.i)^{1/2}$

$n = 1 - 0.56.\log_{10}D$

$U = |x_{14}| / (\mathrm{MCS}.\mathrm{CONV})$

$S_{mob} = [v_*^n / (g.\mathrm{DIAM}.(s-1))^{1/2}].[U / (5.66.\log_{10}(10.h / \mathrm{DIAM}))]^{1-n}$

$\log_{10}C = 2.86.\log_{10}D - (\log_{10}D)^2 - 3.53$

$A = 0.23/D^{1/2} + 0.14$

$m = 9.66/D + 1.34$

$ST_{cap} = C.[S_{mob}/A - 1]^m$ if $S_{mob} > A$

$\quad = 0$ otherwise

X_{vol} $= ST_{cap}.(DIAM/h).(U/v_*)^n$

WSM $= WSM(WH(t))$

ST_{mod} $= C.[S_{mob}.WSM.BSF/A - 1]^m$ if $S_{mob}.WSM.BSF > A$

$= 0$ otherwise

X_{mod} $= ST_{mod}.(D/h).(U/v_*)^n$

x_{71} $= (X_{mod} - X_{vol}).x_{14} / (w.l.(1 - Ps))$ if $x_{14} > 0$

$= 0$ otherwise

x_{72} $= X_{vol}.x_{14} / (w.l.(1 - Ps))$ if $x_{14} < 0$

$= 0$ otherwise

x_{73} $= BF(t)$

Differential Equations

$$\frac{dx_1}{dt} = \Sigma x_{1j} \qquad j = 1,4$$

$$\frac{dx_2}{dt} = \Sigma x_{2j} \qquad j = 1,2$$

$$\frac{dx_3}{dt} = (R_{model} - x_3) / t_1$$

$$\frac{dx_4}{dt} = (F_{model} - x_4) / t_1$$

$$\frac{dx_5}{dt} = (x_{11}/x_{11n} - x_5) / t_2$$

$$\frac{dx_6}{dt} = (FEX - x_6) / t_3$$

$$\frac{dx_7}{dt} = \Sigma x_{7j} \qquad j = 1,3$$

A2 MANAGEMENT EVALUATION COMPONENT

Equations

$MC^J \quad = x_{7c} - x_7^J$ if $x_7^J < x_{7c}$ and $t \in [a1,b1]$ and $t_{dur1} \geq dur1$
$\quad\quad\quad = 0$ otherwise

$PI_1 \quad = \int MC^J\,dt \;/\; \int MC^R\,dt$

$SI^J \quad = ASD^J - ASD_c$ if $ASD^J < ASD_c$ and $t \in [a2,b2]$ and $t_{dur2} \geq dur2$
$\quad\quad\quad = 0$ otherwise

$PI_2 \quad = \int SI^J\,dt \;/\; \int SI^R\,dt$

$SS^J \quad = x_3^J - x_{3c}$ if $x_3^J > x_{3c}$ and $t \in [a3,b3]$ and $t_{dur3} \geq dur3$
$\quad\quad\quad = 0$ otherwise

$PI_3 \quad = \int SS^J\,dt \;/\; \int SS^R\,dt$

$FS^J \quad = x_6^J - x_{6c}$ if $x_6^J > x_{6c}$and $t \in [a4,b4]$ and $t_{dur4} \geq dur4$
$\quad\quad\quad = 0$ otherwise

$PI_4 \quad = \int FS^J\,dt \;/\; \int FS^R\,dt$

$I\,I^J \quad = WL^J - WL_{ci}$ if $WL^J > WL_{ci}$ and $t \in [a5,b5]$ and $t_{dur5} \geq dur5$
$\quad\quad\quad = 0$ otherwise

$PI_5 \quad = \int I\,I^J\,dt \;/\; \int I\,I^R\,dt$

$EI^J \quad = WL_{ce} - WL^J$ if $WL^J < WL_{ce}$ and $t \in [a6,b6]$ and $t_{dur6} \geq dur6$
$\quad\quad\quad = 0$ otherwise

