WATER QUALITY MODELLING OF ESTUARIES

by

J H SLINGER, S TALJAARD, M ROSSOUW & P HUIZINGA

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EXECUTIVE SUMMARY

The CSIR was commissioned by the Water Research Commission to undertake a two-year research project on the modelling of estuarine water quality with the following specific objectives:

1. Establishing the broad context of water quality problems in South African estuaries through a process of consultative interviews with identified key people in the field (and limited postal interviews where personal consultation is not possible) thereby clarifying the required decision support in terms of water quality modelling;

2. Investigating the utility of the Mike 11 water quality module in addressing the defined problems by applying the model to two selected estuaries;

3. Critically assessing the performance of Mike 11 and establishing whether other internationally available models are better suited, or could contribute, to resolving the defined water quality problems in South African estuaries;

4. Recommending a strategy to develop water quality modelling of estuaries in South Africa to the level required for realistic solution of the defined problems, whether this involves application of Mike 11, or not.

Substantial progress was made in addressing the context of water quality problems in South African estuaries and the information and decision support requirements against which the utility and performance of Mike 11 can be measured. The current understanding of the context of estuarine water quality in South Africa was derived through a literature survey and consultations with international scientists and key South Africans and is captured most succinctly in a companion document entitled "*Decision Support Requirements for Estuarine Water Quality Management*" by Taljaard and Slinger (1997). In this document, the correlations between anthropogenic activities and developments, the associated changes in water quality, existing and

potential water quality problems, the highly dynamic environment of South African estuaries and the information and modelling required to support the management decision process are described. In particular, numerical modelling comprises one of a number of techniques supporting water quality management decision making in South Africa. Within the management framework, it is most appropriately used in compliance testing, determining the fate of different water quality constituents in the environment and establishing the natural or ambient variability of different water quality parameters. In these applications, numerical models play the role of associating variable physical forcing and development scenarios with the abiotic responses of the estuary and of establishing the efficacy of management practices by comparing these responses directly with abiotic compliance measures or indirectly, through biotic indicators or prediction techniques, with biotic compliance measures. Thus they explore the matching between likely and required behaviour under variable environmental conditions, thereby enabling more informed decisions.

Particular information on the utility and performance of Mike 11 in addressing water quality issues in South Africa was obtained by examining its application to the two case studies, the Great Berg and Swartkops estuaries. The systems chosen are both relatively long, narrow estuaries with permanently open mouths and a one-dimensional approach is applicable. In addition, a reasonable amount of data existed on each of these systems; in fact, more data than is commonly available on most South African estuaries. In the implementation of the Mike 11 hydrodynamic, transport-dispersion and water quality modules on the Great Berg Estuary, both the high flow conditions typical of the winter season and the low flow conditions typical of the summer season were considered. The water quality module with sediment-water column interaction was implemented and temperatures, dissolved oxygen concentrations and biological oxygen demand levels were simulated in addition to the simulations of water levels, flows and salinities. Representative ranges of thermal variation and dissolved oxygen concentrations were simulated for the winter 1989, summer 1990, winter 1995 and summer 1996 situations. Generally, DO levels at distances greater than 20 km from the mouth were characteristically low compared with those nearer the mouth. A higher degree of variability was also evident in the more saline lower reaches both in the measured and the simulated values. The simulated range of variation of DO in the lower region of the estuary was less than the measured range apart from under the high flow conditions of August 1995 when the system was freshwater dominated. This more limited variation of the simulations is ascribed primarily to limited data for the seaward boundary, but may also arise from the fact that the effect of salinity on the solubility of oxygen is not incorporated in the Mike 11 model (owing to the de-coupling of the transport-dispersion and water quality modules). More extensive information on variations in the DO and BOD concentrations of the inflowing seawater would be required before the reasons for this feature could be determined absolutely.

The sediment oxygen demand, incorporating the natural or ambient biological oxygen demand within the estuary, was discovered to play a critical role in the calibration of the DO component of the model for the Great Berg Estuary. This proved to be true in the Swartkops Estuary as well, although intensive calibration of this application was not undertaken owing primarily to data limitations. While many data exist for this system, they were collected with purposes other than model calibration in mind and so are not necessarily appropriate to this use. Despite these limitations, a number of characteristic run-off situations and thermal conditions were determined and the thermal and dissolved oxygen dynamics of the estuary were simulated. Time periods of twenty-eight days covering a full spring-neap-spring cycle were selected for simulation. The range of variation of the water quality parameters with position along the estuary and variations in inflow conditions could then be examined. Features of the estuary such as the almost uniform salinities under low flow conditions and the relatively well-oxygenated conditions under strong tidal circulation were evident. Clearly, the effects of altered inflow conditions and waste loading could be investigated using the model as it stands at present, but it would be preferable to undertake the necessary calibration studies to ensure the validity of the simulation results.

Field investigations for the winter condition in the Great Berg indicated that dissolved nutrients are strongly linearly correlated with salinity, while dissolved reactive phosphate-P concentrations and dissolved reactive silicate-Si levels in the Swartkops exhibit similarly strong relationships to salinity. Owing to the decoupling between salinity modelling and the water quality module of Mike 11, it seemed superfluous to attempt to model the nutrient dynamics in relation to temperature and dissolved oxygen processes when they can already be predicted fairly accurately from salinity simulations. Under summer conditions in the Great Berg, such relationships do not exist, particularly with regard to total dissolved inorganic nitrogen-N or its components (dissolved nitrite-N, dissolved nitrate-N and total dissolved ammonia-N). Instead, depletion of total inorganic nitrogen appears to occur in the middle reaches of the estuary. A similar situation exists for the Swartkops Estuary where the total dissolved inorganic nitrogen levels show no clear relationship to salinity (note they are thus also decoupled from dissolved reactive phosphate-P concentrations) and there are no clear patterns of generation or depletion in the different areas of the estuary. As no reliable conceptual model of the biogeochemical processes operative in the estuaries could be derived from the available data, no numerical modelling of the complex behaviour of dissolved nutrient levels was attempted.

Previous work on the Great Berg Estuary indicated that the resident nature of the water in the middle reaches over summer led to depletion of dissolved oxygen in the estuarine water. While this is definitely a strong contributory factor, the modelling undertaken in this study provides insight that the quality of the inflowing water (particularly the river water) plays a major role in determining the DO and nutrient levels of the middle reaches of the estuary towards the end of summer. Evidently, the water entering the system is low in oxygen over the summer period. The reasons for this should be established as this may be a natural feature or may be aggravated by the severe infestation of the upper estuary and river by water hyacinth. Establishing the reasons for such features emphasises the need for monitoring, particularly of the boundary conditions.

An additional test of the utility and performance of the Mike 11 water quality module was undertaken by applying it to the low oxygen event or 'black tide' which occurred in St Helena Bay in March 1994. This event was marked by the intrusion of water entirely depleted in oxygen into the estuary. Massive fish and invertebrate kills were observed. Information on the possible limit of intrusion and the duration of low oxygen conditions in the estuary will contribute to current studies of the recovery of the fish and benthic invertebrates of the Great Berg Estuary following the event. Thus all available data on the event were collated and, despite little scientific data at the onset of the event and slightly better data only well into the event, simulations of the probable conditions were undertaken. Results indicated that DO levels below 2 $mg.l^{-1}$ persisted in the lower estuary for six to seven days, but that such low dissolved oxygen concentrations did not extend 10 km up the estuary. This means that the Great Berg Estuary, which is highly saline in summer, could act as a refuge to any mobile animal able to make its way this distance upstream. This application also demonstrates the efficacy of the Mike 11 model in enabling the association of water quality to biological assessment and prediction, a desirable development in terms of the needs of water quality management in South Africa.

More general information on the acknowledged strengths and weaknesses of the Mike 11 modelling system was obtained from the literature and through consultations with international scientists. In brief, the numerical stability and reliability of the model is deemed its most outstanding feature. Weaknesses include an older file management structure and the inclusion of fewer alternative formulae in the water quality modules than other comparable systems.

In applying Mike 11 to the case studies and analysing its capabilities, it is evident that model performance and utility are intimately linked with data availability. The design of appropriate monitoring methods for estuarine water quality is thus an area requiring attention if adequate decision support for management is to be provided both now and in the future. Additionally, the link between the river and estuary monitoring programmes needs to be established well so that the long term response of an estuary can be liked to specific sources and not just to some isolated change or unknown source in the quality and quantity of the inflowing river water. This is an area which has received considerable attention internationally, particularly in recent years. Much monitoring of the effects of catchment land-use practices on the water quality of downstream rivers and estuaries is taking place. Technical developments are focussed on the synthesis of information for management through the development of decision support systems which integrate monitoring, models of water flow, waste loading and water quality and ecological responses to provide measures of system performance useful in testing management practices and supporting decision making. South Africa could benefit by recognising the importance of monitoring and by adopting some of the recent technological developments in integrated decision support for holistic river basin management.

For all water quality constituents, however, estuarine specific water quality guidelines are necessary if model applications and compliance testing are to become routine decision support tools in water quality and river basin management. Such guidelines will assist in ensuring that the focus remains on linking the predicted or measured water quality changes to the observed effects and not on the study of the biological response in isolation of the causative effects. Thus skills in associating water quality changes and biotic responses are essential. Mike 11 can play a role in linking human activities and developments to water quality changes and hence to the prediction of biological responses, thereby assisting in ensuring the continued existence of healthy estuarine environments.

Limitations in the current modelling activities relate to the prediction of bacterial contamination and the simulation of dissolved nutrient distributions under all flow conditions, that is, both when bio-geochemical regeneration or depletion processes become significant as well as when the distributions are circulation-dependent. In the case of the prediction of bacterial contamination, further mathematical modelling skills need to be developed or acquired particularly as this is an acknowledged problem in the KwaZulu-Natal area and on other South African estuaries e.g. the Swartkops system. In the case of the simulation of dissolved nutrient distribution, investigative field research is necessary to establish and quantify a conceptual model of nutrient dynamics before full scale predictive modelling can be undertaken.

Accordingly, to ensure that present and future estuarine water quality management can act from **the perspective of preventing the occurrence of problems** (the approach strongly advocated by key South African scientists and managers) rather than allowing them to develop and then attempting to solve them, the following strategy is recommended:

1. The formulation of a pragmatic decision support framework for estuarine management in which information on aspects such as water quality monitoring and prediction, water quantity management and development options and the effects of catchment land-use practices on downstream estuaries is integrated, in such a way as to enable the implications of different management measures to be explored realistically;

2. The establishment of South African water quality guidelines for estuaries;

3. The development of a comprehensive monitoring strategy for South African estuaries and the establishment of appropriate procedures for processing and analysing monitoring data;

4. The initiation of investigative studies to address deficiencies in the understanding of nutrient dynamics in South African estuaries;

5. The acquisition and/or development of a capability to model bacterial contamination in South African estuaries;

6. Encouragement of the inclusion of water quality aspects in existing or developing biological response models so that the prediction of the response of biota to water quantity and quality changes can be addressed more comprehensively in future.

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GLOSSARY OF TERMS

Biochemical oxygen demand (BOD)	-	The amount of dissolved oxygen consumed by micro-biological action when a sample is incubated (usually for 5 days at 20 °C in the presence of a nitrification inhibitor).
Benthic	-	Bottom-dwelling (animals).
Biological monitoring	-	Assessment of the biological status of populations and biotic communities or indicator organisms that are at risk, in order to protect them and to provide an early warning system of possible hazards to human and environmental health.
Chemical oxygen demand (COD)	-	The amount of oxygen consumed by a specified oxidising agent in the chemical oxidation of the matter present in a sample.
Compliance	-	Adherence to pre-set objectives, guidelines or standards.
Decision support system	-	A computer-based system that assists people in making decisions or drawing conclusions based on (environmental) data and includes a database management system, models and user-interface components.
DELFT3D	-	A suite of three dimensional hydrodynamic, transport-dispersion and water quality numerical models, and a two-dimensional morphological model, designed and developed by Delft Hydraulics.
DELWAQ	-	A water quality model developed by Delft Hydraulics which comprises an extensive library of water quality process routines applicable to marine or freshwater environments and which links to one, two or three dimensional numerical models.
Dissolved reactive phosphate	-	Also referred to as soluble reactive phosphate
DO	-	Dissolved oxygen
Mike 11	-	A one dimensional hydrodynamic, transport-dispersion and water quality numerical modelling system designed and developed by Danish Hydraulics Institute.

Model	-	A formal representation of a component of the world or a mathematical function with parameters that can be adjusted so that the function closely describes a set of empirical data. A mathematical or mechanistic model is usually based on biological, physical or chemical mechanisms or processes and has model parameters that have real world interpretation.
Monitoring	-	Long term, standardised measurement, observation, evaluation and reporting of the environment to define status and trends.
MSL	-	Mean sea level
Numerical modelling	-	Computer-based solution of the equations of a mathematical model using numerical algorithms.
SOBEK	-	A one dimensional hydrodynamic, transport-dispersion and water quality numerical modelling system designed and developed by Delft Hydraulics.
Sediment oxygen demand (SOD)	-	The oxygen consumption or production of the sediment as well as the natural biological oxygen demand of the estuarine water. This parameter therefore represents the net oxygen demand within the estuary apart from the requirements for biotic respiration.
Water quality guideline	-	a numerical concentration or narrative statement recommended to support and maintain a designated water use.

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1. INTRODUCTION

In 1993 the development of water quality modelling expertise was identified as a priority research requirement by the Co-ordinated Research Programme on Decision Support for the Conservation and Management of Estuaries, undertaken by the Consortium for Estuarine Research and Management (CERM) of which the CSIR is a member. Two estuarine water quality models, namely, the Mike 11 water quality module, and the water quality systems model developed by Dr A Ramm were feasible in South Africa at the time. The Mike 11 modelling system had been used by the CSIR in the simulation of hydrodynamic and transport -dispersion aspects of estuarine behaviour in systems such as the St Lucia, Knysna, Great Berg, Great Brak, and Kromme Estuaries, but the water quality module of the Mike 11 system had not been calibrated on a South African estuary. In view of the international acceptability of the Mike 11 system, Dr A Ramm recommended that Mike 11 be applied rather than the water quality systems model, which is most applicable to the small lagoonal systems typical of the KwaZulu-Natal coast. The CSIR was subsequently commissioned by the Water Research Commission to develop expertise in the modelling of estuarine water quality, focusing primarily on Mike 11 applications.

The objectives of the research programme are:

1. To establish the broad context of water quality problems in South African estuaries through a process of consultative interviews with identified key people in the field (and limited postal interviews where personal consultation is not possible) and so to clarify the required decision support in terms of water quality modelling.

2. To investigate the utility of the Mike 11 water quality module in addressing the defined problems by applying the model to two selected estuaries.

3. To critically assess the performance of Mike 11 and establish whether other internationally available models are better suited, or could contribute, to resolving the defined water quality problems in South African estuaries.

4. To recommend a strategy to develop water quality modelling of estuaries in South Africa to the level required for realistic solution of the defined problems, whether this involves application of Mike 11, or not.

Initially, the first objective did not include any description of the means of establishing the context of water quality problems in South Africa and the required decision support for estuarine water quality modelling. However, early in 1995 this objective was modified to reflect the consultative approach to be adopted. Similarly for objective 2, it was originally intended that the utility of the Mike 11 water quality module be tested on three estuarine case studies. Early in 1996, however, after a successful and data intensive implementation of the model on the first case study, the scope of this objective was reduced to two estuaries to enable the study team to devote the requisite attention to each case study and conduct the necessary research as comprehensively as possible.

The successes achieved in addressing the four objectives of the two-year research programme are described in this report. Section 2 deals with the methods and materials used in the study, while the results of various components of the research programme are described in Section 3. These include:

- the outcome of the literature survey and consultations with a number of international experts,
- the production of a document describing the context of estuarine water quality management in South Africa and decision support requirements,
- the application of the Mike 11 model to the first case study, the Great Berg Estuary, and
- the application of the Mike 11 model to the second and last case study, the Swartkops Estuary.

The implications of the findings are then discussed in Section 4, both in terms of their relevance for estuarine water quality modelling in South Africa and their significance for the management of the water quality of the case study estuaries. Finally, the conclusions and recommendations arising from the study are presented in Section 5.

Certain words and terminology common to water quality and modelling studies, but which might cause confusion or uncertainty for the reader, are clarified in the glossary.

2. MATERIALS AND METHODS

The activities undertaken during the two year research programme included:

- a literature survey to establish the state of estuarine water quality modelling internationally; also to identify the common elements and the differences between the water quality problems encountered in South Africa and elsewhere and so to address the applicability of water quality modelling techniques to the South Africa situation,
- consultation with a number of foreign scientists active in the fields of water quality, ecology, environmental engineering and decision support to discover their evaluation of the most effective approach to water quality modelling of estuaries in South Africa,
- the formulation of a document describing the decision support requirements for estuarine water quality management in South Africa and incorporating the views of key people in the fields of estuarine water quality modelling and management in South Africa,
- winter and summer field exercises to the Great Berg Estuary to collect data for the calibration of the Mike 11 water quality module under high and low flow conditions,
- the calibration of Mike 11 for both winter and summer conditions on the first case study, the Great Berg Estuary,
- simulation of the intrusion of anoxic water into the Great Berg Estuary during March 1994, an event devastating to certain of the coastal and estuarine fauna,
- the implementation of the hydrodynamic, transport-dispersion and water quality modules of Mike 11 on the second case study, the Swartkops Estuary, over one month periods and under various freshwater flow rates.

The methods adopted in undertaking each of these activities are described below.

2.1 Literature Survey & Consultations with International Scientists

Two composite databases were searched, namely:

- WATERLIT (which includes Waterlit SA, Delft Hydro and Aquaref), and
- Marine, Oceanographic and Freshwater Resources (which includes Aquatic Sciences and Fisheries Abstracts, the NOAA database and eleven other smaller databases).

The WATERLIT database covers articles from 1970 to the present, while the earliest records searched in the Marine, Oceanographic and Freshwater Resources database stretched back as far as 1960. The fields of freshwater, brackish and marine water quality management, data collection methods and modelling techniques are well covered by these databases.

In addition, an Internet search was conducted using the Alta Vista search engine (located at www.altavista.com). The keywords used in all the searches included estuary, model, modelling, numerical, mathematical, salinity, TDS, conductivity, oxygen, BOD, nutrients, bacteria, faecal coliform, trace metal, turbid and silt as well as variations of these words.

A total of 377 records from the composite databases was tagged and printed out. Each of these records was then examined for relevance to the study and copies of a number of articles and books were requested. The Internet search yielded a total of twenty-two highly relevant articles. A sub-total of 135 articles and five books comprised the final basis on which the literature study was conducted. These articles and the extracts from the books were then categorised according to:

A. their management orientation,
B. the modelling technique used or the variables modelled, and
C. the water quality data used in the study.

The key findings from the most applicable articles in each of these categories were then collated and summarised.

Additionally, the opportunities presented by Ms J H Slinger attending the Conference on Strategies and Methods in Coastal and Marine Management at Trinity College, Dublin, from 11-16 September 1995 and subsequently visiting Delft Hydraulics in The Netherlands in September 1995 and again from November 1996 to March 1997, were utilized to consult a number of international scientists with expertise in the management and modelling of estuarine water quality. These discussions were instrumental in defining the need for quality modelling in South Africa and in directing the actions taken in implementing the Mike 11 modelling system on the two case studies.

2.2 Water Quality Discussion Document

In order to develop a common understanding of water quality issues in South Africa and the decision support needs for management, it was decided that a process of consultative interviews should be undertaken in association with the circulation of a discussion document. In the original planning of the research programme, the interviews and finalisation of the document were scheduled for completion within the first year of the research programme. However, on commencing the drafting of the document, it became evident that a deeper understanding of water quality issues and possible decision support requirements was required before a well balanced document could be formulated. There was a strong temptation to be idealistic in setting the context of the water quality problems of South African estuaries and the associated decision support needs, whereas a robust and pragmatic approach was required. The finalisation of the initial discussion document and the interview process were, therefore, postponed until the insights acquired during 1995 and early 1996 could be incorporated in the document. These derived from three sources, namely, the literature survey, Ms J H Slinger's attendance of the international estuaries conference in September 1995 and subsequent visit to Delft Hydraulics (developers of 1-, 2- & 3-D water quality models), and the experience gained by applying the Mike 11 water quality module to the first estuarine case study. Thus, the discussion document was circulated in late March 1996 (Taljaard & Slinger 1997) to key people in the estuarine water quality field, namely, selected members of the Department of Water Affairs and Forestry (DWA&F), the Department of Environment Affairs & Tourism (DEA&T), Western Cape Nature Conservation, Eastern Cape Nature Conservation, the Natal Parks Board, the CSIR, CCWR and the estuarine scientific community (CERM). All recipients of the document were requested to comment. Additionally, interviews were scheduled with fifteen people in order to garner further insights into the decision support needs for estuarine water quality management (Table 1). During the interviews, comments and suggested modifications to the document were specifically solicited. All of the comments received were taken into consideration in the final revision of the document.