$PI_6 \quad = \int EI^J\,dt \;/\; \int EI^R\,dt$

A3 SYMBOLS

x_{11}	=	fresh water inflow rate ($m^3.yr^{-1}$)
IF(t)	=	inflow function ($m^3.yr^{-1}$)
x_{12}	=	precipitation/evaporation ($m^3.yr^{-1}$)
RF(t)	=	rainfall function ($m.yr^{-1}$)
EF(t)	=	evaporation function ($m.yr^{-1}$)
SA	=	surface area (m^2)
x_1	=	water volume (m^3)
x_{13}	=	groundwater flow ($m^3.yr^{-1}$)
GF(t)	=	groundwater function ($m^3.yr^{-1}$)
TWL	=	tidal water level (m)
TF(t)	=	tide function (m)
MF(t) .	=	meteorological function (unitless)
WL	=	estuary water level (m)
WLF	=	water level function (m)
HH	=	hydraulic head (m)
TTFLUX	=	target tidal flux ($m^3.yr^{-1}$)
L	=	characteristic estuary length (m)
SCF	=	storage capacity function ($m^3.yr^{-1}$)
VF	=	velocity function (unitless)
GCF	=	gravitational circulation function (unitless)
x_4	=	circulation index (unitless)
h	=	effective depth of flow (m)
x_7	=	sill height (m)
MCS	=	mouth cross-sectional area (m^2)
w	=	width of the estuary mouth (m)
UC	=	critical flow velocity ($m.s^{-1}$)
g	=	gravitational acceleration ($m.s^{-2}$)
CONTRF	=	control factor (unitless)
CONV	=	conversion factor ($s.yr^{-1}$)
x_{14}	=	tidal flow ($m^3.yr^{-1}$)
x_{21}	=	salt import ($kg.yr^{-1}$)
RS	=	reduced salinity ($kg.m^{-3}$)
x_3	=	stratification index (unitless)
x_4	=	circulation index (unitless)
x_{22}	=	salt export ($kg.yr^{-1}$)
x_2	=	salt (kg)
SEM	=	salinity export multiplier (unitless)
x_5	=	fresh water flushing (unitless)

x_6	= tidal flushing (yr^{-1})
SEF	= stratification export function (unitless)
Ut	= tidal velocity ($m^3.s^{-1}$)
x_{min}	= minimum tidal flow ($m^3.yr^{-1}$)
R_{model}	= analogue Estuarine Richardson number (unitless)
DF	= density anomaly function (unitless)
B	= characteristic estuary width (m)
F_{model}	= analogue densimetric Froude number (unitless)
FEX	= fractional tidal exchange rate (yr^{-1})
ASD	= axial salinity difference ($kg.m^{-3}$)
ASF	= axial salinity function ($kg.m^{-3}$)
x_5	= fresh water flushing (unitless)
D	= dimensionless grain diameter (unitless)
DIAM	= grain diameter (m)
s	= mass density of sand relative to fluid (unitless)
υ	= kinematic viscosity of fluid ($m^2.s^{-1}$)
i	= hydraulic gradient at the mouth (unitless)
l	= length of the estuary mouth (m)
v_*	= shear velocity ($m.s^{-1}$)
n	= transitional grain size exponent (unitless)
U	= mean velocity of flow ($m.s^{-1}$)
S_{mob}	= sediment mobility number (unitless)
C	= sediment transport coefficient (unitless)
A	= threshold value (unitless)
m	= exponent in sand transport function (unitless)
ST_{cap}	= dimensionless sediment transport capacity (unitless)
X_{vol}	= volumetric sand transport rate (10^{-6} m^3 sand per unit volume flow rate)
WSM	= wave stirring multiplier (unitless)
WH(t)	= wave height (m)
ST_{mod}	= modified dimensionless sediment transport capacity (unitless)
BSF	= beach slope factor (unitless)
X_{mod}	= modified volumetric sand transport (10^{-6} m^3 sand / unit volume flow rate)
x_{71}	= accretion rate of the sill ($m.yr^{-1}$)
X_{vol}	= volumetric sand transport rate (10^{-6} m^3 sand / unit volume flow rate)
w	= width of the estuary mouth (m)
l	= length of the estuary mouth (m)
Ps	= porosity of the sandy bed (unitless)
x_{72}	= erosion rate of the sill ($m.yr^{-1}$)
t_1	= time delay constant 1 (yr)
x_{11n}	= fresh water inflow rate normal ($m^3.yr^{-1}$)
t_2	= time delay constant 2 (yr)