Table 1. **Schedule of interviews with key estuarine scientists, water quality specialists and managers**

INSTITUTION	KEY PEOPLE	INTERVIEW
DWA&F Institute for Water Quality Studies	Mr G Quibell Ms A Belcher	10:15 on 16 April 1996
DWA&F Environmental Services	Mr K Legge Ms B Weston Mr C Bruwer	09:00 on 17 April 1996
DWA&F Water Quality Management	Mr S van der Westhuizen Dr D van Driel	13:30 on 17 April 1996
Walmsley Consultants	Dr H MacKay	14:00 on 18 April 1996
CSIR (Environmentek, Pretoria & Durban)	Dr P Ashton Mr N Rossouw Mr T Harrison Dr A Connell Mr C Archibald	08:30 on 18 April 1996 & 08:30 on 2 May 1996
University of Port Elizabeth	Prof G Bate Dr J Adams	09:00 on 30 April 1996

2.3 Implementation of Mike 11 on First Case Study, Great Berg Estuary

The Great Berg Estuary (Figure 1) was selected as the first case study site because it exhibits common water quality concerns for South African estuaries such as, upstream impoundment (resulting in alterations in the quantity and quality of freshwater supplied to the estuary), increased agricultural activity (affecting run-off, sediment and nutrient supplies) and seasonality in the abiotic environment and the effects of this on the biota. Other water quality issues such as fish factory discharges in the lower reaches, a marina near the mouth and the effect of episodic occurrences of low oxygen conditions along the Cape West Coast provide added interest. Additionally, the long (> 60 km), permanently open Great Berg Estuary is suitable for one-dimensional modelling as the vertical differences in salinity generally are small compared with the longitudinal extent of the system (Slinger & Taljaard 1994). The location of the estuary some 150 km north of Cape Town also meant that it was within easy reach of the study team should additional fieldwork be required.

Following the selection of the study site, field excursions to the Great Berg Estuary were undertaken from 14 to 18 August 1995 and on 27 August 1995, with the purpose of collecting data relevant to the calibration of the Mike 11 water quality module (DHI 1992) under the high flow conditions typical of the winter season. The details of the methods and activities of the winter field exercise are described in Slinger and Taljaard (1996). Using the model parameter ranges or suggested values derived from the field data, initial calibration of the Mike 11 water quality module commenced. This calibration was undertaken for the time period 17 to 23 September 1989, because field data were available for the first four days of this period (Slinger & Taljaard 1994). Satisfactory calibration was achieved by fine tuning the parameters within the range of values suggested by the field excursion results. However, in view of the extent of seasonal variation in salinities and temperatures known to occur in the Great Berg Estuary (Slinger & Taljaard 1994), the extension of this parameterisation to the summer low flow state was considered unsatisfactory. Accordingly, a second series of field excursions to the Great Berg Estuary was undertaken on 5, 24 February 1996 and 4 March 1996. The details of the methods and activities of the summer field exercise are described in Slinger *et al.* (1996). These data were used in conjunction with data collected from 29 January to 3 February 1990 (Slinger & Taljaard 1994) to calibrate the Mike 11 water quality module for the 1990 summer condition. Satisfactory calibration was achieved by some adjustment of parameters relative to the winter situation. The hydrodynamics and salinity of the estuary were simulated from winter 1989 through to summer 1990, whereas the water quality module was implemented for two periods of five day duration within this time.

Thereafter, the winter and summer calibrations of the water quality module were tested by applying Mike 11 to the measured winter 1995 and summer 1996 situations. The results of the winter 1989 and summer 1990 calibrations and the winter 1995 and summer 1996 simulations are described in Section 4.

Additionally, background data on the hydrodynamics and water quality of the Great Berg Estuary were collated. Data were obtained from the Zoology Department, University of Cape Town (Mr S Lamberth pers.comm.), the Sea Fisheries Research Institute (Dr G Bailey pers.comm.), the Botany Department, University of Port Elizabeth (Dr J Adams pers.comm.) and the CSIR, Durban (Mr T Harrison pers.comm.). Together with the data held by CSIR, Stellenbosch, this data set represents a comprehensive collection of available scientific data on the water quality of the Great Berg Estuary.

Supplementary data were obtained from newspaper and popular articles on the 'black tide' of March 1994, providing the background against which the effects of this episodic low-oxygen event on the Great Berg Estuary could be simulated. Intriguing results from these simulations are presented in Section 4 and discussed further in Section 5.

2.4 Implementation of Mike 11 on Second Case Study, Swartkops Estuary

The Swartkops Estuary, located immediately to the north of Port Elizabeth, is a large permanently open estuary approximately 16,4 km in length. Located in a highly urbanised area and draining a catchment subject to substantial agricultural and industrial activity, the Swartkops Estuary provides an ideal case study for testing the applicability of Mike 11 further. The type of water quality problems which could occur (and the issues presently encountered in the system) include:

- eutrophication (nutrient enrichment, primarily from sewage effluent, urban storm water and industrial discharges and agricultural drainage, nuisance macro algal growth);
- suffocation of biota (limited oxygen availability following organic enrichment);
- other physiological stresses to biota (owing to altered salinity and temperature regimes, altered turbidity and siltation and/or the presence of toxic substances);
- human health risks (bacterial contamination, primarily from sewage effluent and untreated urban storm water drainage), and;
- unpleasant aesthetics (colour change, foam, litter and debris, oily films, odours).

A major advantage in the selection of this estuary as the second case study was considered to be the availability of substantial information on the water quality of the river and estuary and the monitoring programme which is conducted on an ongoing basis (MacKay 1993, Environmental Services 1993, Scharler pers. comm., Adams pers. comm., Belcher pers. comm.). A further advantage lay in the opportunity to investigate the utility of the Mike 11 modelling system in addressing water quality issues different from those of the Great Berg Estuary.

Ms U Scharler, Prof D Baird, Dr J Adams, Prof G Bate and Prof T Wooldridge of the University of Port Elizabeth, Ms A Belcher of the Water Quality Institute of the Department of Water Affairs and Forestry and Dr H MacKay of Walmsley Consultants were contacted in connection with the collation of relevant water quality data for the Swartkops Estuary. Additionally, two field visits were undertaken by project participants. The first involved an aerial inspection of the study site on 30 April 1996 in company with Prof G Bate and Ms U Scharler. This reconnaissance visit proved invaluable in familiarising both Ms J H Slinger and Ms Taljaard with the extensive development of the lower catchment and with the location of various inflows to the estuary e.g. Motherwell Canal, Chatty River. The second field visit to the system by the project participants was undertaken on 16 August 1996. On this occasion, salinity and temperature were measured and water samples collected at several positions and depths along the channel of the estuary and analysed according to the methods described in Slinger *et al.* (1996). Relevant field results are presented in Section 4.2, while full details are provided in APPENDIX A.

The available data were analysed in terms of their utility in calibrating and implementing the Mike 11 modelling system. Unfortunately, much of the available information was collected for purposes other than model applications and consequently lacked the required spatial, temporal or parameter coverage. For instance, measurement stations were often located only on the sides of the estuary and not in mid-channel; the river inflows at the head of the estuary were measured infrequently, if at all; the states of the river and sea boundaries (e.g. salinities, temperatures, dissolved oxygen (DO) levels) were often not measured; only salinity and temperature were measured, no corresponding water level variations, dissolved oxygen concentrations or dissolved nutrient levels at the time are known; detailed bacterial contamination studies were conducted in a sub-region of the system, but the ambient and boundary conditions at the time were not well established. Consequently, rather than calibrating the Mike 11 model for specific measurement periods and simulating measured events, the approach adopted was that of applying a number of representative flow rates over periods of a month and evaluating the resulting variations in water levels, flows, salinities, temperatures and DO levels. These simulation results are presented in Section 4.2.

The available nutrient data and data on bacterial contamination were collated and analysed in an attempt to identify representative river-borne influx rates, and to quantify lateral inputs along the estuary and exchanges with the marine environment. Results from investigations such as those of Emmerson (1985), Winter and Baird (1991), Baird and Winter (1992), MacKay (1993), MacKay (pers. comm.) and Von der Molen (pers. comm.), the ongoing monitoring by the University of Port Elizabeth (Ms U Scharler pers. comm.) and the one-day monitoring exercise conducted by CSIR are presented in Section 4.2 and their utility for the modelling of dissolved nutrient concentrations and bacterial contamination is assessed.

3. RESULTS: THE STUDY CONTEXT

An understanding of the context of water quality management in South Africa and modelling requirements in this regard was derived from a variety of sources. These included the literature survey, discussions with international scientists, and the preparation and finalisation of the document on the decision support requirements for estuarine water quality modelling, which involved consultation with key people in the fields of estuarine management and water quality in South Africa. The results of these activities are reported subsequently. Considerable overlap is apparent in the findings from the different sources. This overlap is deemed encouraging and has assisted in providing a clear picture of the requirements for water quality modelling of South Africa estuaries.

3.1 Literature Survey

First, the selected articles and extracts from books were categorised broadly according to:

A. their management orientation;
B. the modelling technique used or the variables modelled; and
C. the water quality data used in the study.

A number of sub-categories were also introduced to facilitate the comparison of techniques and approaches: Category A. Management Orientation was sub-divided into a further 3 sub-categories (decision support systems, water quality indices and general); Category B. Modelling Techniques and Variables into 14 sub-categories, and Category C. Water Quality Data into 7 sub-categories (salinities and hydrodynamics, ground water, biological response, monitoring design, dissolved oxygen and dissolved nutrients, trace metals, and catchment influences). Clearly, articles could fall into more than one category or sub-category. An analysis of the distribution of the articles amongst the categories and sub-categories yielded interesting results.

Only ten articles fell into category A and of these the majority focused on the use of water quality indices in the management of water quality. Some 40 articles were allocated to category C. The sub-categories with the most articles were those dealing with salinity and hydrodynamics or dissolved oxygen and dissolved nutrient data. However, a limited number of interesting articles on the response of the biota to water quality were found. The vast majority of the articles, about 150, were allocated to category B. The sub-category with the most articles dealt with the modelling of dissolved oxygen and dissolved nutrient concentrations. Other sub-categories receiving moderate attention were those related to the effects of catchment activities, biological response prediction, salinity/stratification and circulation, and one and two dimensional modelling approaches. It was extremely interesting to note that most of the literature obtained via the composite database search dealt with the application of older approaches. The most recent advances in water quality modelling, such as the three-dimensional modelling system (DELFT3D) from Delft Hydraulics (Delft Hydraulics 1996a), were located via the Internet search and discussions with international scientists. Similarly, the exciting developments in the areas of decision support systems for catchment water quality management and biological response prediction also received only limited coverage in the composite databases, but were extensively alluded to in discussions with foreign scientists.

The geographical extent of the articles selected covered twenty countries ranging from developed countries such as the United States of America, the United Kingdom, Denmark, Belgium, The Netherlands and Australia to developing countries such as South Africa, Namibia, Venezuela, Chile, Mexico, Malaysia and Indonesia. Publications from countries with similar estuarine systems to South Africa were limited. However relevant publications were found on systems in Australia, USA (California and the Gulf Coast), Mexico, Malaysia and Indonesia.

In describing the conclusions drawn from the literature survey, comprehensive referencing of all relevant publications will not be attempted as the large number of articles reviewed means that this would be extremely cumbersome. Instead, a full listing of the articles consulted in this review is provided in the bibliography, while only the most influential articles are referenced in the text hereafter.

Most of the literature covered the development and/or implementation of standard hydrodynamic and water quality models to various real and theoretical water systems (Category B). The models ranged from box models and one-dimensional vertically-averaged models to fully three dimensional systems (e.g. DELFT3D from Delft Hydraulics (1996a)), while the water bodies on which they were applied included lakes, rivers, bays, coastal seas and estuaries. Mike 11 is one of a suite of standard one-dimensional hydrodynamic, morphological and water quality modelling systems available on the international market (DHI 1992). Other similar systems include those developed by Delft Hydraulics (the SOBEK modelling system, Mr J van Gils pers. comm.) and the Wallingford Hydraulics Institute (Ms P Brown pers. comm.), while various software packages are available from Applied Science Associates (Mr R Thomas pers. comm.) but are not linked to a comprehensive system. There are also many modelling packages with part of the Mike 11 functionality e.g. only the hydrodynamics component, aspects of the water quality model, more comprehensive water quality models without the hydrodynamic and morphological components. Thus in gathering information from the literature to assess both the capacity and suitability of Mike 11 to water quality modelling of South African estuaries, two aspects require attention. These are:

- the applicability of a Mike 11-type modelling system to the kind of water quality problems and management requirements that occur in South African estuaries;
- whether Mike 11 displays more or less functionality than similar internationally available modelling systems.

In addressing these aspects, we will tackle the applicability of 1-D models to South African estuaries first, then consider the type of water quality problems and required management information needs and whether other modelling approaches need consideration. Thereafter, an assessment of Mike 11 functionality will be provided. This cannot be comprehensive, as we have not had the opportunity to apply other comparable modelling systems, but have merely seen demonstrations, reviewed literature and held discussions with international experts. However, a reliable first indication can be obtained.

One-dimensional, vertically-averaged models are commonly applied in estuaries around the world, but are best suited to long, relatively narrow systems which are morphologically stable and exhibit vertically mixed water columns and fairly long residence times (Leendertse 1967, Fischer 1979, Smith 1986). If strong stratification is a feature of the estuary, it may be appropriate to apply a box model (Fischer 1979, Hoch *et al.* 1993) or a two dimensional vertical (2dv) model (Tee & Lim 1987). On the other hand, if the system is broad rather than narrow (for instance, gravitational circulation effects such as Coriolis force causing differences in lateral circulation), it may be more appropriate to apply a two-dimensional horizontal (2dh) model (Postma 1984). If the system is both broad and the circulation is strongly influenced by density currents, a full three dimensional treatment may be necessary (Falconer *et al.* 1989, Delft Hydraulics 1996a). Thus the shape and ambient variability of the estuary play a role in determining the appropriate modelling system for application. Additionally, the degree of morphological variability acts as a limiting factor in model applicability (Slinger 1996a). Many hydrodynamic modelling systems assume a stable bottom or channel topography and do not accommodate changes in morphology. Those that simulate morphological change may not be able to accommodate rapid changes, such as the closure of an estuary mouth within one tidal cycle, as this can cause numerical instabilities (Mr P Huizinga pers. comm.). The highly dynamic nature of some South African estuaries (Heydorn & Tinley 1980, Whitfield 1995), therefore, implies limited applicability of models which do not accommodate morphological change.

Higher dimensional models are commonly used where spatial problems require resolution (Falconer *et al.* 1989), for instance, where the movement of a sand bar in a navigable river is of concern or where the three dimensional behaviour of a pollutant plume is of relevance in a marine bay or wider estuary (Ms N Villars pers. comm.). While such problems do require resolution in harbour areas e.g. Saldanha Bay, generally South African estuaries are relatively long and narrow and consequently are more amenable to 1-D treatments (Huizinga 1985, Slinger 1996b). The computational requirements of higher dimensional modelling also limit the time horizons which can be used with these models. For instance, in a recent application of the morphological module of the three-dimensional DELFT3D system to reclamation alternatives for the expansion of Rotterdam Harbour, the time horizon of five years was determined on the basis of computational constraints (Mr D Walstra pers. comm.).

Additionally, the complexity of the input and calibration data increases with an increase in the number of dimensions considered (Rasmussen 1989, Delft Hydraulics 1996a). Thus, in South Africa where data are often limited and rarely extend over long time periods (and thus do not cover the full range of variability exhibited by the estuarine systems), higher dimensional models do not provide a general, pragmatic solution to predictive requirements for hydrodynamics and morphology, let alone water quality (Slinger 1996a, 1996b). The applicability of higher dimensional models is confined to marine bays with fairly stable entrance configurations at this stage e.g. Durban Bay, Richards Bay, Knysna Estuary and Saldanha Bay. In contrast, one-dimensional modelling (with branching) provides a more generally applicable approach (Huizinga 1985, CSIR 1993), particularly when morphological, hydrodynamic and water quality aspects are linked (Falconer *et al.* 1989, Knoester *et al.* 1991). For a limited number of strongly stratified South African estuaries, the application of a two-dimensional, laterally-averaged model may be justified, particularly where there is concern regarding the flushing of hypoxic bottom waters (Largier & Slinger 1991, Slinger *et al.* 1994). This application is receiving attention worldwide (MacDonald & Weisman 1977, Van Es & Ruardij 1982, Malmgren-Hansen *et al.* 1984, Butt 1992, Nelson *et al.* 1994, Cerco 1995)

However, the above considerations do not mean that standard one-dimensional systems can address all the abiotic and water quality problems which occur in South African estuaries. We have seen so far that these types of models are generally more readily applicable than higher dimensional approaches, but questions remain as to whether they are also better than other types of modelling approaches. As described in the Predictive Capability sub-project (Slinger 1996b) and discussed in certain of the papers of Category A (Chartock 1982, Humphries et al. 1984, Wood 1985, Adams & Bate 1994, Cerco 1995, Lynch & Wiebe 1996), it is often necessary to supplement or move beyond standard modelling approaches to address the natural variability of the physical system effectively and resolve the relevant management issues and water quality problems. Clearly, non-standard techniques such as systems modelling, expert systems, decision mapping, supplementary statistical methods, spatial overlaying and the generation and use of water quality indices are often very appropriate tools in translating process-based modelling into information useful for management. In the South African context, non-standard approaches are necessary primarily to compensate for the dynamic nature of the mouth, and for stratification effects which are ignored by vertically-averaged one dimensional models. However, these aspects have been tackled swith some success in the development and implementation of the Estuarine Systems Model and its use as a complementary tool to a standard 1-D approach (Slinger 1995, 1996a & b). Thus a Mike 11 type approach is applicable to the majority of South African estuaries. It is principally useful for permanently open systems over time horizons of up to a year, but may also be applied (optimally, in conjunction with the Estuarine Systems Model) to temporarily closed systems over time periods of a few months (Slinger 1996a & b).

Very apparent in the literature reviewed is the paucity of water quality data available for analysis in South Africa in comparison with the data-rich environments elsewhere in the world (Butt 1992, San Francisco Estuary Project 1992, Maheshwari et al. 1994, EHP 1996, Jensen 1996, Napa Valley Economic Development Corporation 1996, TBNEP 1996). While lack of data quite obviously limits the type of model application that can be undertaken, it also limits the depth of understanding of South African water quality issues. In South Africa, the abiotic and chemical environment of an estuary is highly variable and few data exist with which to establish the degree of variation from ambient conditions (Taljaard & Slinger 1997). In contrast, the severity of the problems internationally and the measurable nature of the differences from the ambient natural situations mean that descriptions of the effects of waste inputs on estuaries and even more notably the effects on water column productivity are a predominant theme in the many papers dealing with dissolved oxygen and dissolved nutrient modelling and measurement (Parra-Pardi 1980, Van Es & Ruardij 1982, Humphries *et al.* 1984, Malmgren-Hansen *et al.* 1984, Lindeboom *et al.* 1989, DeGroodt & De Jonge 1990, Butt 1992, Cerco 1995, Lewis 1992, Bach *et al.* 1992). However, there are indications that enhanced water column productivity is not a typical response of South African estuaries, but rather enhanced benthic productivity and macrophyte growth (Adams & Bate 1994, Whitfield & Wooldridge 1994). This implies that sophisticated techniques to predict water column productivity may be inappropriate for South African conditions. Instead, biological and appropriate habitat modelling approaches may be necessary to establish the true impact of water quality changes in estuaries. The fact that many non-water column effects on biota occurred and were not predicted by standard approaches was noted by some authors (Lindeboom *et al.* 1989, Lynch & Wiebe 1996, James *et al.* 1996). This was indicated as a future direction for development in modelling the effects of water quality changes and mention was made of the role of linking different types of models to predict biological responses and evaluate changes against criteria (Knoester *et al.* 1991, Peviani *et al.* 1996). International examples of specific attempts in this direction, relevant to estuaries, include the modelling of macrophyte responses to dredging in the Baltic region (Dr M Lingby pers. comm.) and the investigation of the effects of alterations in habitat suitability on fish biomass owing to water quality changes (dissolved oxygen levels) in the Sea of Azov (De Vries 1996). The latter study is also a notable example of a situation in which physical and biological process models were incorporated in a decision support system for estuarine water quality management. Decision support systems for management of freshwater on the basis of both water quantity and water quality concerns are routinely implemented by Delft Hydraulics (1994a) for river basins.

Besides the need to indicate the biological effects of water quality changes, there is a need to indicate which management measures would ameliorate a situation and which exacerbate it. Similarly there is a need to be able to establish the effects of remote and diffuse waste sources so that the severity of agricultural pollution and bacterial contamination from catchments can be adequately controlled. Internationally, situations exist in which appropriate techniques and management support were not timeously available and so severe problems have arisen. These include the eutrophication of many rivers and estuaries in Australia, Europe and the United States (Humphries *et al.* 1984, Maheshwari *et al.* 1995, DeGroodt & De Jonge 1990, Butt 1992, Van Gils 1993, Lewis 1992, Bach *et al.*1992), the toxic compounds in the waters and sediments of many systems worldwide (Bourg 1987, Van Gils 1993, Groot *et al.* 1994, De Vries 1996) and the bacterial contamination of rivers and seas of many developing and developed countries (Delft Hydraulics 1997). In comparison, the degree and prevalence of water quality problems are limited in South African estuaries (Whitfield 1995, DWA&F 1995), although catchment-derived bacterial contamination is fairly widespread in the estuaries of KwaZulu-Natal (Gardner & Archibald 1990). Adequate monitoring data are required to maintain and even improve the South African situation as is management-related catchment and estuarine water quality modelling.

Finally, in investigating the functionality of Mike 11, only other one-dimensional systems covering hydrodynamics, morphology and water quality were considered i.e. only relatively comparable systems were assessed. Modelling systems without all these aspects obviously exhibited less functionality and were therefore excluded at the outset. Thus, the SOBEK system of Delft Hydraulics was considered as were the systems from Wallingford Hydraulics Institute.