t_3	=	time delay constant 3 (yr)
x_{73}	=	breaching rate ($m.yr^{-1}$)
BF(t)	=	breaching function ($m.yr^{-1}$)
MC^J	=	mouth condition under policy J (m)
x_{7c}	=	critical sill height (m)
x_7^J	=	sill height under policy J (m)
a1	=	initial critical time 1 (yr)
b1	=	final critical time 1 (yr)
t_{dur1}	=	duration of mouth deepening event (yr)
dur1	=	critical duration 1 (yr)
PI_1	=	management performance index 1 (unitless)
t_i	=	initial time (yr)
t_f	=	final time (yr)
MC^R	=	mouth condition under the reference policy P (m)
SI^J	=	salinity index under policy J ($kg.m^{-3}$)
ASD^J	=	axial salinity difference under policy J ($kg.m^{-3}$)
ASD_c	=	critical axial salinity difference ($kg.m^{-3}$)
a2	=	initial critical time 2 (yr)
b2	=	final critical time 2 (yr)
t_{dur2}	=	duration of salinity event (yr)
dur2	=	critical duration 2 (yr)
PI_2	=	management performance index 2 (unitless)
SI^R	=	salinity index under the reference policy R ($kg.m^{-3}$)
SS^J	=	stratification state under policy J (unitless)
x_3^J	=	stratification index under policy J (unitless)
x_{3c}	=	critical stratification index (unitless)
a3	=	initial critical time 3 (yr)
b3	=	final critical time 3 (yr)
t_{dur3}	=	duration of stratification event (yr)
dur3	=	critical duration 3 (yr)
PI_3	=	management performance index 3 (unitless)
SS^R	=	stratification state under the reference policy R (unitless)
FS^J	=	flushing state under policy J (yr^{-1})
x_6^J	=	tidal flushing under policy J (yr^{-1})
x_{6c}	=	critical tidal flushing rate (yr^{-1})
a4	=	initial critical time 4 (yr)
b4	=	final critical time 4 (yr)
t_{dur4}	=	duration of tidal flushing event (yr)
dur4	=	critical duration 4 (yr)
PI_4	=	management performance index 4 (unitless)
FS^R	=	flushing state under the reference policy R (yr^{-1})

II^J = inundation index under policy J (m)

WL^J = water level under policy J (m)

WL_{ci} = critical water level for inundation (m)

a5 = initial critical time 5 (yr)

b5 = final critical time 5 (yr)

t_{dur5} = duration of inundation event (yr)

dur5 = critical duration 5 (yr)

PI_5 = management performance index 5 (unitless)

II^R = inundation index under the reference policy R (m)

EI^J = exposure index under policy J (m)

WL_{ce} = critical water level for exposure (m)

a6 = initial critical time 6 (yr)

b6 = final critical time 6 (yr)

t_{dur6} = duration of exposure event (yr)

dur6 = critical duration 6 (yr)

PI_6 = performance index 6 (unitless)

EI^R = exposure index under the reference policy R (m)

APPENDIX B

NUMERICAL METHODS

B.1 DIFFERENTIAL EQUATION SOLVER

The model comprises a system of first-order, inhomogenous, non-linear, ordinary differential equations, that is, the equations are of the form:

$$\frac{dx_j}{dt} = F_j(\mathbf{x}, \mathbf{p}, t) \quad j = 1, 2, ..., 7$$

where $\mathbf{x} = (x_1, x_2, ..., x_7)^T$ and $\mathbf{p} = (p_i, p_2, ..., p_m)^T$ are the state variables and the parameters of the system, respectively. Additionally, the values of the state variables at the start of a simulation are known (that is, $x_j(0) = x_{j0}$ $j = 1,2, ...,7$) and the problem is termed an initial value problem.