Mike 11 (DHI 1992) was found to be one of the most advanced modelling systems of its type, in that:

- where uncertainty exists as to the most appropriate formulae (e.g. sediment transport formulae, re-aeration formulae), various alternatives are provided and the option of a user-specified formula is allowed;
- it has a modular structure, enabling the user to simulate only the aspects of interest in a particular situation e.g. hydrodynamics without water quality or morphology, hydrodynamics and morphology without water quality and so on;
- it is numerically stable and robust. If errors do occur they are usually not related to numerical considerations, but to errors in input data or boundary conditions. Good error messages are provided, and;
- the user interface allows for easy viewing of the data and comparison of the output at different sites.

Specific limitations of Mike 11 in relation to aspects of the other systems under discussion are:

- its file management and system architecture are relatively old, so that despite having a good user interface, navigation between modules and the inter-comparison of data from different modules is not as efficient as in the systems with more modern system structures;
- its water quality module is more limited in the range of variables considered and the number of aspects addressed than the DELWAQ component of SOBEK. DELWAQ is a complete water quality library, applicable across both marine and freshwater systems, and has a help facility regarding which formulae are most appropriate to which situations (Delft Hydraulics 1994b);
- Mike 11 has separate modules for the simulation of the transport-dispersion of conservative constituents, nutrient dynamics and bacterial contamination. These have to be purchased separately and run separately. In some ways this can be advantageous (see above), but this can also be more limiting as all aspects may not be available for use.

In summary, Mike 11 was found to be a numerically robust and efficient modular one-dimensional modelling system, although there are other systems with an equal number of advantages.

3.2 Discussions with International Scientists

Specific discussions were held in September 1995 with Dr Marcus Lingby of the Danish Hydraulics Institute (developers of Mike 11), Dr Robert Vos, Dr Hans Los and Prof Guus Stelling of Delft Hydraulics (developers of DELFT3D and the DELWAQ water quality library), Mr Rodney Thomas of Applied Science Associates (based in the United Kingdom) and Dr Zelina Ibrahim of the University of Malaysia. These engineers and scientists are specialists in various aspects of hydrodynamic and water quality modelling and ecological response prediction. Opinions were sought from them on approaches best suited to South African estuarine conditions rather than the conditions generally found in the estuaries of Europe and North America. Additionally, further discussions were held with various staff members of Delft Hydraulics during the period November 1996 to March 1997, confirming many of the initial insights gained in September 1995. These further discussions also highlighted the role that integrated decision support systems can play in the management of estuarine water quality and the control of catchment-derived contaminants and emphasised the need for improvements and developments in biological prediction. A summary of all the discussions follows.

The discussions revolved around the differences between South African estuaries and those abroad and the implications of these differences for water quality modelling. In contrast to the large lowland coastal systems of much of Europe and North America, South Africa has small estuaries with highly variable hydrodynamic regimes and morphology. Owing to the rapidity of change in the mouth configuration of many South African estuaries, standard hydrodynamic, morphological and water quality models cannot be applied in routine fashion. Such implementations have to be undertaken with insight and time horizons and boundary conditions (e.g representative mouth configurations) chosen with care. Additionally, because the freshwater discharge can vary considerably according to seasons, dry and wet cycles and in response to episodic storms, the small estuarine water bodies of South Africa do not have long residence times. Sediment and nutrient supplies are also highly variable and may not be sufficient to sustain phytoplankton growth at all times. These factors mean that water column productivity and diversity are generally low, while the productivity of the benthic invertebrate and fish communities is high and considerable macrophyte biomass occurs. Thus the estuaries are benthically-orientated systems rather than water column-driven systems. The prediction of water quality changes and the ecological consequences of alterations in the supply of freshwater, nutrients or sediments to estuaries must therefore address this type of system rather than focusing on the prediction of water column productivity alone. Accordingly, alternative modelling approaches (expert systems, systems models, habitat evaluation techniques) need to be considered and their utility investigated further, particularly in addressing water quality and ecological response predictions.

Possible research areas and questions in this regard were suggested. These include:

- Is the growth of macrophytes, macro algae and micro algae in South African estuaries nutrient-limited or not?
- Are there other factors besides nutrients which would limit growth e.g. space, limited available habitat, reduced habitat suitability, rapidity of physical changes, harsh abiotic conditions; and what role do these factors play in determining biomass densities and species occurrence?

There was general consensus that it is advisable to link biological prediction to a realistic habitat approach as this facilitates focusing on a number of indicator species, or even communities, and assists in relating ecological results to the scale at which management decisions are made. Note that an instream flow incremental methodology (IFIM) type approach (Bovee 1982, 1995) was not advocated, as such an approach and its associated computer modelling system PHABSIM assumes direct discharge and invertebrate response links and ignores alterations in geomorphology owing to changes in discharge or feedback between the components of an ecosystem e.g. vegetation and sedimentation. Additionally, species specific life stage models are considered interesting and valuable, but not necessarily very useful in reflecting the response of the ecosystem as a whole. A more robust and appropriate approach is to select a suite of indicator species from different faunal and floral communities and assess the temporal change in habitat availability and suitability in response to changes in abiotic factors and water quality (note that the IFIM ignores all factors apart from discharge) and then predict all the species responses. This was deemed a reasonable approach for the South African situation.

Future trends in water quality modelling and management internationally include:

- the improvement of water quality models for application to the large, highly nutrient-enriched, lowland systems (e.g. the Danube, Rhine and Meuse systems) to assist in solving existing water quality problems more effectively;
- holistic management of the natural system by linking the assessment of different elements such as habitats, landscape and species and not just assessing phytoplankton or water column responses to water quality changes;
- developing linked modelling systems where hydrological, hydrodynamic, morphological, water quality and ecological models run in a coupled fashion;
- modelling water quality from catchment to sea;

- developing decision support systems which include standard models as well as modules to evaluate and assess the implications of management decisions. These systems enable managers to envisage changes and appreciate the complexity of their decisions and the influence they exert on the natural environment and its human use.

Monitoring was viewed as critically important as this supplies the data without which modelling, prediction and decision support (support tools for management) are not possible. Data processing and analysis are an integral part of monitoring (Villars 1995). If these aspects are not included in a monitoring plan then a large quantity of near useless data can be generated and not detected as such until the monitoring programme has been running for a few years. In particular, monitoring of boundary conditions is critical to the successful use of monitoring data in model calibration and implementation.

Most European and American scientists have available many years of monitoring data which reveals the progressive development of water quality problems. Effects such as algal or phytoplankton blooms owing to nutrient enrichment, fish kills owing to oxygen depletion and/or toxic compounds and bacterial contamination from waste inputs, amongst others, have been observed. The large data resource as well as the severity of the problems makes modelling of these events tractable, primarily because the natural or ambient variability is orders of magnitude smaller than that occasioned by anthropogenic impacts. This contrasts strongly with most South African systems where ambient variability and human-derived impacts are of the same order of magnitude at present.

The scientists from Malaysia shared their experiences of problems with estuaries exhibiting high abiotic and biotic variability, very few data, considerable development pressures on estuaries and the urgent need to predict and assess changes in the environment. These problems are similar to those of South African estuaries. Their response was to develop a classification system for estuaries based on hydrodynamic, water quality and use criteria. It is hoped that against this classification background, changes will be able to be observed or predicted and attention given to the reasons for, and the acceptability of, these changes. This is a useful attempt to address the management dilemma of too little background information accompanied by incremental changes in estuarine water quality and human use that may have effects that are not easily distinguished within the range of natural variability. However, this approach has not yet been applied comprehensively as only ten representative estuaries have been classified so far.

All in all, no definitive answers regarding future requirements for water quality modelling and management in South Africa could be derived directly from overseas expertise or the literature review. Instead the following foci were seen as beneficial:

- thoughtful use of existing modelling tools as the problems and issues in South African estuaries differ from those of highly developed countries with large, wedge-shaped estuaries;
- the development of integrated decision support systems for the management of estuarine water quality and the control of catchment-derived contaminants;
- improvements and developments in biological prediction (not just the prediction of water-column responses);
- the implementation of a pragmatic and well structured monitoring programme to address the present and long term need for reliable data.

3.3 Water Quality Discussion Document

The final document entitled "*Decision Support Requirements for Estuarine Water Quality Management*" (Taljaard & Slinger 1997), reflects the views of the authors as well as a wide spectrum of South Africa's water quality specialists, estuarine scientists and water managers. An important perspective, which represents a consensus among those consulted in the finalisation of the document, is that only a limited number of South African estuaries have serious water quality problems at present but that this may alter in future unless estuarine water quality management proceeds from a viewpoint of:

> *"What do we need to do to prevent the occurrence of water quality problems in estuaries?"*,

rather than allowing problems to develop and then trying to solve them. The purpose of the document therefore was to describe the present context of water quality management in South African estuaries and provide a first assessment of the decision process and the decision support techniques and information best able to support this. A summary of the contents of the document follows.

At this stage, the need to manage water quality was explained in terms of the conflict between the beneficial uses of estuarine waters (natural environment[1], recreation, mariculture, industrial use) and the developments or human activities which may influence the quality of the water. The goal of water quality management of the marine and coastal environment is to keep the water resources suitable for all designated uses. To achieve this goal, the receiving water quality objectives approach has been adopted (DWAF 1995). Water quality objectives are set for a particular marine or estuarine environment in terms of the requirements of all designated uses[1] in that area. All activities which may influence water quality are taken into account in setting these site specific objectives as well as the capacity of the natural environment to assimilate changes in water quality without detrimental effects.

Typical developments or human activities influencing water quality include:

- point source waste discharges to the estuary e.g. industrial waste, sewage;
- diffuse stormwater run-off entering along the banks of the estuary;
- constructions in an estuary e.g. marinas, bridges, which influence water circulation;
- dredging or sand mining operations within an estuary;
- manipulation of the mouth of an estuary (breaching, closure);
- water extraction for industrial use e.g. salt extraction and seafood processing;
- catchment activities which influence the quantity and quality of river water reaching an estuary e.g. dam construction, water releases from impoundments, inter basin transfers of water, afforestation, infestation by alien vegetation, point source waste discharges, diffuse run-off from agricultural lands, urban and industrial areas and informal settlements.

In terms of the general objectives for the beneficial uses of estuaries (natural environment, recreation, mariculture and industrial uses), typical water quality problems which may occur in South African estuaries include:

- eutrophication;
- suffocation of biota;
- other physiological stresses;

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1 The South African Water Law is currently under review. In the process, many of the terms and definitions presently applicable may alter. For instance, it is agreed in principle that in future the natural environment will be viewed as part of the resource rather than as a user. However, the concept of managing towards environmental and system objectives based on an understanding of the assimilative capacity of the natural system still applies.

- reduction in primary production;
- human health risks;
- interference with migration patterns;
- introduction of alien species;
- unpleasant aesthetics;
- mechanical and process interferences (clogging and blockages).

The links between typical developments or human activities and estuarine water quality problems comprise essential information required by managers in anticipating possible changes and/or problems owing to developments, or in determining the possible causes of existing problems. Accordingly, these links are explored in the document and the information is summarised in two useful ways:

- by starting with a specific development or activity and linking the resulting water quality changes to potential water quality problems and the affected beneficial uses;
- by starting with a water quality problem, identifying the causative water quality change and hence the specific developments / activities possibly causing or contributing to the problem.

Other essential information required in water quality management is knowledge of the character and functioning of South African estuaries in general, and particularly of the estuary under consideration. This assists in establishing the range of natural variability in water quality parameters as well as the capacity of the natural environment to assimilate changes without detrimental effects.

Thus, with information on the receiving water quality objectives, the development or activity, the fate in the natural environment of the materials (or the water quality parameters) concerned and knowledge of the ambient variability, the adherence or non-adherence to the site specific water quality objectives can be established. This is the central action in the *Problem Identification Phase* (often termed situation analysis) of the management decision process. If a potential or existing problem is identified, the next phase of *Identifying Feasible Management Alternatives* is implemented before the final phase of *Decision Making* is initiated. As in the initial phase, various techniques and tools can be used to support decision making. These include:

- field measurements, monitoring;
- numerical modelling;

- analytical techniques;
- statistical data analysis and assessments;
- water quality guideline documents;
- specialist inputs and assessments;
- the inputs of interested and affected parties e.g. the public, fisherman, industrialists.

The choice of an appropriate technique will depend on the information requirements for management, as summarised in Table 2. It is noteworthy that numerical modelling, particularly hydrodynamic, morphological, water quality and ecological modelling, is a fundamental tool used in establishing or predicting ambient variability and testing for compliance under various development scenarios or management options. Clearly then, predictive techniques are best able to support decision making in establishing the existence/possibility of a problem and its likely severity against a background of a highly variable natural system. Mike 11 forms one of a number of such techniques. Its role in supporting decision making in the management of the water quality of South African estuaries can thus be assessed specifically in terms of:

- its utility in simulating the natural variability of South African estuaries;
- its performance in predicting the variation in water quality occasioned by specific development scenarios or human activities;
- its capability to provide information on the likely frequency, duration and severity of exceedance of site-specific water quality objectives, that is, predicting compliance or non-compliance.

It is recommended that the document by Taljaard and Slinger (1997) be read in full, rather than only this brief summary, as it provides the wider context within which the utility and performance of Mike 11 on South African estuaries will be evaluated. It is anticipated that the decision support requirements will be refined, and the document updated, as more experience is gained in the field of estuarine water quality management and modelling in South Africa in the future.

Table 2. The decision support requirements for water quality management and the techniques appropriate to these requirements

REQUIREMENTS FOR MANAGEMENT	DECISION SUPPORT TECHNIQUES / INFORMATION
Qualify and quantify development activity	Monitoring; Specialist inputs; I&AP inputs
Set water quality objectives	Water quality guidelines; Specialist inputs; I&AP inputs
Check for compliance	Monitoring; Hydrodynamic, morphological and water quality modelling; Ecological modelling; Statistical assessment
Determine the fate of different water quality constituents in the environment	Specialist assessments; Analytical techniques; Monitoring; Numerical (chemical) modelling.
Establish the natural / ambient variability of different water quality constituents	Monitoring; Numerical modelling (hydrodynamic, morphological, water quality & ecological); Analytical techniques; Specialist assessments.

4. RESULTS: MODEL APPLICATIONS

4.1 Implementation of Mike 11 on First Case Study, Great Berg Estuary

4.1.1 Hydrodynamics

Tidal variations were recorded at six locations in the Great Berg Estuary (Figure 1) from 6 to 20 March 1990. These data were used in conjunction with the river flow data from the gauging station G1H031 (Misverstand/Die Brug), provided by the Department of Water Affairs and Forestry, in the calibration of the hydrodynamic module of Mike 11. The water levels recorded at the mouth were used as the downstream open boundary condition and a constant discharge of 0,2 $m^3.s^{-1}$ was considered representative of the summer flow at the head of tidal influence, 68 km from the mouth (the upstream boundary). Good agreement between the recorded and computed water levels was achieved with moderate adjustments to the bottom shear stress coefficients. This calibration is reported comprehensively in CSIR (1993). A notable feature of the estuary under low river flow conditions, namely the amplification of tidal variation between Kliphoek (14 km upstream of the mouth) and Kersefontein (45 km upstream of the mouth), was well represented.

4.1.2 Salinity

Winter 1989 and Summer 1990

The salinity data collected during September 1989 and January/February 1990 (Taljaard & Slinger 1992) were used in conjunction with river flow data from the gauging station G1H031 (Misverstand/Die Brug) for the period September 1989 to March 1990 to calibrate the transport-dispersion module for the simulation of salinity distributions. The river flow over this period did not vary substantially and was well reflected by the average flow rate. Thus the hydrodynamic data (water levels and flows) required as input by the transport-dispersion module of Mike 11, were generated using a constant inflow of 0,6 $m^3.s^{-1}$ as the upstream boundary condition and a sinusoidal tidal variation with spring-neap and semi-diurnal features as the downstream open boundary condition. Evaporation data in the form of mean monthly evaporation rates were provided by Ninham Shand Inc. (CSIR 1993). A dispersion co-efficient of 35 $m^2.s^{-1}$, for the section of estuary from the mouth to the Sishen-Saldanha railway bridge and a co-efficient of 25 $m^2.s^{-1}$ for the area upstream of the bridge yielded the calibration run depicted in Figure 2.

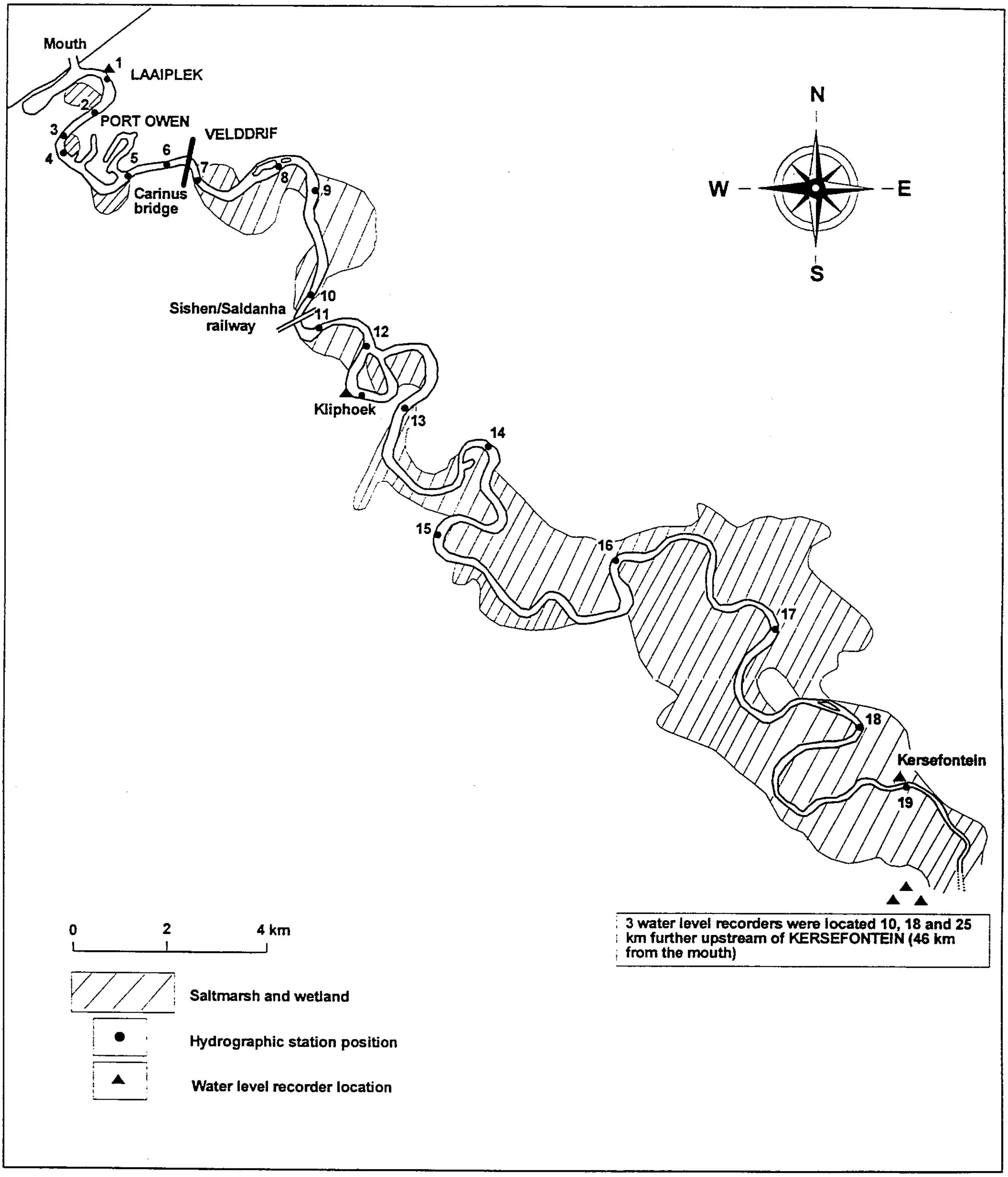

Figure 1. **Map of the Great Berg Estuary, indicating the positions of hydrographic stations at which measurements of salinity, temperature, dissolved oxygen and dissolved nutrient concentrations were taken. The positions at which water level recorders were installed during March 1990 are also indicated**

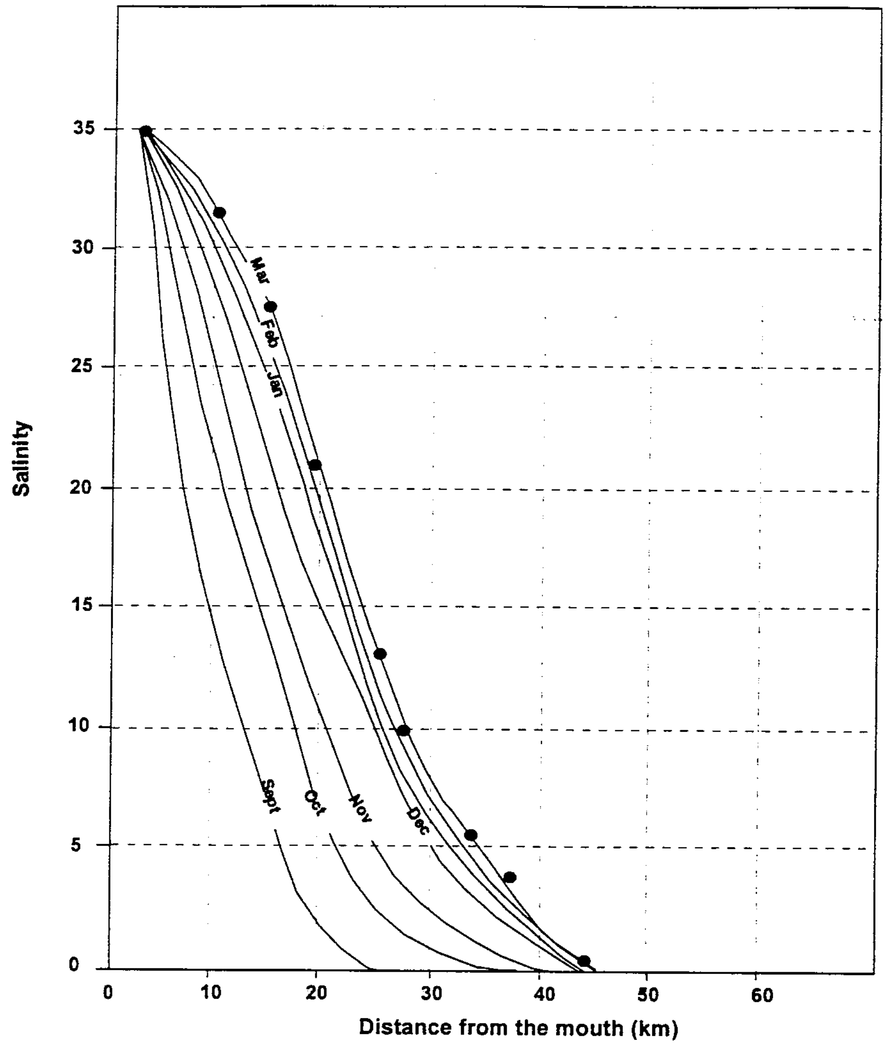

Figure 2. **Simulations of the longitudinal distribution of salinities (ppt) in the Great Berg Estuary from September 1989 to March 1990 under a constant river inflow of 0,6 $m^3.s^{-1}$. Salinities measured in the estuary in March 1990 are indicated as dots**

Winter 1995 and Summer 1996

Data from the winter 1995 and summer 1996 field expeditions were used in the simulation of the salinity distributions in the Great Berg Estuary. The hydrodynamic data required as input by the transport-dispersion module of Mike 11, were simulated using the dispersion coefficients and evaporation data as for the calibration run. Sinusoidal tidal variations co-incident in phase with the predicted spring-neap and semi-diurnal tides in St Helena Bay for the simulation periods (SAN 1995, 1996) comprised the downstream boundary conditions, and measured inflows from the gauging station G1H031 (Misverstand/Die Brug) provided the upstream boundary conditions. The hydrodynamic simulation for the high flow conditions of winter 1995 commenced on 13 August and continued for fifteen days. The river inflow rose from 4 $m^3.s^{-1}$ on 13 August to a maximum of 39 $m^3.s^{-1}$ on 24 August before decreasing again to about 13 $m^3.s^1$ on 27 August.

The transport-dispersion simulation commenced at 18:00 on 15 August, using as initial conditions the salinities measured at nine positions in the estuary on the corresponding day. A comparison of the simulated and measured longitudinal salinities on four occasions during the thirteen day transport-dispersion simulation, reveals that remarkably good agreement was obtained (Figure 3) in view of the substantial variation in inflow.

During late summer 1996, the gauged river inflow varied between a minimum of zero on 5 February and a maximum of 0,7 $m^3.s^{-1}$ on 16 February. As the gauging station is some distance above the head of the estuary and return flow via seepage is a feature of the Great Berg floodplain, representative inflow conditions were difficult to determine. After some initial testing, the hydrodynamic simulation of the low flow summer 1996 situation was initiated on 4 February at 12:00 for a period of one month using an average inflow rate of 0,3 $m^3.s^{-1}$ until 21 February and 0,6 $m^3.s^{-1}$ thereafter. The initial conditions for the transport-dispersion simulation comprised the salinities measured at twelve positions along the length of the estuary in the early afternoon of 5 February 1996 (Slinger *et al.* 1996). The measured and simulated salinities in the estuary on 21 February and 4 March 1996 are presented in Figure 4. The agreement between actual and simulated data is reasonable with deviations ascribed primarily to differences between the actual inflow conditions and those simulated.

4.1.3 Temperature

The simulation of temperature forms a component of the water quality module of Mike 11. Hydrodynamic data (water levels and flows) are required as input in addition to the temperature of the inflowing river water and the seawater. Other parameters which require specification relate to the heating of the water body by solar radiation during the daylight hours and the loss of heat by radiation from the water surface during the nocturnal hours. The feasible parameter ranges obtained from the literature and the field expeditions during winter 1995 and summer 1996 are listed in Table 3.

The temperature component of the water quality simulation was calibrated first as it is independent of the oxygen processes. The calibrations of the winter high flow condition and the summer low flow condition were undertaken using the temperature data measured in the Great Berg Estuary in September 1989, January/February 1990 and March 1990.

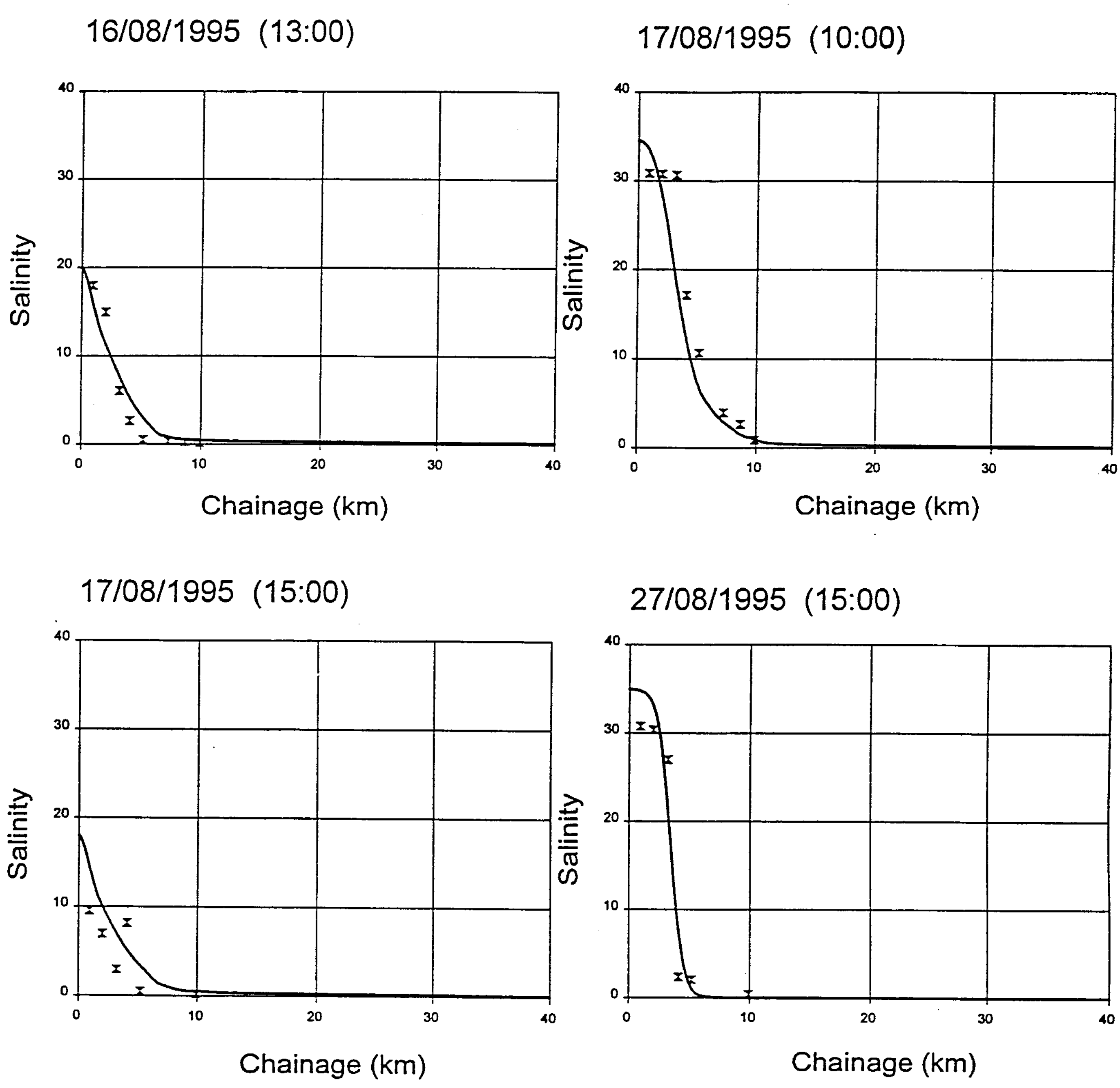

Figure 3. **Simulations of the longitudinal distribution of salinities (ppt) in the Great Berg Estuary under the high flow conditions of August 1995. Salinities measured in the estuary on four occasions during August 1995 are indicated as X**

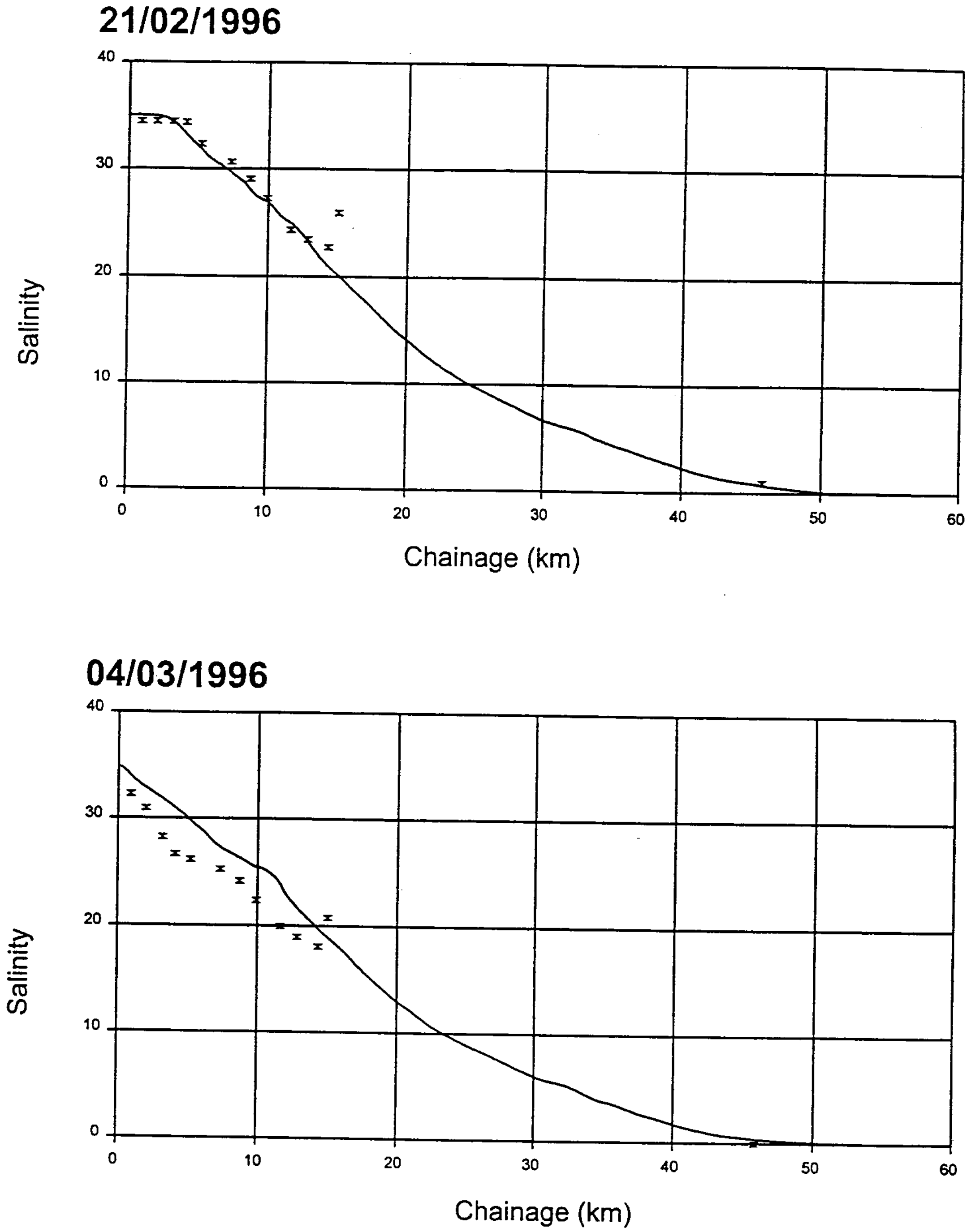

Figure 4. **Simulations of the longitudinal distribution of salinities (ppt) in the Great Berg Estuary under the low flow conditions of February/March 1996. Salinities measured in the estuary on two occasions are indicated as X**

Table 3. Parameter ranges selected for calibrating and running the Mike 11 water quality module based on findings from literature and the results of the winter 1995 and summer 1996 field excursions to the Great Berg Estuary

WQ MODEL COMPONENT	PARAMETER RANGE SELECTED
Temperature	Latitude: 35 °S Maximum solar radiation: 8000 - 15500 $kJ.m^{-2}.day^{-1}$ Displacement: 1 hr Emitted radiation: 1800 - 5000 $kJ.m^{-2}.day^{-1}$ Boundary conditions: linear interpolation of the values measured in the river (upstream) and the sea (downstream).
Dissolved oxygen (DO)	Re-aeration expression: Thyssen Re-aeration temperature co-efficient: 1,024 Respiration @ 20 °C: 0.3 g $O_2.m^{-3}.day^{-1}$ Respiration temperature co-efficient: 1,024 Photosynthesis production: 1,44 g $O_2.m^{-3}.day^{-1}$ Displacement: 1 hr Upstream boundary condition: linear interpolation of the values measured at Kersefontein Downstream boundary condition: linear interpolation of the values measured in the sea.
Biological oxygen demand (BOD)	First order decay rate k_3 at 20°C: 0,411 per day - 0,496 per day Temperature coefficient: 1,024 Half-saturation oxygen concentration: 2,0 $(g\ O_2.m^{-3})^2$ Upstream boundary condition: 0,5 $mg.l^{-1}$ - 0,9 $mg.l^{-1}$ Downstream boundary condition: 1,0 $mg.l^{-1}$ - 1,2 $mg.l^{-1}$
Sediment oxygen demand (SOD)	Lower - and lower middle reaches: 0,2 - 0,4 g $O_2.m^{-2}.day^{-1}$ Upper middle - and upper reaches: 0,02 - 0,2 g $O_2.m^{-2}.day^{-1}$

Winter 1989

The upstream boundary condition for the winter calibration (68 km from the mouth) was set as a linear variation from 16,23 °C on 18 September 1989 to 26,2 °C on 31 January 1990 and the downstream boundary condition comprised a linear variation from 15,2 °C to 16,9 °C over the same period. Thereafter, simulations were conducted using the measured data of the flood tide of 18 September 1989 as the initial conditions for temperature. The hydrodynamic run was initiated earlier (on the flood tide of 17 September 1989) to allow any transitory adjustments in the hydrodynamics, owing to possible inaccuracies in the initial conditions, to settle out before the water quality simulations commenced. Accurate simulation of measured temperatures was not possible as the longitudinal temperature distribution was only measured on three occasions from 18 to 20 September 1989. However, the temperature parameters were adjusted within their feasible ranges until a realistic thermal variation was obtained over the simulation period of 5 days (from 15:30 on 18 September to 15:30 on 23 September). The parameter choices yielding the winter calibration run for temperature are listed in Table 3. The simulated variations in temperature at six positions in the estuary are presented in Figure 5.

The temperature near the mouth of the estuary (0,5 km upstream of the downstream boundary) varies between 15,25 °C and 15,5 °C, whereas the variation in temperature in the upper reaches of the estuary is from 16,6 °C to 17 °C (Figure 5), indicating that the temperature of the sea is consistently lower than that of the inflowing river water. A marked diurnal signal is evident in the regular variation in temperature in the upper reaches (45 km and 60 km) and as far downstream as 20 km from the mouth. The irregular form of the thermal variation within the first 10 km from the mouth is indicative of the effects of tidally-driven circulation in enhancing mixing between river and seawater and thus disturbing the diurnal heating signal. The greatest variation in temperature therefore occurs at positions 5 km and 10 km upstream of the mouth, the location of the head of tidal intrusion during the high flow conditions typical of the Great Berg Estuary during winter.

Summer 1990

The boundary conditions for the summer calibration comprised a linear variation from 26,2 °C to 23,9 °C over the period 29 January to 31 March 1990 for the upstream boundary, and a linear variation from 17,03 °C to 16,9° C over the same period for the downstream boundary. Simulations were conducted using data measured at nine positions on the flood tide of 29 January 1990 as the initial conditions for temperature. The water quality simulation commenced at 16:00 on 29 January 1990. The simulated temperatures were compared with the longitudinal temperature distributions measured in the estuary on three occasions from 30 January to 1 February 1990 and the model parameters were adjusted slightly from those used in the winter calibration (the maximum solar radiation increased marginally) until a realistic thermal variation was obtained over the simulation period from 29 January to 3 February 1990. The thermal variations at six positions in the estuary (0,5 km from the downstream boundary to 8 km before the upstream boundary) are depicted graphically in Figure 6.

The temperature near the mouth of the estuary varies between 17 °C and 19,6 °C, whereas the variation in temperature in the upper reaches of the estuary is from 24,5 °C to 25 °C (Figure 6), indicating that the temperature of the sea is substantially lower than that of the inflowing river water. As in the winter situation, a marked diurnal heating pattern is evident in the thermal variation of the upper reaches (45 km and 60 km). However, in the summer situation, this signal is only unaffected by tidal circulation at the position 60 km upstream of the mouth. Both the positions 20 km and 45 km upstream of the mouth show evidence of both effects.

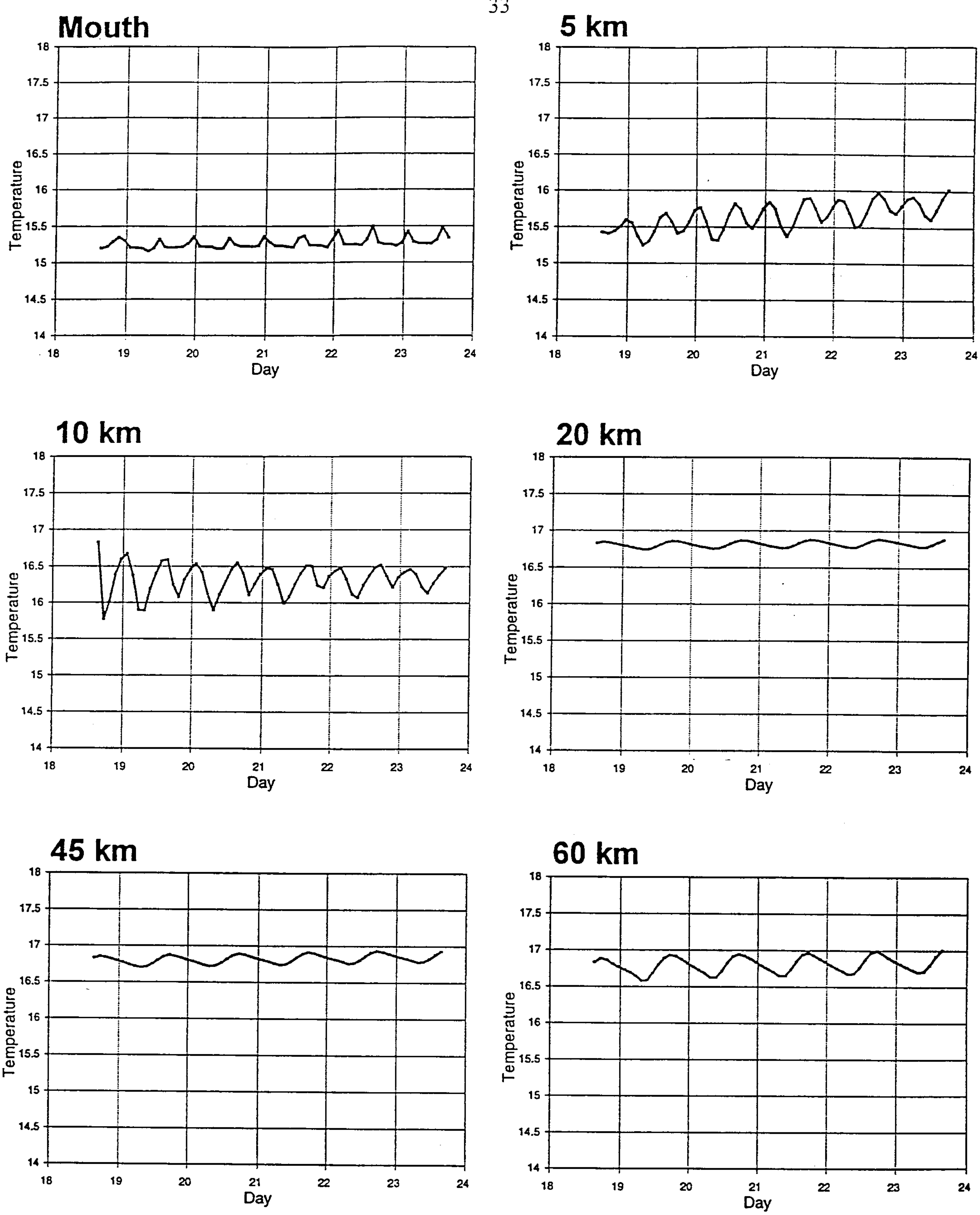

Figure 5. **Simulated temperature (°C) variations at six positions in the Great Berg Estuary from 18 to 23 September 1989 (winter 1989)**