In selecting a method to numerically integrate these equations, the following aspects were considered:

(i) the *accuracy* requirements, in terms of the desired accuracy of the solution, the precision of the input data and the accuracy of the method itself,

(ii) the ability to handle *discontinuities*,

(iii) the ability to *efficiently accomodate highly eventful and non-eventful periods* ,

(iv) the *stiffness* of the problem, and

(v) the *output frequency* required,

(vi) the *availability/accessibility* and *ease of use* of the numerical routine.

First, the estuarine systems model was designed with two purposes in mind, namely, to adequately represent the physical dynamics of an estuary over time periods of months to years and to enable evaluation of the efficacy of different management policies involving water releases and mouth breachings in maintaining the physical character and functioning of estuaries. Because the physical functioning of an estuary and the efficacy of management actions are described and evaluated in terms of aspects such as response times after shocks to the system (eg. floods), alterations in seasonality or the frequency with which certain physical states are occupied, the resilience of the system and the trends exhibited, to name but a few, the desired accuracy of the solution is at best moderate. Certainly, a highly accurate differential equation solver is not required. This is borne out by the fact that there is considerable uncertainty in the assignment of parameter values. Although, the parameter sensitivity analysis (Section 4.4) indicates that one percent inaccuracies in the majority of parameters do not significantly affect the accuracy of the output, it would be artificial to implement a highly accurate numerical integration routine when the input data is itself fairly inaccurate. Thus a moderately accurate numerical integration routine is required.

Although Pugh (1980) advocates the selection of the simple Euler method for the solution of the system of differential equations comprising a typical systems dynamic model, on the basis that accuracy requirements need not be higher than 0.1 %, this family of methods is not sufficiently accurate for the solution of the equations of the estuarine systems model. Moderate rather than low accuracy is required in this case, particularly in the resolution of the effects of sudden shocks such as floods or mouth breachings. Furthermore, a method with the ability to determine a solution to within a specified accuracy is preferred and the Euler method does not have this facility.

Secondly, one of the desirable aspects of the system dynamics methodology is its ability to incorporate discontinuities (Wolstenholme 1982). As the management action of breaching the mouth may be viewed as a discontinuity, a stringent requirement in the selection of the numerical integration routine is its ability to accomodate such discontinuities. Thus a method which involves the calculation of higher order derivatives is not appropriate. The third requirement that the chosen routine should handle efficiently both highly eventful (eg. floods) and non-eventful periods (eg. when the mouth is closed for a few months) implies that a variable step method should be selected.

Fourth, at the outset little was known about the degree of stiffness of the problem. As the only situation when some of the state variables would alter rapidly while others changed slowly would be the onset of a flood under closed mouth conditions, the solutions were deemed unlikely to contain rapidly decaying transient terms. Thus the problem was thought to have at most a mild degree of stiffness. Therefore, the ability to cope with stiffness is an advantage for a particular solver, but not a prerequisite at the outset. The approach adopted was to select a routine based on the assumption that the problem was not stiff and later to move to a routine with the capability to handle stiffness if this turned out to be necessary.

Integration of the equations from the point $t = 0$ to a specified end point with frequent intermediate output is a requirement. Thus the total length of the simulation is generally 2, 5 or 10 years, but output of moderate accuracy is required at intervals of the order of 3 hours to a day. This criterion eliminates from consideration routines which are highly accurate over the entire interval, but do not supply intermediate output or supply low accuracy intermediate output.