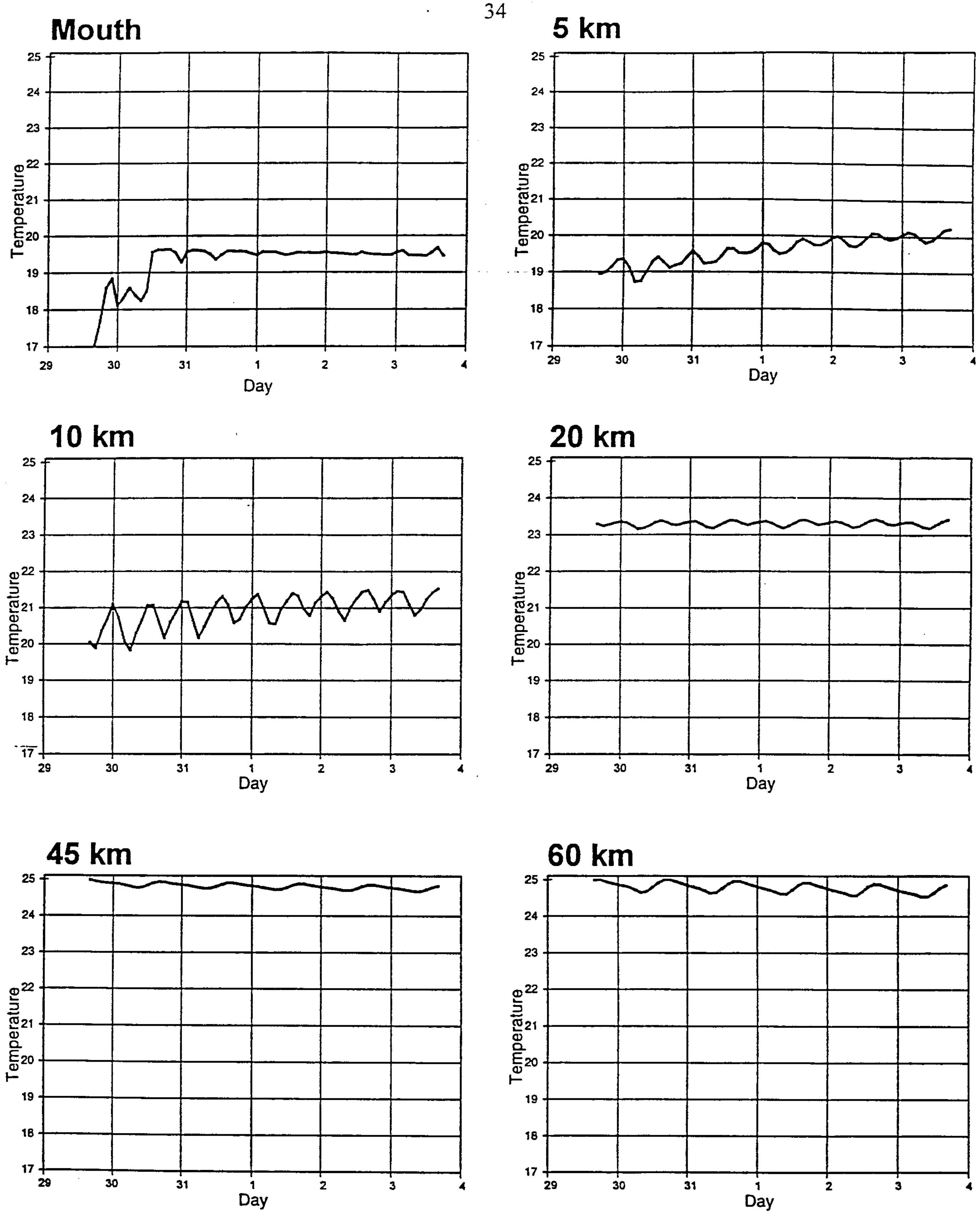

Figure 6. **Simulated temperature (°C) variations at six positions in the Great Berg Estuary from 29 January 1990 to 3 February 1990 (summer 1990)**

Within 10 km of the mouth, however, the tidally-induced circulation appears to exert the dominant influence on the temperature, probably because the difference in temperature between sea water and river water is large (> 5 °C) and so the effects of mixing between these water types overrides the more subtle effects of diurnal heating.

Winter 1995 and Summer 1996

The calibration of the temperature component of the Mike 11 water quality module was tested by applying the selected parameters without adjustment to the winter 1995 and summer 1996 situations. The water level and flow data required by the water quality module for the simulation of thermal variations were obtained from the hydrodynamic simulations conducted for winter 1995 and summer 1996 situations (section 4.1.2).

The winter 1995 water quality simulation was initiated at 18:00 on 15 August and run for a period of 13 days. The temperature at the downstream boundary was set at 13,5 °C for 24 hours before increasing to 14 °C and remaining at this level throughout the rest of the simulation period, while the temperature of the water at the upstream boundary remained constant at 15 °C. Initial conditions for temperature comprised the values measured at nine positions in the estuary on the flood tide of 15 August. The variations in thermal conditions from 15 to 18 August are depicted in Figure 7 for six positions in the estuary.

The thermal variation in the estuary during winter 1995 is particularly interesting, in that the effects of a freshette are evident. At the outset (15 to 21 August), the river flow is moderate and the tidally-induced mixing between the colder seawater (increasing from 13,5 °C to 14 °C in the first day) and the warmer estuarine water causes the temperatures at all positions in the estuary to exhibit an overall increasing trend. Additionally, the net heating effect of insolation during the day compared with heat loss during the night is evident at positions 45 km and 60 km from the mouth. The increased freshwater inflow on 22 August followed by a pulse of freshwater on 23 and 24 August causes the temperatures in the upper reaches of the estuary to decrease and then oscillate around 15 °C, the temperature of the inflowing water. At 20 km distance from the mouth, the effect is delayed by about a day and temperatures only decline on 25 August. Nearer the mouth, the effect of the tide is more pronounced and the transition from neap to spring tide which coincides with the freshwater pulse means that enhanced mixing occurs. This results in a larger amplitude of variation in temperatures, but also an overall increase in temperatures as the influence of the warmer river water extends further downstream.

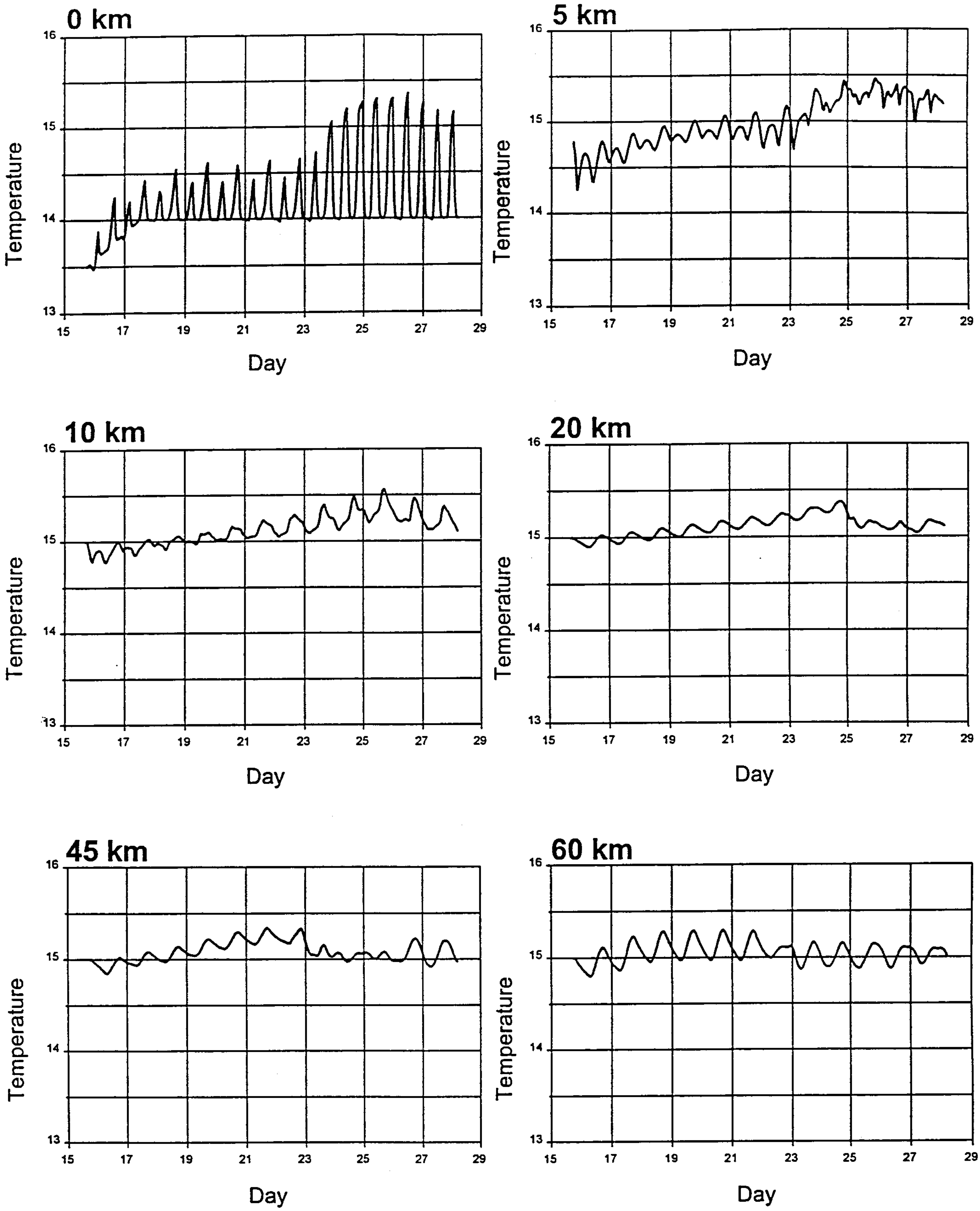

Figure 7. **Simulated temperature (°C) variations at six positions in the Great Berg Estuary from 15 to 18 August 1995 (winter 1995)**

The summer 1996 water quality simulation commenced at 14:00 on 5 February 1996 and ran until 24:00 on 23 February 1996, a period of about eighteen days. The thermal boundary conditions comprised a linear variation from 16 °C (25,9 °C) at 14:00 on 5 February to 14,3 °C (22,5 °C) at 15:00 on 21 February and were constant thereafter for the downstream (upstream) conditions. Simulations were conducted using data measured at nine positions on the flood tide of 5 February 1996 as the initial conditions for temperature. The simulation results are depicted in Figure 8 for six positions in the estuary.

Temperatures near the mouth vary between 19,3 °C and 14,3° C over the simulation period, whereas the temperature variation in the upper reaches is from 24 °C to 26 °C. A marked diurnal heating pattern is evident at positions 60 and 45 km upstream of the mouth as is the trend of decreasing temperatures from 5 February to 21 February. While the influence of diurnal heating is overridden by that of the tidally-induced circulation and mixing downstream of these reaches, the trend of decreasing temperatures is evident throughout the estuary. As in the summer 1990 situation, the difference in temperature between the mouth and the head reaches of the estuary is substantial (> 9 °C) and so the greatest variation in temperature is found where the strongest tidal mixing occurs, that is, in the lower reaches of the estuary as represented by the positions 5 km and 10 km upstream of the mouth. Comparison of the simulation results with the longitudinal temperature distributions measured in the Great Berg Estuary on 21 February yields surprisingly good agreement in that the measured values fall within the range of variation of the simulation results.

Thus, although we cannot simulate the temperature at a particular position in the estuary at a particular time, we can simulate the probable range of variation of temperatures in the Great Berg for a particular season. In conducting such simulations, the importance of information on the boundary conditions, that is, the temperatures of the river and seawater, has become evident. Initial indications are that the thermal behaviour of the estuary is strongly influenced by the upstream and downstream boundary conditions. This is particularly evident in the summer 1996 simulation in which the decreasing temperatures of the sea and river water masses impose a downward trend on the thermal variations throughout the estuary. Improved accuracy and reliability in simulating temperatures requires improved boundary input data, that is, enhanced monitoring.

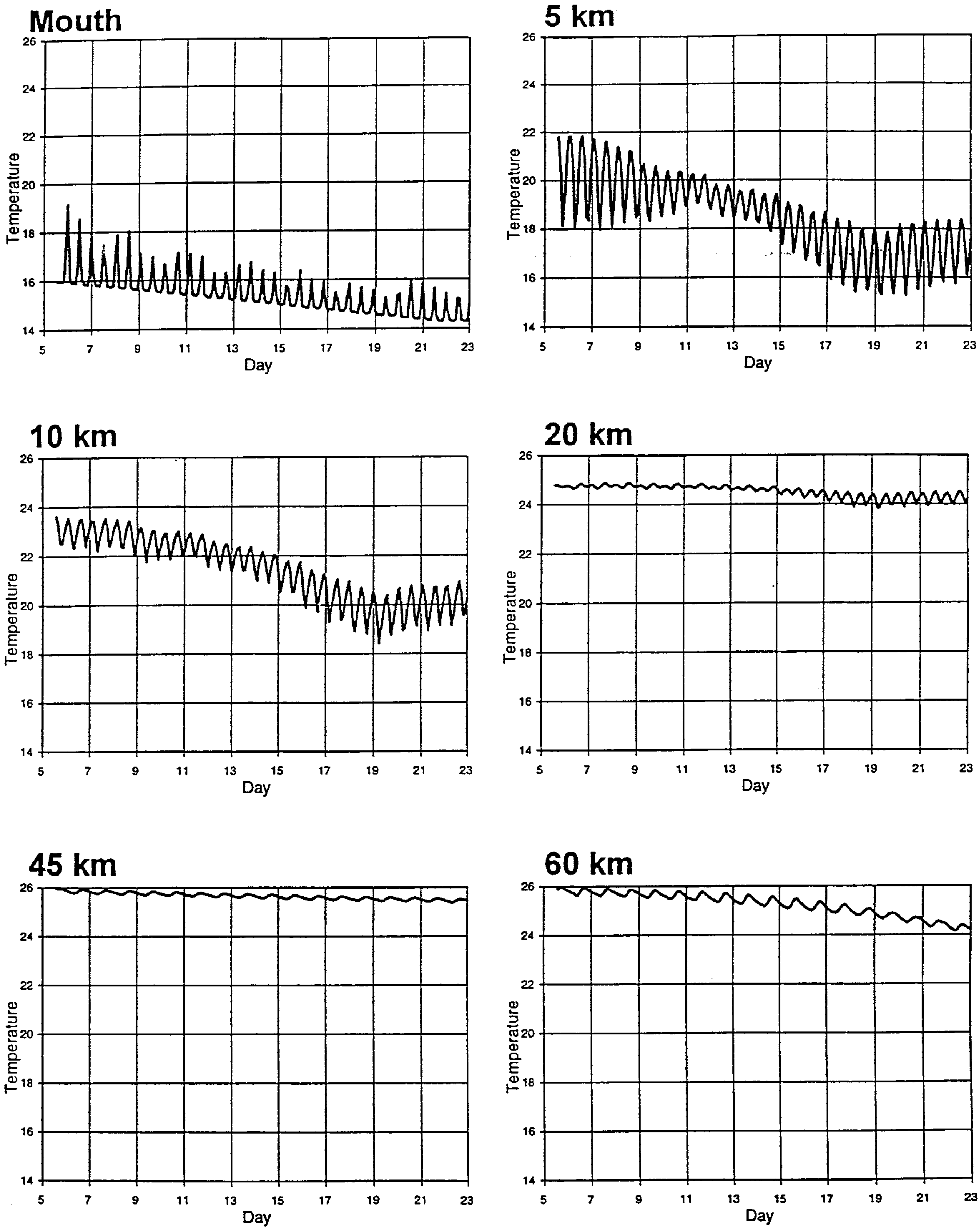

Figure 8. **Simulated temperature (°C) variations at six positions in the Great Berg Estuary from 5 to 23 February 1996 (summer 1996)**

4.1.4 Dissolved oxygen processes (including BOD)

The simulation of dissolved oxygen (DO) concentrations forms a component of the Mike 11 water quality module. Hydrodynamic data (water levels, flows and velocities) are required as input as are temperatures and the dissolved oxygen concentrations and biological oxygen demand of the sea and river water. The other parameter values required relate to the determination of the rates of re-aeration of the water body, photosynthetic production, respiration, biological decay and sediment oxygen demand. Apart from the inflow of water with high DO levels, re-aeration and photosynthesis are the only processes increasing oxygen concentrations in the water body; all other processes tend to deplete oxygen. This study is required to model the relationship between the inflows (external effects) and the internal processes increasing DO levels and the internal processes depleting oxygen and so to simulate the dynamic behaviour of dissolved oxygen concentrations in the Great Berg Estuary. The primary purpose of the winter 1995 field trip was to collect data for the specification of these rates, whereas the summer 1996 field excursions were designed to establish whether or not these rates differed substantially according to seasons. The calibration efforts for the winter high flow and the summer low flow conditions commenced using the parameter values recommended by the winter study (Slinger & Taljaard 1996). As the modelling investigation proceeded and the results of the summer study became available, the range of feasible parameter values altered slightly and adjustments were made to certain parameter values to ensure agreement between the simulated DO levels and the measurements of dissolved oxygen concentrations taken in the Great Berg Estuary during September 1989 and January/February 1990. The final parameter ranges selected are as specified in Table 3 and detailed in the subsequent description of the process followed in achieving these calibrations and conducting the simulations for the winter 1995 and summer 1996 situations.

Winter 1989

The upstream boundary condition for dissolved oxygen for the winter calibration run was entered as a linear variation from 7,32 $mg.l^{-1}$ at 15:30 on 18 September 1989 to 6,15 $mg.l^{-1}$ at 13:00 on 31 January 1990. The downstream boundary condition for dissolved oxygen was set as a linear variation from 6,6 $mg.l^{-1}$ to 6,0 $mg.l^{-1}$ over the corresponding period. The boundary conditions for the biological oxygen demand were set at a constant value of 0,9 $mg.l^{-1}$ at the head of the estuary and as a linear variation from 1,2 $mg.l^{-1}$ at 15:30 on 18 September 1989 to 1,0 $mg.l^{-1}$ at 13:00 on 31 January 1990 at the mouth. Simulations of dissolved oxygen levels and biological oxygen demand in the Great Berg Estuary commenced at 15:30 on 18 September, about twenty-four hours after the initiation of the hydrodynamic run using the Mike 11 water quality module

(Level 1). The initial conditions for the DO (and BOD) calibration run comprised a linear variation from 6,6 $mg.l^{-1}$ (1,2 $mg.l^{-1}$) at the mouth to 7,32 $mg.l^{-1}$ (0,9 $mg.l^{-1}$) at 11, 6 km upstream of the mouth and constant at this level from this position until the head of the estuary.

A range of alterations to the parameter values was attempted and numerous simulations were conducted to observe the effects of these adjustments on dissolved oxygen concentrations both in isolation and in combination. The process followed in attempting to obtain agreement between measured and simulated DO values using the Mike 11 water quality module at its simplest level included:

- selecting the initial and boundary conditions for DO and BOD with care and re-examining them regularly throughout the calibration process to ensure that they were as representative of measured conditions as possible;

- altering the selection of the O'Connor-Dubbins re-aeration expression (as initially recommended in Slinger & Taljaard 1996) to that of the Thyssen expression for streams. This resulted in lower and more realistic DO levels, especially in the upper reaches where wide pools and narrow sections are interspersed. The O'Connor-Dubbins expression indicated that significant re-aeration would occur as water passed through the narrow portions of the upper reaches. This did not accord with observations. Consequently, the Thyssen expression was adopted as being more representative of the re-aeration processes in the Great Berg Estuary;

- adjusting the respiration rate of animals and plants at 20 °C in the range 0,3 g $O_2.m^{-3}.day^{-1}$ to 1 g $O_2.m^{-3}.day^{-1}$ and observing the extent of reduction of the upstream DO levels with increasing respiration. The effects were not as dramatic as anticipated and the maximum change in average DO level over a thirty-six hour simulation was less than 0,5 $mg.l^{-1}$. A respiration rate of 0,3 g $O_2.m^{-3}.day^{-1}$ was selected even though DO levels in the middle reaches of the estuary remained high compared with measurements. A respiration rate well outside the feasible range identified in the winter 1995 field trip would be required to ensure agreement between the measured and simulated levels. It was concluded that a calibration could not be achieved by unjustifiably altering the respiration rate;

- decreasing the photosynthetic production rate was even less effective in reducing DO levels overall than increasing the respiration rate, because photosynthesis only occurs during daylight hours. A photosynthetic production rate of 1,44 g $O_2.m^{-3}.day^{-1}$, as recommended from results obtained in the winter 1995 field excursion, was therefore adopted;

- various adjustments to the BOD first order decay rate were tested as well as the effects of altering the initial and boundary conditions for BOD, because little was known regarding BOD levels in the estuary and the DO levels in the middle reaches were high compared with measurements. The boundary values tested ranged from 0,5 $mg.l^{-1}$ to 1,2 $mg.l^{-1}$ for both the river and sea water. The effects of these adjustments were confined to the head and mouth regions of the estuary. Alterations in the initial conditions for BOD in the same range had transitory effects which died out within a few hours of the start of the simulations. Increasing the BOD first order decay rate caused the influence of BOD influx to become even more confined to the head and mouth regions, whereas a lower rate meant that this influence in depleting oxygen concentrations extended a little further into the water body. However, the problem did not lie in the boundary regions and so the BOD decay rate suggested by the winter 1995 field excursion (0,411 per day) was accepted, as were the initial and boundary conditions enumerated previously.

At this stage, it was evident that an additional oxygen depleting term was required over the majority of the estuary as even significant alterations in the respiration rate, the photosynthetic production rate and the first order biological decay rate did not succeed in reducing DO levels to within the range of the measured values. Accordingly, a more complex version of the Mike 11 water quality module, which includes sediment oxygen demand in the computation of DO levels, was implemented. The results from the winter 1995 field excursion indicated that SOD was not likely to exert a significant influence on dissolved oxygen concentrations in the water body (Slinger & Taljaard 1996). It was this information which led to the initial application of the water quality module at its simplest level (which excludes SOD). However, when a sediment oxygen demand value of 1 g $O_2.m^{-2}.day^{-1}$ was applied over the whole estuary, the DO levels decreased below the measured values for the first time. The effects of SOD values in the range 0,1 g $O_2.m^{-2}.day^{-1}$ to 0,5 g $O_2.m^{-2}.day^{-1}$ were then tested and a SOD value of 0,2 g $O_2.m^{-2}.day^{-1}$ was selected as yielding fair agreement between measurements and simulated values over the period from 18 to 23 September 1989.

Between 15 and 60 km upstream of the mouth, the DO levels exhibited dynamic stability in their behaviour, initially decreasing, but then varying within the initial range (Figures 9 & 10). Within 15 km of the mouth, the agreement between measured values and the envelope of simulated values is fair (Figure 9), while the variability in the simulated values manifested at the mouth, 5 km and 10 km upstream (Figure 10) is indicative of active tidal exchange in this region. The full range of variation of the measured values cannot be simulated accurately by the Mike 11 water quality module primarily because data for the seaward boundary are lacking.

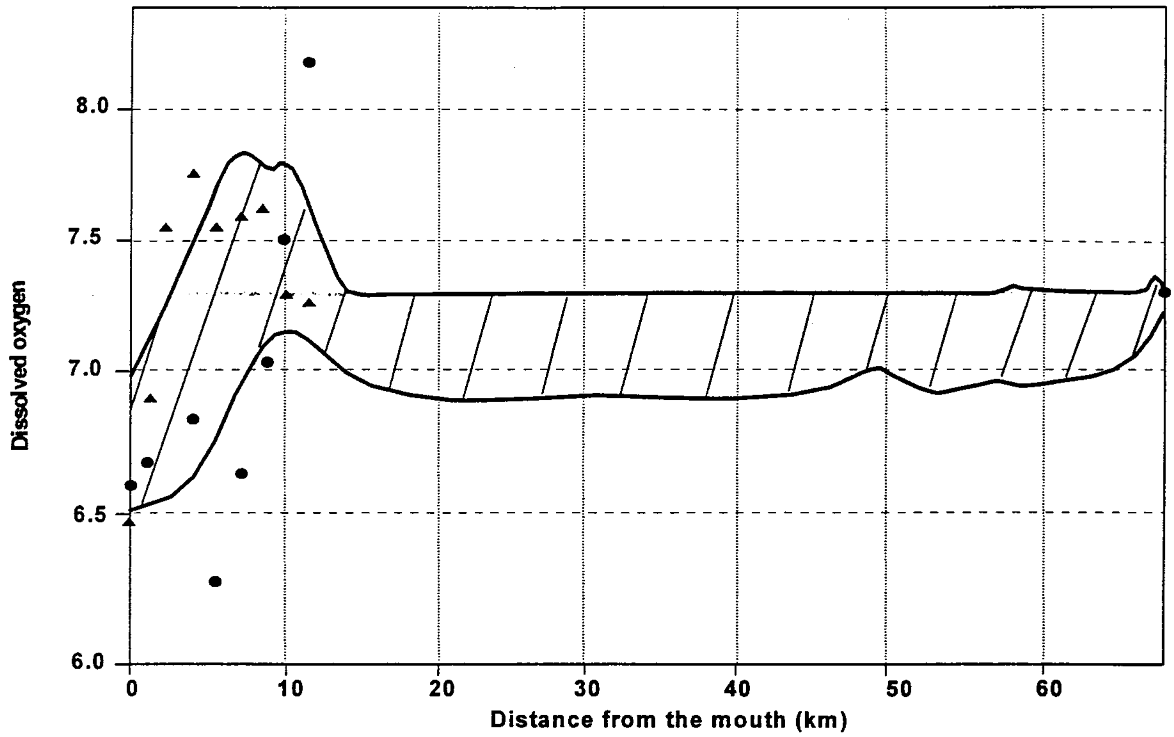

Figure 9. **Comparison between the range of dissolved oxygen concentrations ($mg.l^{-1}$) simulated by Mike 11 (shaded area) and the dissolved oxygen concentrations measured in the Great Berg Estuary on 18 (circles) and 19 (triangles) September 1989**

Accurate prediction of the dissolved oxygen dynamics within an estuary is dependent on accurate information on the variation in the DO concentrations of the inflowing waters. A further factor which may contribute to this effect is that the water quality and transport-dispersion (TD) modules of Mike 11 are de-coupled. This means that the effects of salinity (simulated by the TD module) on oxygen solubility are ignored by the Mike 11 water quality module. In South African estuaries, where DO levels and dissolved nutrient concentrations are often strongly linked to tidally driven circulation patterns, the influence of salinity may be significant. Despite these limitations, a fairly representative band of DO concentrations can be predicted for the winter 1989 condition in the Great Berg Estuary.

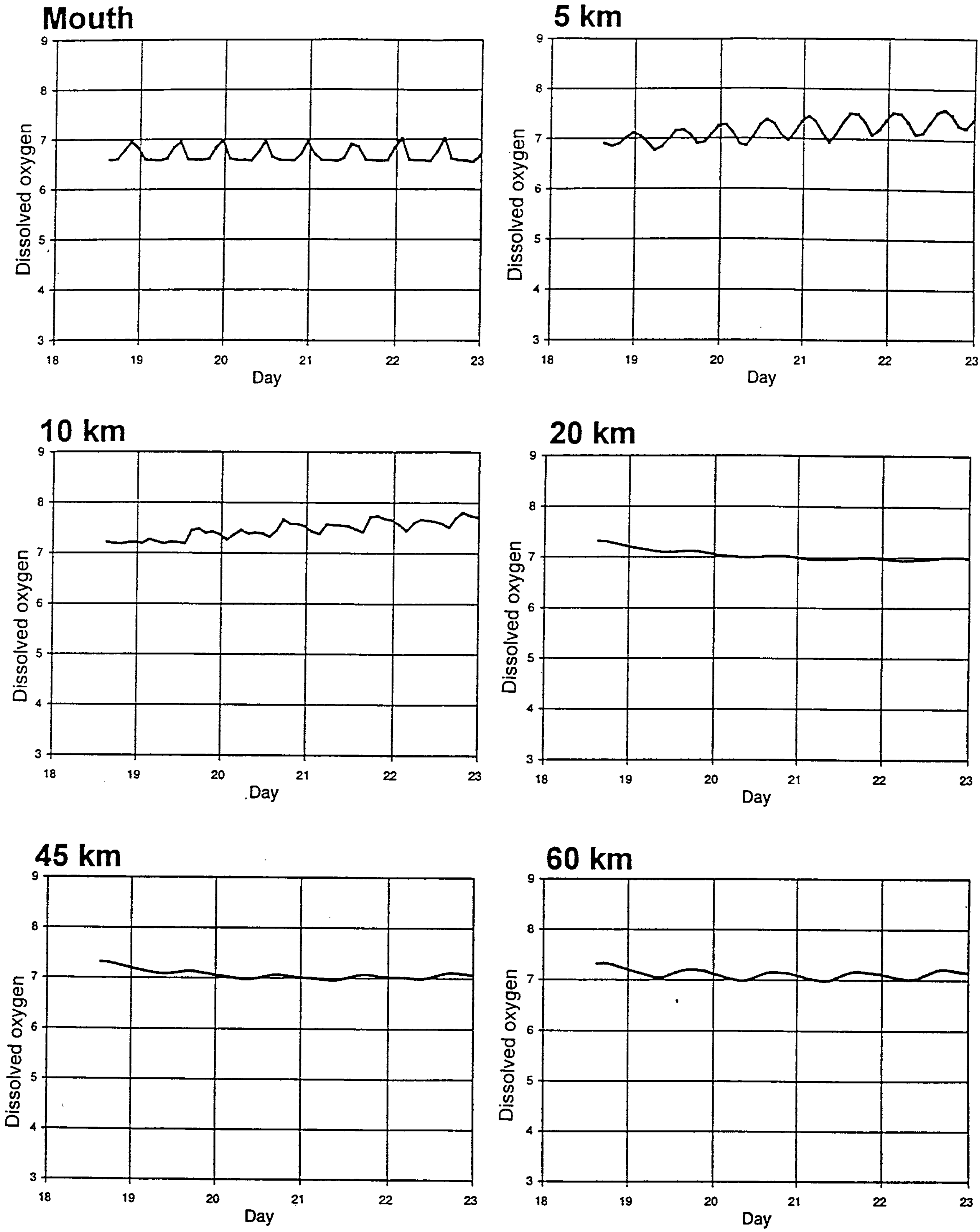

Figure 10. **Simulated dissolved oxygen concentrations ($mg.l^{-1}$) at six positions in the Great Berg Estuary from 18 to 23 September 1989 (winter 1989)**

Summer 1990

The upstream boundary condition for dissolved oxygen for the summer calibration run was set as a linear variation from 5,6 $mg.l^{-1}$ at 16:00 on 29 January 1990 to 6,15 $mg.l^{-1}$ at 16:00 on 31 March 1990. The downstream boundary condition for dissolved oxygen was set as a linear variation from 8,4 $mg.l^{-1}$ at 16:00 on 29 January 1990 to 6,4 $mg.l^{-1}$ at 12:00 on 30 January 1990 and to 6,0 $mg.l^{-1}$ at 24:00 on 31 March 1990. The boundary conditions for the biological oxygen demand were set at constant values of 0,9 $mg.l^{-1}$ at the head of the estuary and 1,0 $mg.l^{-1}$ at the mouth. Simulations were conducted using data measured at thirteen positions on the flood tide of 29 January 1990 as the initial conditions for dissolved oxygen, while the initial BOD conditions comprised a linear variation from 1 $mg.l^{-1}$ at the mouth to 0,9 $mg.l^{-1}$ at the head of the estuary. Simulations of dissolved oxygen levels and biological oxygen demand in the Great Berg Estuary commenced at 16:00 on 29 January 1990, about twenty-four hours after the initiation of the hydrodynamic run, using the Mike 11 water quality module and including sediment oxygen demand.

Using the DO and BOD parameter set from the winter calibration run with the first order BOD decay rate set at 0,496 per day (as determined in the summer 1996 study (Slinger *et al.* 1996)) and the thermal parameter set from the summer 1990 calibration run, dissolved oxygen and biological oxygen demand levels were computed over the period 29 January to 3 February 1990. The simulated DO levels were compared with the longitudinal DO distributions measured in the estuary on 30 January 1990. The initial results indicated that the simulated DO values within 15 km of the mouth exceeded those measured in the system, whereas those predicted for the middle and upper reaches were too low in comparison with measurements. Because in this study the Thyssen expression for re-aeration was definitely considered more appropriate for the Great Berg Estuary than any other feasible choice (the O'Connor-Dubbins expression, in particular) and because there was little justification for altering the process parameters related to photosynthetic production, respiration or biological decay, which have mechanisms internal to Mike 11 to compensate for the differences in temperature between the summer and winter situations, the only further parameter adjustment which appeared reasonable was that of the sediment oxygen demand. Accordingly, sediment oxygen demand values in the range from zero to 0,5 g $O_2.m^{-2}.day^{-1}$ were applied over the whole estuary and then over sections of the estuary. The final parameterisation selected as yielding the most realistic variations in simulated DO levels over the simulation period from 29 January to 3 February 1990 was that of higher SOD values in the lower reaches (0,4 g $O_2.m^{-2}.day^{-1}$) than for winter 1989 and low values at distances of more than 20 km upstream of the mouth (0,02 g $O_2.m^{-2}.day^{-1}$). The resulting DO variations over the length of the estuary and at six positions in the estuary are depicted graphically in Figures 11 & 12.

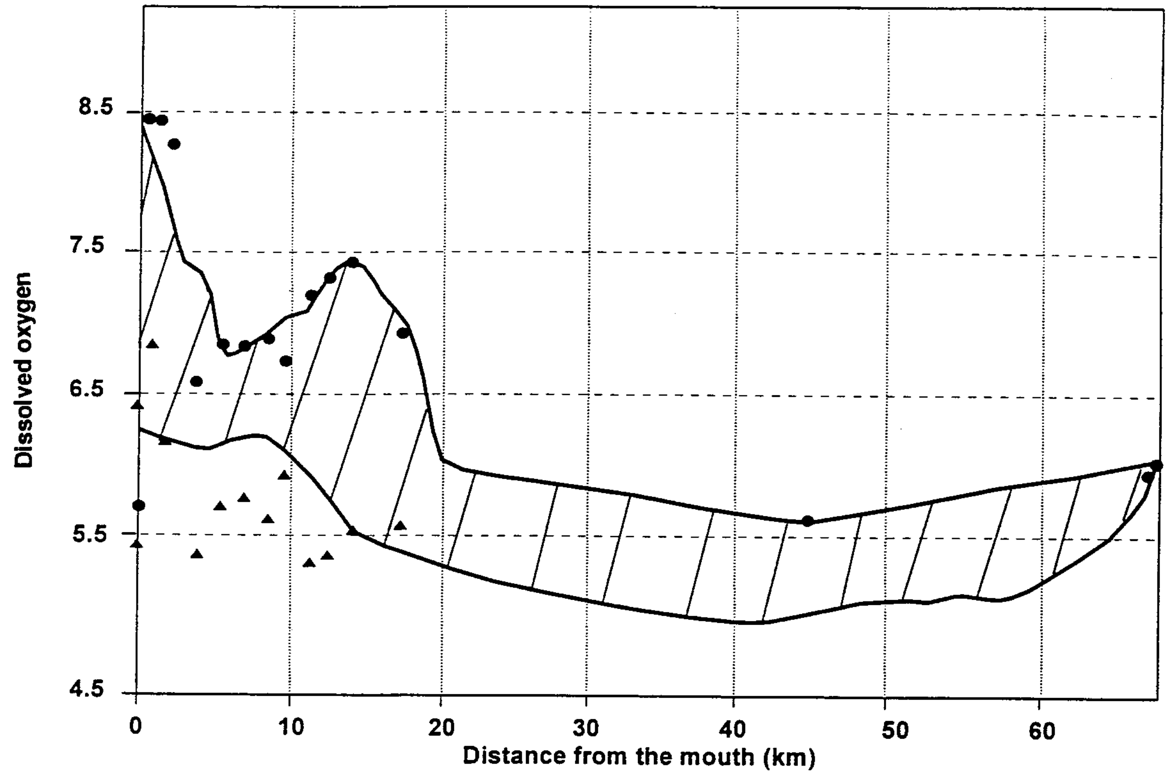

Figure 11. **Comparison between the range of dissolved oxygen concentrations (mg.l^{-1}) simulated by Mike 11 (shaded area) and the dissolved oxygen concentrations measured in the Great Berg Estuary on 29 (circles) and 30 (triangles) January 1990**

Over the last three days of the five day simulation period, the dissolved oxygen concentrations at distances greater than 20 km from the mouth exhibit dynamically stable behaviour (Figures 11 & 12). After an initial decrease in value, they settle into a repetitive cycle and do not decrease much below 5 mg.l^{-1} over the simulation period. In contrast, the variation in DO levels nearer the mouth is substantial with the high dissolved oxygen concentrations of 29 January dissipating rapidly so that fairly low values are simulated from 30 January onwards. Despite this trend, the simulated values do not decrease as far as the measured values. As in the winter 1989 case, the inability of the simulated DO levels to match the level of variation exhibited by measurements, particularly in the lower reaches of the estuary, is primarily ascribed to insufficient information regarding the seaward boundary. The fact that the influence of salinity on oxygen solubility is not taken into account in the Mike 11 water quality module may also contribute to the differences prevailing between measured and simulated values in the highly saline, lower reaches of the estuary. However, given the restrictions imposed by the limited data available for calibration of the summer 1990 condition, reasonable coherence between measurements and simulated variations is deemed to have been achieved.

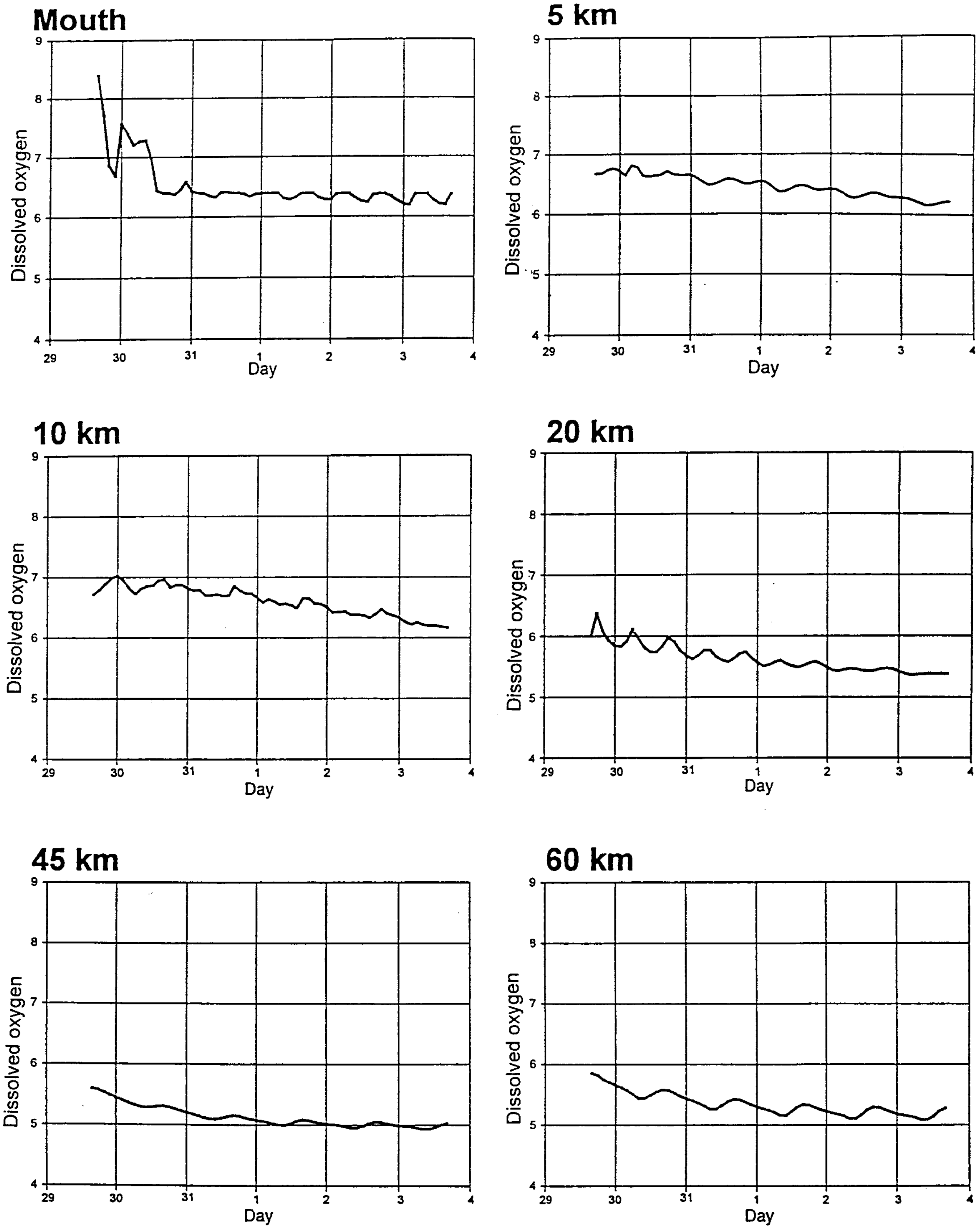

Figure 12. **Simulated dissolved oxygen concentrations (mg.l^{-1}) at six positions in the Great Berg Estuary from 29 January 1990 to 3 February 1990 (summer 1990)**

Winter 1995

The upstream boundary condition for dissolved oxygen was set at 8 $mg.l^{-1}$ throughout the simulation period, whereas the downstream boundary condition was initially constant at 5,5 $mg.l^{-1}$ from 18:00 on 15 August to 18:00 on 16 August, then increased linearly to 6,8 $mg.l^{-1}$ at 12:00 on 17 August and remained constant at this level. The upstream and downstream boundary conditions for BOD were constant throughout the simulation and comprised 0,9 $mg.l^{-1}$ and 1,0 $mg.l^{-1}$, respectively. The initial conditions for DO comprised the values measured at nine positions in the estuary flood tide on 15 August 1995.

The initial BOD conditions were set as a linear variation from 1,0 $mg.l^{-1}$ at the mouth to 0,9 $mg.l^{-1}$ at 11,6 km upstream of the mouth and constant at this level until the head of the estuary. Using the parameter set from the winter 1989 calibration run, the simulation of dissolved oxygen concentrations and BOD levels was initiated at 18:00 on 15 August and run for about thirteen days. The simulation results are depicted graphically in Figures 13 & 14.