Finally, the model, which is programmed in Fortran, runs on a SUN670 located at the CSIR in Stellenbosch. The programmed IMSL MATH/LIBRARY of Fortran routines is loaded on this UNIX server, whereas the NAG Fortran Library is available on computers (SUN UNIX servers) housed at the CSIR in Pretoria and the University of Natal in Pietermaritzburg. There is no technical reason why a routine from the NAG Fortran Library cannot be used, because the required network links are available and access to, and use of, the geographically remote computing facilities is feasible. However, the convenience of using the locally-supported IMSL routines lends preference to the selection of an IMSL routine rather than a NAG routine.

The latter consideration led to the pragmatic approach of first examining the IMSL

MATH/LIBRARY for an appropriate routine, on the understanding that the search would be extended to the more extensive NAG facilities (NAG 1990) should a method satisfying the above requirements not be available on IMSL. The IMSL routine satisfying the requirements is the Runge-Kutta-Verner 5th order and 6th order method, called DIVPRK (IMSL 1991). This is a variable step method employing only first derivatives, with a facility to provide output within specified moderate accuracy at chosen intervals (Gear 1971). This routine is not recommended for stiff problems, but the approach advocated is first to apply this method and if the problem is found to be stiff to then implement a method called DIVPAG (IMSL 1991).

When DIVPRK was used to solve the equations comprising the estuarine systems model, the problem was found not to be stiff. Thus the routine selected as appropriate was the Runge-Kutta-Verner 5th order and 6th order method with the accuracy tolerance set to 0.01%.

B.2 OPTIMISATION ROUTINE

The form of the optimisation problem to be solved is:

$$\min_{y \in \mathbf{R}^n} - f(y) \qquad \text{subject to } l_i \leq y_i \leq u_i \text{ for } i = 1,2, \ldots n$$

where $f : \mathbf{R}^n \rightarrow \mathbf{R}$ is the management performance index to be optimised, and all the variables are not necessarily bounded. The problem, therefore, is classified as minimisation with simple bounds (IMSL 1991)

The routine used to optimise the management performance index was selected on the following grounds:

(i) *Compatibility with ODE solver and accessibility*

By selecting a routine from the IMSL MATH/LIBRARY, possible problems of compatibility with the differential equation solver employed in the estuarine systems model, were avoided. The locally-supported IMSL Library was also the most accessible source for an optimisation routine.

(ii) *Non-analytical method required*

As it is impossible to solve the problem analytically and the partial derivatives cannot be calculated analytically, the chosen method must not require this. This leaves finite-difference gradient methods, finite-difference Hessian methods and finite-difference Jacobian methods for consideration (IMSL 1991).

(iii) *Rapid convergence to the vicinity of the mimimum*

The location of the minimum is completely unknown. A requirement for the method selected, therefore, is that it converge to the vicinity of the minimum fairly rapidly. The method which best meets this requirement is the finite-difference gradient method (Brent

1973). A method such as the finite-difference Hessian method converges slowly when far from the optimal location, but converges rapidly in the vicinity of the minimum.

(iv) *Moderate accuracy*

The method is not required to determine the minimum very accurately as the accuracy of the model solutions is only moderate and the input parameter values are themselves fairly inaccurate. An accuracy in the region of 5% is considered sufficient. Again, the finite-difference gradient method best satisfies this requirement. If further accuracy were required, subsequent application of the Hessian method in the vicinity of the minimum, where it is most efficient, could be considered.

(v) *Low computational efficiency*

The computational efficiency of the method (in terms of CPU time) is not a strong consideration, because the optimisation routine will not be implemented frequently. Generally, the optimisation routine will be implemented once for each case study. Thus, although the total number of times the model is run within the optimisation may be more using the finite-difference gradient method than the finite-difference Hessian or Jacobian methods, this disadvantage is not considered to outweigh the advantages described above.

Consequently, the finite-difference gradient method of the IMSL MATH/LIBRARY, known as DBCONF (IMSL 1991), was implemented with standard parameter selections apart from a scaling of the start time (y_1) by one tenth in order to assist the routine in converging quickly (i.e. XSCALE(1) = 0.1 and XSCALE(2) = 1.0).

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