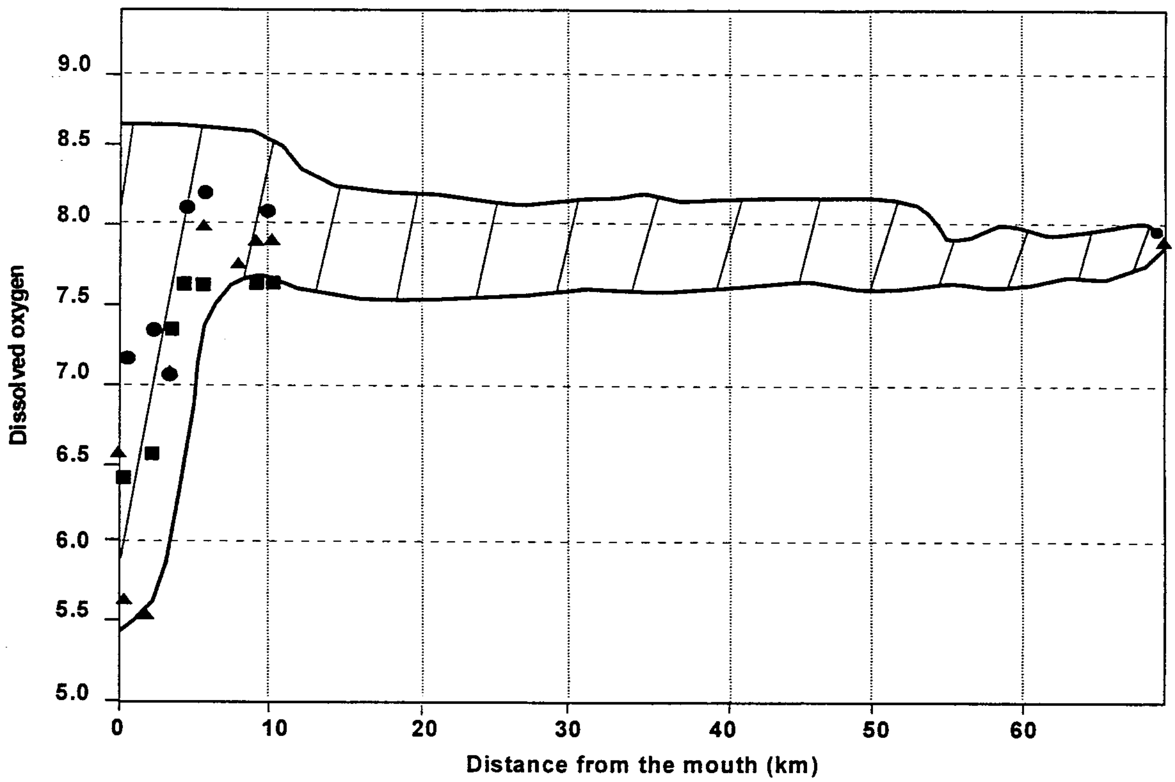

Figure 13. **Comparison between the range of dissolved oxygen concentrations ($mg.l^{-1}$) simulated by Mike 11 (shaded area) and the dissolved oxygen concentrations measured in the Great Berg Estuary on 15 August (circles), 18 August (triangles) and 28 August (squares) 1995**

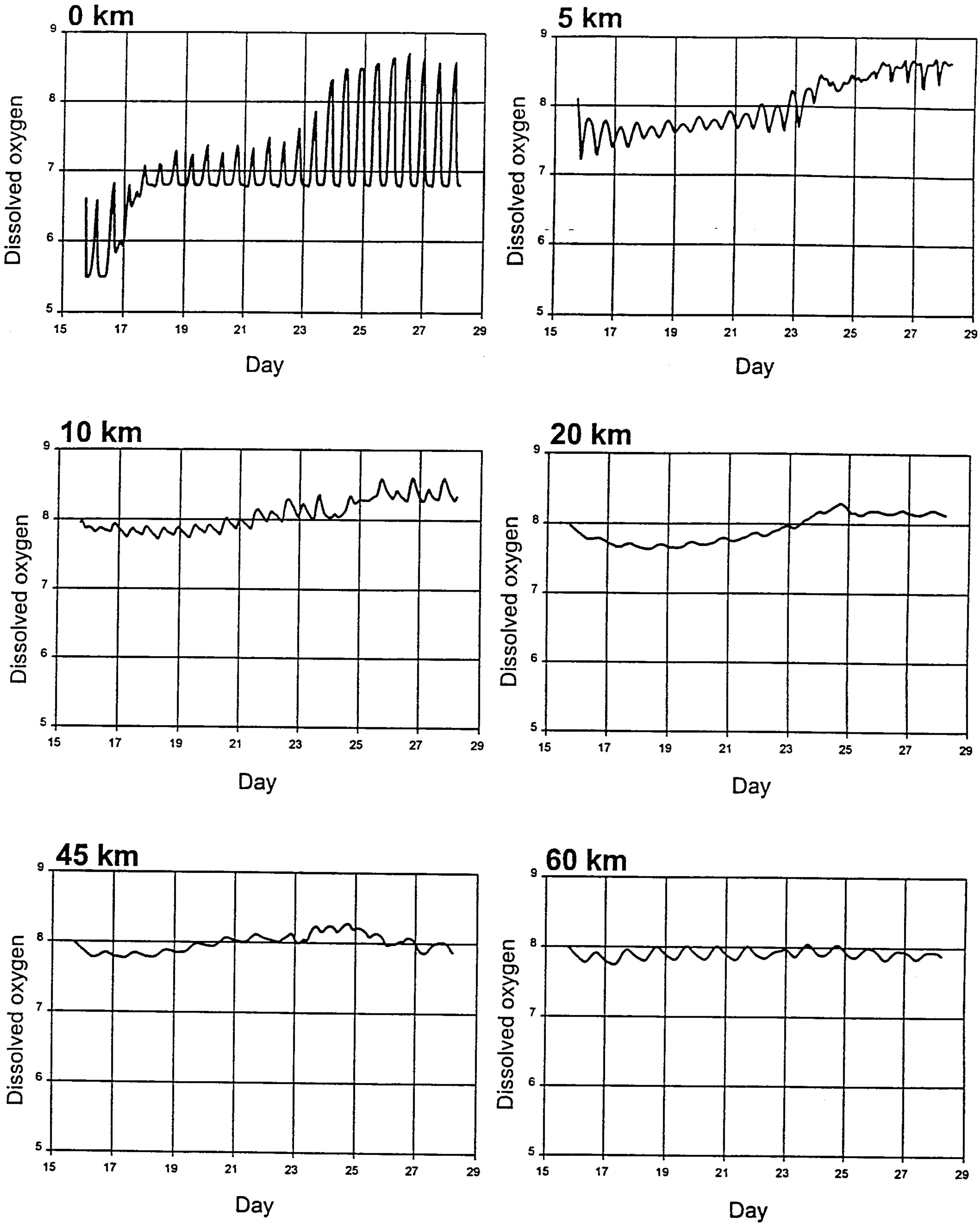

Figure 14. **Simulated dissolved oxygen concentrations ($mg.l^{-1}$) at six positions in the Great Berg Estuary from 15 to 18 August 1995 (winter 1995)**

Between 15 and 60 km upstream of the mouth, the DO levels exhibit dynamic stability in their behaviour. Within 15 km of the mouth, the agreement between measured values and the envelope of simulated values is good (Figure 13) and the full range of measured variation is simulated. In contrast to the previous simulations of dissolved oxygen concentrations, the strong influence of high volume river flows means that the variability in the lower reaches is somewhat reduced and only at the mouth are substantial variations in DO concentrations evident between flood and ebb tides. The effect of the pulse of oxygenated freshwater from 22 to 24 August is evident in the rise in DO levels at positions 5 km, 10 km, 20 km and 45 km upstream of the mouth over the last three or four days of the simulation (Figure 14). All in all, the variations in dissolved oxygen levels in the Great Berg Estuary over the winter 1995 period are considered to accord well with measured and expected dynamic behaviour and the envelope of variation is deemed realistic.

Summer 1996

The upstream (downstream) boundary condition for dissolved oxygen was set as a linear variation from 3,75 $mg.l^{-1}$ (6,23 $mg.l^{-1}$) at 14:00 on 5 February 1996 to 4,34 $mg.l^{-1}$ (5,24 $mg.l^{-1}$) at 15:00 on 21 February 1996 and constant thereafter. The boundary conditions for BOD were set at constant levels of 0,5 $mg.l^{-1}$ at the head of the estuary and 1,0 $mg.l^{-1}$ at the mouth on the basis of the summer 1996 field observations (Slinger *et al.* 1996). The initial conditions for DO were specified as the values measured at thirteen positions along the length of the estuary on 5 February 1996. The initial BOD conditions were set as a linear variation from 1,0 $mg.l^{-1}$ at the mouth to 0,5 $mg.l^{-1}$ at 11,6 km upstream of the mouth and constant at this level until the head of the estuary. The water quality simulation commenced at 14:00 on 5 February 1996 and ran until 24:00 on 23 February 1996, a period of about eighteen days, using the full parameter set from the summer 1990 calibration run. The simulation results are presented in Figures 15 & 16.

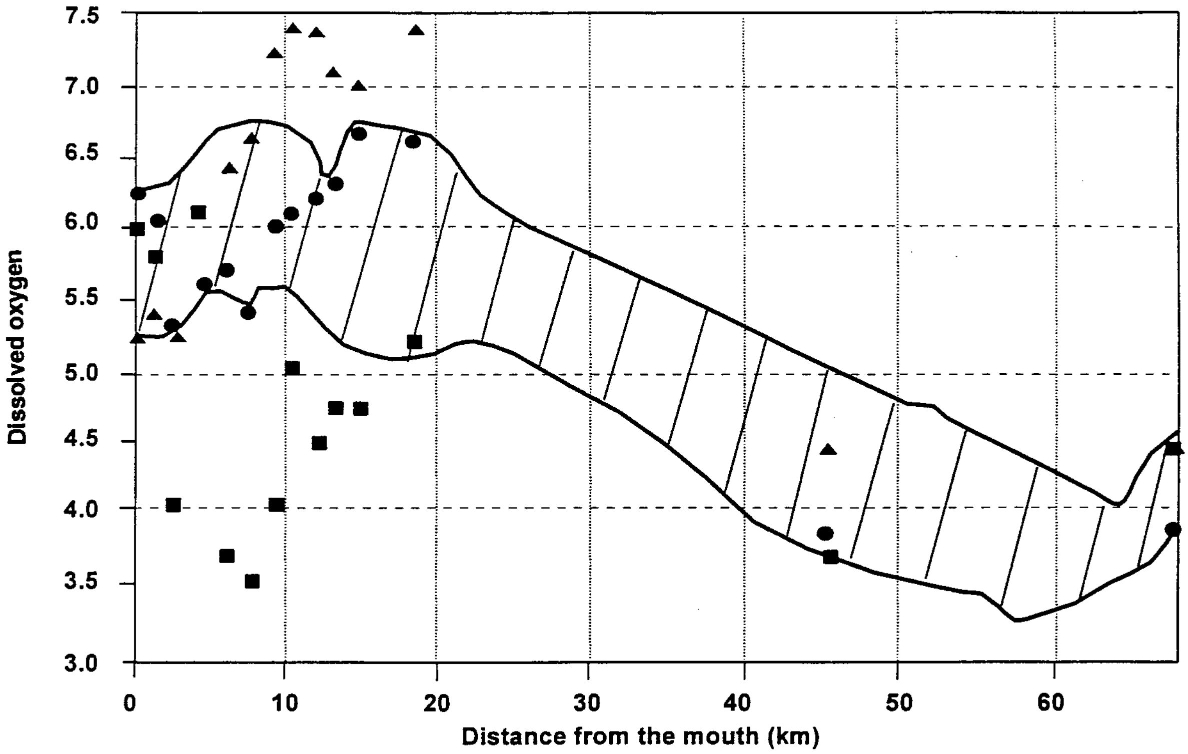

Figure 15. **Comparison between the range of dissolved oxygen concentrations (mg.l^{-1}) simulated by Mike 11 (shaded area) and the dissolved oxygen concentrations measured in the Great Berg Estuary on 5 February (circles), 21 February (triangles) and 4 March (squares) 1996**

As in summer 1990, the envelope of simulated DO values shows characteristically lower values in the upper reaches of the Great Berg Estuary compared with the values for the lower 20 km of the estuary (Figure 15). The limited variation in the simulated DO levels in the lower, more saline reaches compared with the measured values is also a familiar feature from the winter 1989 and summer 1990 calibration runs. However, the fact that the envelope of simulated DO values falls almost midway within the measured range of variation induces some confidence in the water quality simulation. Additionally, the dynamic stability exhibited in the last four days of the eighteen day simulation period (Figure 16) provides an indication that a fairly representative band of DO concentrations is predicted for the inflows, thermal conditions and DO and BOD influxes particular to the summer 1996 situation in the Great Berg Estuary.

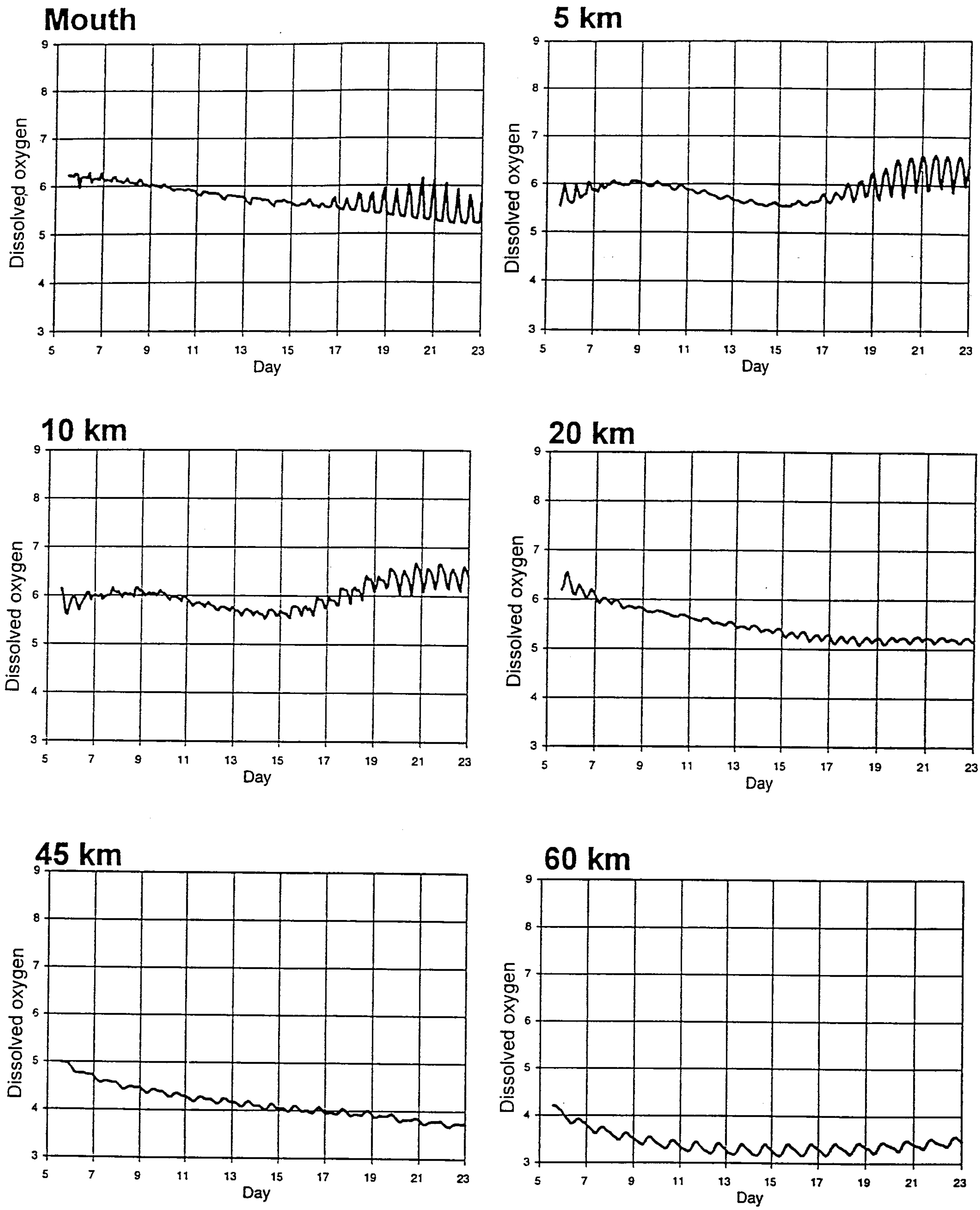

Figure 16. **Simulated dissolved oxygen levels at (mg.l^{-1}) six positions in the Great Berg Estuary from 5 to 23 February 1996 (summer 1996)**

4.1.5 Dissolved nutrients

Winter 1995

Results from the winter 1995 field excursion indicated that apart from dissolved reactive phosphate, the dissolved nutrient concentrations were generally linearly related to salinity (Table 4). This implies that the major determinant of the nutrient dynamics in the estuary under the high flow conditions of winter is the tidally-driven circulation and mixing rather than processes of regeneration or depletion in the estuary. In view of the fact that the Mike 11 water quality module is de-coupled from the transport-dispersion module, which is used to simulate salinities, it seemed pointless to attempt the simulation of dissolved nutrient concentrations in relation to DO and temperature dynamics, particularly as such simulations were likely to be less accurate than estimations based on the derived linear relationships. Instead, it is recommended that, apart from dissolved reactive phosphate-P and perhaps dissolved reactive silicate-Si, the dissolved nutrient concentrations should be estimated from the salinity predictions (Section 4.1.2) using the linear relationships given by Slinger & Taljaard (1996). In the case of dissolved reactive phosphate-P and dissolved reactive silicate-Si, there is insufficient information on the bio-geochemical processes affecting these dissolved nutrients for accurate prediction of their concentrations.

Table 4. **The r^2 values for the regression analyses of the dissolved nutrient concentrations against salinities measured in the Great Berg Estuary on 15 and 16 August during winter 1995**

DISSOLVED NUTRIENTS	15 AUGUST 1995	16 AUGUST 1995
Dissolved nitrite-N	0,96	0,90
Dissolved nitrate-N	0,88	0,54
Dissolved total ammonia-N	0,85	0,93
Dissolved reactive phosphate-P	0,55	0,16
Dissolved reactive silicate-Si	0,77	0,00

Summer 1996

In contrast to the winter situation when the concentrations of dissolved nutrients were strongly related to the circulation in the estuary and the mixing of the riverine and marine water masses, no clear relationships to salinity distributions are evident in summer. In fact, if one assumes conservative mixing between the concentrations of dissolved nutrients in the sea and the river, then there are indications that:

- removal of nitrite and nitrate occurs in the estuary during summer;
- removal of total ammonia may also be occurring (clear trend later in the summer on 21 February and 4 March, but indistinct on 5 February);
- dissolved total inorganic nitrogen (nitrite, nitrate plus total ammonia) removal occurs in the estuary during summer;
- dissolved reactive phosphate is generated in the lower and lower-middle reaches of the estuary or may have entered the estuary earlier in the summer when upwelled water intruded.

As there is little information available on the bio-geochemical processes resulting in the removal or generation of these nutrients in the Great Berg Estuary and thus not even a conceptual understanding of nutrient dynamics in this large floodplain estuary, attempting to simulate the concentrations of dissolved nutrients was not considered feasible at this stage. This is an area where future development of modelling capabilities could occur.

4.1.6 Low oxygen event (March 1994)

During March 1994, the worst mass mortality of marine biota ever recorded along the South African coast occurred in St Helena Bay on the West Coast when an estimated 60 tons of rock lobster and 1500 tons of fish were washed ashore and most mussels, limpets, sea urchins and other intertidal life died (Matthews 1995). According to Matthews and Pitcher (1996), this extreme event was caused by an extensive red tide which was trapped in St Helena Bay and subsequently decayed causing dissolved oxygen concentrations to decrease, especially in the bottom waters. On 17 March 1994, sixteen days after the initial onset of a series of red tides in the bay, the seawater along a 30 km stretch of coast was black and entirely depleted of oxygen (Gosling in the Cape Times on 18 March 1994). Oxygen concentrations of below 0,5 $mg.l^{-1}$ were measured in the bay on 18 March 1994 and hydrogen sulfide concentrations greater than 50 $\mu mol.l^{-1}$ were measured in the bottom water (Matthews & Pitcher 1996). Marine life died because of suffocation or hydrogen sulfide poisoning.

Both the hypoxic water of the preceding days and the de-oxygenated water of 18 March entered the Great Berg Estuary and caused extensive mortalities to both the resident estuarine fauna in the lower reaches and the marine life which entered the estuary in search of refuge. The extent to which the Great Berg Estuary acted as a refuge, the time period for which estuarine fauna were exposed to low oxygen concentrations and the limit of intrusion of oxygen-depleted water are interesting and ecologically relevant questions. It is anticipated that an application of the Mike 11 modelling system, including water quality, may assist in resolving these questions.

The first step in the implementation of the model was the collation of all relevant scientific data. These comprised:

- Measurements of all relevant water quality parameters in the lower reaches of the estuary on 24 January 1994 (Mr T Harrison pers. comm.);
- Measurements of salinity, temperature and dissolved oxygen concentrations in St Helena Bay on 18 March and along a transect extending from 1,5 km offshore up the Great Berg Estuary to the Sishen/Saldanha Bridge on 24 March, by the Sea Fisheries Research Institute (Dr G Bailey pers. comm.). Isolated measurements of hydrogen sulphide levels were also taken on 24 March;
- Measurements of temperature and salinity along the length of the estuary on 21 March (J H Slinger pers. obs.);
- Observations of biotic mortalities in the estuary on 25 March (Mr I Bickerton pers. comm.).

Additionally, newspaper and popular articles were reviewed and information from all of these sources was collated to provide a picture of the sequence of events (Table 5). The newspaper articles were particularly useful as they described the early stages of the 'black tide', whereas the scientific data were collected towards the middle and end of the event.

Table 5. Sequence of events in the development of the 'black tide' of March 1994

DATE IN MARCH 94	OBSERVED / RECORDED EVENTS	SOURCE
1	Onset of first of a series of red tides in St Helena Bay	Matthews & Pitcher (1995)
13 - 17	Extensive red tide in St Helena Bay, transition to black water, massive mortalities of marine biota by 17-03-94	Yeld (1994)
18	Seawater black over a 30 km stretch of the bay (directly in front of the mouth of the Great Berg Estuary), more marine mortalities, low DO concentrations in the bottom water of St Helena Bay	Gosling (1994), Matthews (1994), Matthews & Pitcher (1995)
21	Black water extends into the estuary past the Carinus Bridge, but not as far as Fishwater Flats, dead fish and invertebrates line the banks of the lower estuary	Slinger (pers. obs.)
22	Estuary water dark and smelly in the lower reaches	Underhill (1994)
24	Water in Great Berg Estuary low in oxygen, particularly in the lower reaches	Matthews & Pitcher (1995)
25	Water in the lower reaches of the estuary remains black	Bickerton (pers. comm.)

Hydrodynamics and salinity

The simulation of conditions in the Great Berg Estuary during March 1994 was subsequently undertaken. The tidal variations at the seaward boundary were assumed to concur with predicted conditions in St Helena Bay over the simulation period from 12:00 on 13 February to 12:00 on 31 March 1994 (SAN 1994), while the upstream boundary condition was assumed typical of the summer situation in the Great Berg Estuary and was set at a constant inflow of 0,6 $m^3.s^{-1}$. The salinities measured in the estuary over high water at spring tide on 21 February 1996 were assumed representative of the situation in the estuary at 06:00 on 14 February 1994, that is over high water at spring tide in summer 1994, and formed the initial conditions for the transport-dispersion simulation. At the height of the flood tide on 21 March 1994, the simulated salinities in the lower estuary are a little less than those measured (Figure 17). Similarly, on 23 March 1994, the simulated salinities were slightly lower than those measured.

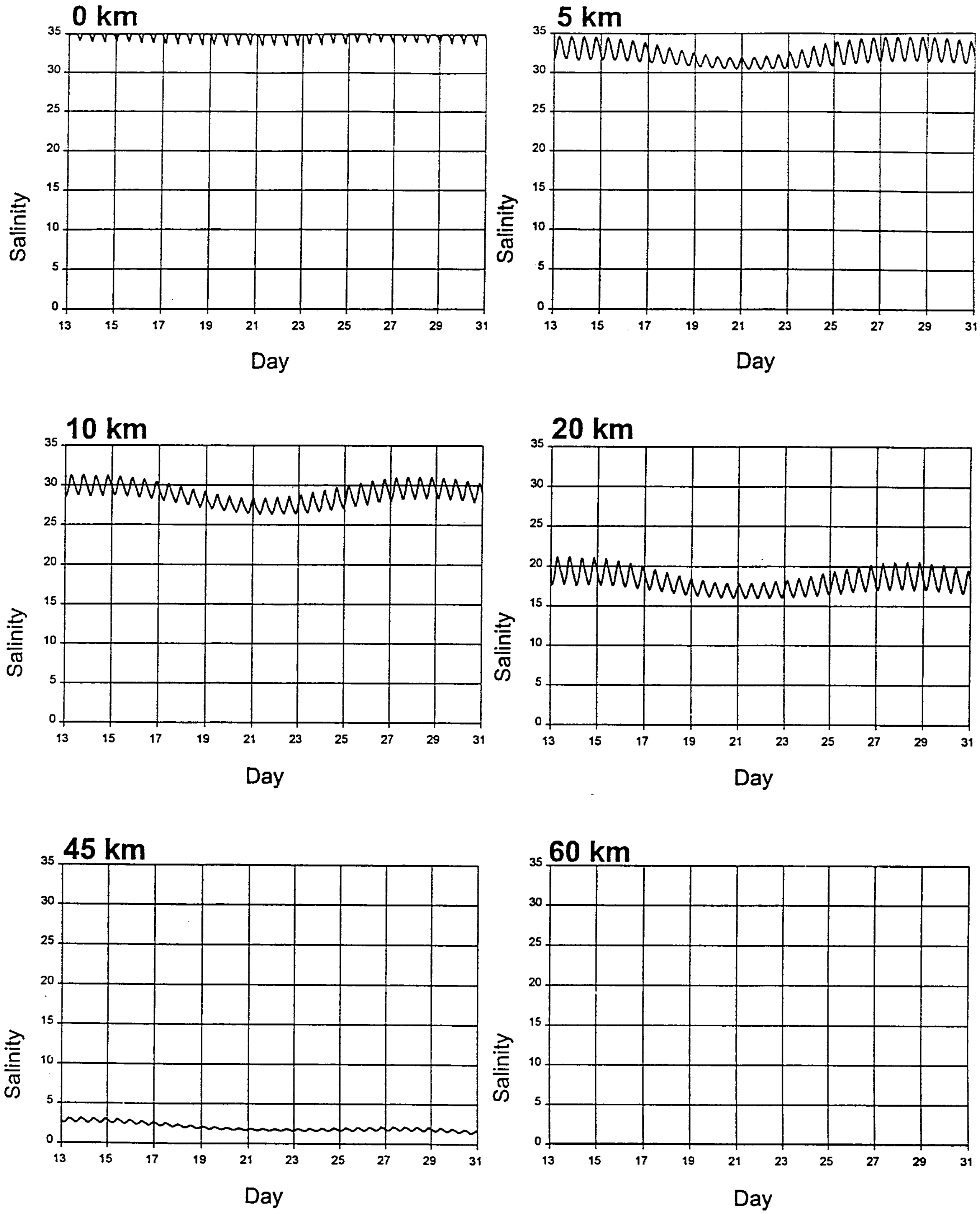

Figure 17. **Simulated salinities (ppt) at six positions in the Great Berg Estuary from 13 to 31 March 1994 (low oxygen event)**

Accordingly, minor adjustments to the freshwater inflow rate and tidal conditions were made and the predicted salinity distributions compared with measurements. Lower inflow rates did not yield better agreement. Enhanced tidal amplitudes increased the salinities in the system, but not in accordance with the measured distributions. In view of the inaccuracies in the boundary conditions and the assumed initial conditions, and the unsatisfactory response to small, reasonable adjustments in the boundary conditions, further adjustments to these conditions or the initial conditions in an attempt to achieve slightly better agreement were deemed highly artificial and consequently were not undertaken. The fact that salinities in the lower reaches are slightly underestimated and hence, the extent of tidal influence underestimated, is taken into account in the interpretation of the water quality results.

Temperature

The water quality simulation commenced at noon on 13 March and ran until 31 March, using the parameter set as for the summer 1990 and summer 1996 simulations. The upstream thermal boundary condition was set at 23 °C throughout the eighteen day simulation period. The downstream boundary conditions for temperature consisted of a linear variation from 13 °C on 13 March to 16,5 °C on 15 March, 18,5 °C on 16 March, 19 °C on 17 March, 19,5 °C on 21 March, 21 °C on 24 March and 21 °C on 31 March. The initial conditions for temperature consisted in a linear variation from 13 °C at the mouth, to 16,5 °C at 5 km upstream, 20 °C at 10 km upstream, 23 °C at 20 km upstream and constant at this level until the head of the estuary. Much of the information on which these thermal conditions are based was obtained from newspaper reports of the unusually high temperatures in the sea. The simulation results for temperature are depicted in Figure 18. Very little variation occurs in the upper reaches of the estuary (45 km & 60 km); the temperatures remain around 23 °C and the small fluctuations are ascribed to diurnal heating. At 20 km upstream of the mouth, the effects of tidal circulation are evident in that over the spring tides the amplitude of thermal variation is slightly larger than over neap tide (20 to 22 March). The temperatures vary mainly between 22 °C and 23 °C at this position over the whole simulation period. In contrast, the positions nearer the mouth (0,5 km, 5 km and 10 km) show substantial variations in temperature over the eighteen day simulation. The strong variations at the start of the simulation are ascribable primarily to the initial mixing between cold seawater and warmer estuarine water, while the overall increasing trend is ascribed to the gradual warming of the seawater as the 'black tide' persists. These thermal variations are considered reasonably representative of the likely sequence of events in the estuary in March 1994, particularly in view of the limited data available for model initiation and validation over this period.

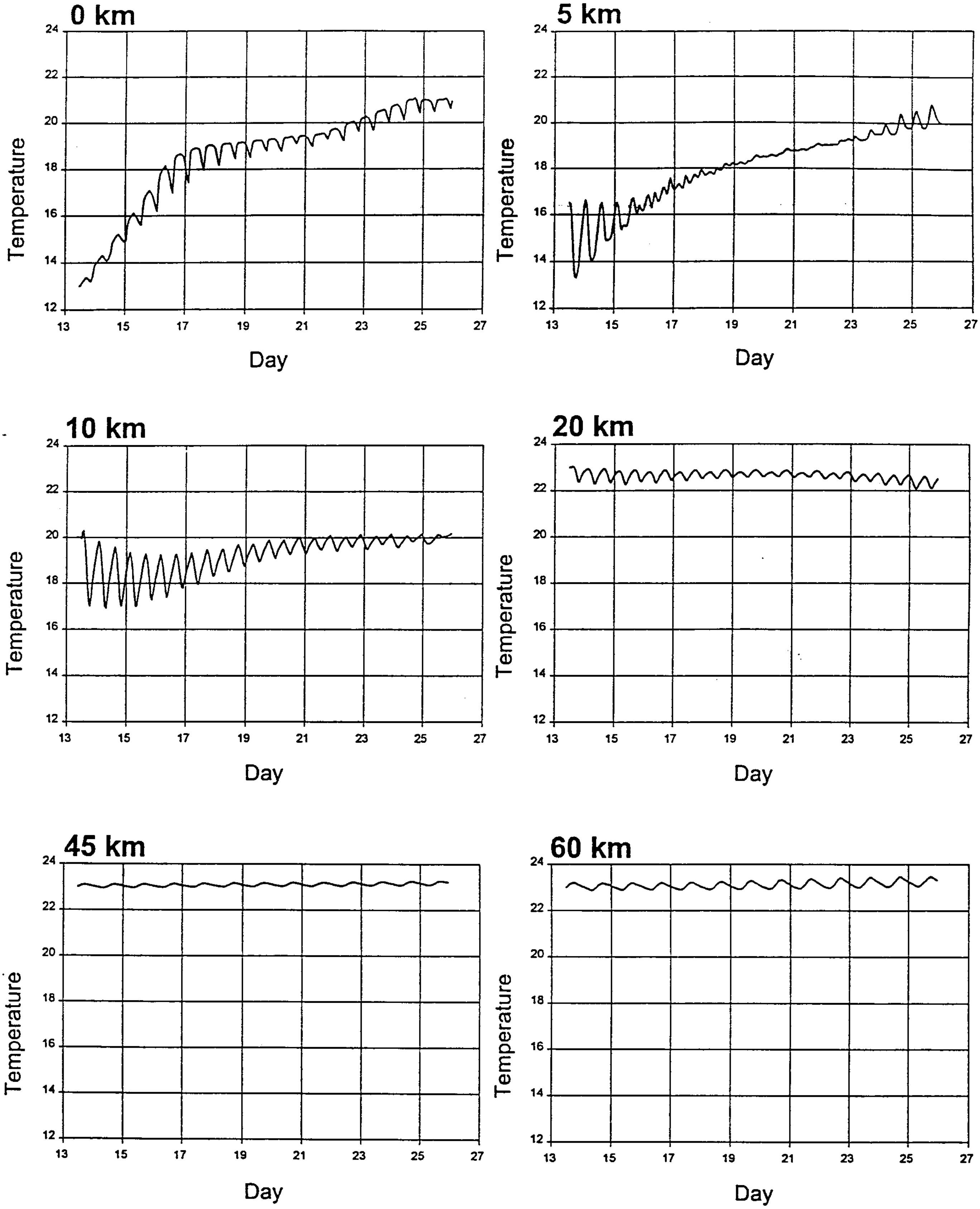

Figure 18. **Simulated temperatures (°C) at six positions in the Great Berg Estuary from 13 to 31 March 1994 (low oxygen event)**

Dissolved Oxygen and BOD

The upstream boundary condition for dissolved oxygen was set at 5 $mg.l^{-1}$ and the BOD of the inflowing river water was constant at 0,5 $mg.l^{-1}$ throughout the simulation. In contrast, the downstream boundary conditions for DO and BOD had to reflect the changing oxygen levels and organic content of the sea as the 'black tide' progressed and so varied substantially over the simulation period as listed in Table 6. The initial conditions for dissolved oxygen were set at 5 $mg.l^{-1}$ throughout the estuary and those for BOD varied linearly from 1 $mg.l^{-1}$ at the mouth to 0,5 $mg.l^{-1}$ at the head of the estuary. As in the simulation of temperatures in the estuary, the dissolved oxygen simulations (including BOD) commenced at 12:00 on 13 March 1994 and ended on 31 March 1994.

Table 6. The dissolved oxygen concentrations and BOD levels assumed to have occurred in the sea at the mouth of the Great Berg Estuary during the 'black tide' of March 1994 under different Mike 11 simulations

DATE	SIMULATION 1		SIMULATION 2		SIMULATION 3	
	DO ($mg.l^{-1}$)	BOD ($mg.l^{-1}$)	DO ($mg.l^{-1}$)	BOD ($mg.l^{-1}$)	DO ($mg.l^{-1}$)	BOD ($mg.l^{-1}$)
13	5	1	5	1	5	1
15	-	10	-	10	-	10
16	1	18	1,5	15	1	18
17	0	22	0,5	20	0,5	22
18	0	25	0	25	0	25
21	0	19	0	25	0	20
24	3	10	1,5	19	2	15
28	5	3	5	10	5	5
31	5	1	5	1	5	1

Three simulation scenarios were investigated (Table 6), namely: simulation 1 which is considered the most representative of reality, simulation 2 in which the decline and later increase in dissolved oxygen values in the sea is slower and the biological oxygen demand remains higher for longer than in simulation 1, and simulation 3 which exhibits almost the same dissolved oxygen concentrations in the seawater as in simulation 1 (a slightly slower decline and later rise), but which has a higher biological demand from 19 to 31 March 94. The results of these simulations are depicted in Figure 19 for four positions in the Great Berg Estuary.

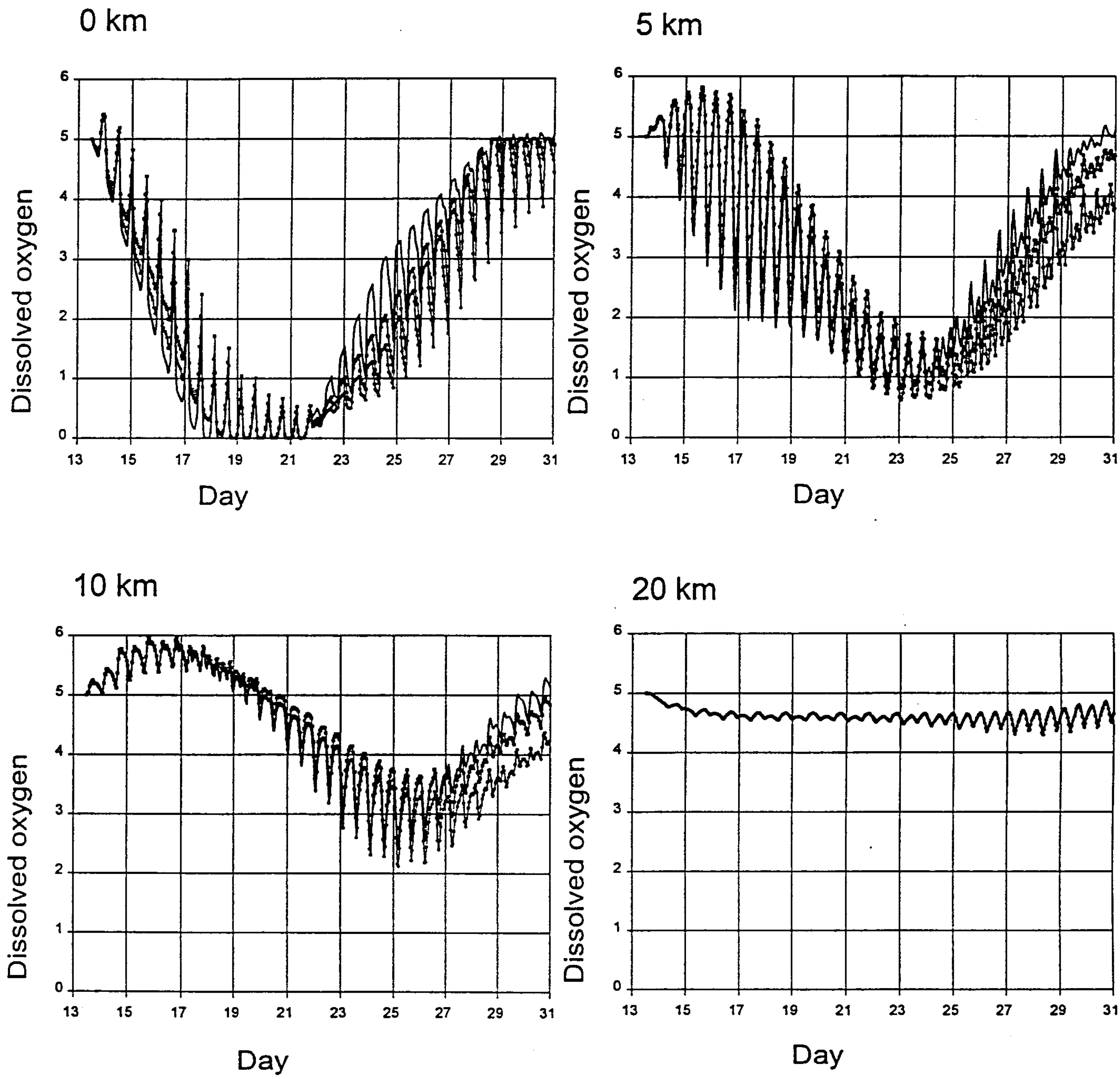

Figure 19. **Simulated dissolved oxygen concentrations (mg.l^{-1}) at six positions in the Great Berg Estuary from 13 to 31 March 1994 (low oxygen event) under Simulation 1 (straight line), Simulation 2 (squares) and Simulation 3 (dots)**

The most rapid decline in oxygen values near the mouth occurs under simulation 1, while the slowest decline occurs under simulation 2. These differences occur because de-oxygenated seawater is considered to enter the estuary on 17 March under simulation 1, while the seawater still has a DO level of 0,5 $mg.l^{-1}$ under simulations 2 and 3. Dissolved oxygen levels do decline more rapidly under simulation 2 than under simulation 3, because the concentrations in the sea are slightly lower and the biological oxygen demand higher prior to 18 March. By the 18 March the DO levels have decreased below 2 $mg.l^{-1}$ under all three simulation scenarios and between 19 and 21 March the differences between the three simulations are marginal. However, from 22 to 28 March the DO levels in the estuary increase more rapidly under simulation 1 than under the other two simulations, because the DO of the sea is considered to recover more quickly and the BOD levels are lower. The slowest recovery is exhibited under simulation 2, because this has the highest biological oxygen demand and the low oxygen levels in the sea are considered to persist for longer. Under simulation 1, DO levels remain less than 2 $mg.l^{-1}$ for six days, whereas under simulations 2 and 3 these low levels are maintained for seven days. Under simulation 2, full recovery has not occurred near the mouth by the end of March 1994, unlike simulations 1 and 3.

At distances of 5 km and 10 km from the mouth, the most rapid decline is again exhibited under simulation 1 and the least rapid under simulation 2. Similarly, the most rapid recovery occurs under simulation 1 and the slowest under simulation 2. In contrast to the mouth region, where the lowest DO levels (less than 0,5 $mg.l^{-1}$) occurred on 21 March, the lowest levels occurred on 24 March at 5 km from the mouth (less than 1,8 $mg.l^{-1}$) and on 25 March at 10 km from the mouth (less than 3,7 $mg.l^{-1}$). It can be seen that in addition to a shift in the phase of the event further upstream in the estuary, the severity declined with distance. At 20 km from the mouth very little effect was discernible, although DO levels declined from 5 $mg.l^{-1}$ at the start of the simulations to around 4,6 $mg.l^{-1}$ in the last week of March. As a DO concentration of 2 $mg.l^{-1}$ is considered critical to the respiration of biota, it is relevant to note that at 5 km from the mouth, the DO concentrations were below this level for 4 days under simulation 1 and 4,5 days under simulations 2 and 3. This level of oxygenation is insufficient for most non-migratory organisms to maintain life. In contrast, at 10 km from the mouth, the DO levels never dropped below 2 $mg.l^{-1}$, although they decreased below 4 $mg.l^{-1}$ for about 4 days. This would have caused stress to biota and perhaps the death of weaker animals, but is not life threatening in itself.

The final investigation was into the possible effects and extent of the intrusion of low oxygen water into the Great Berg Estuary if the worst of the black tide had occurred at spring tide rather than at neap tide. The tidal time series at the mouth was shifted by seven days so that a spring tide occurred on 21 March rather than a neap tide. All other parameters and the initial and boundary conditions were maintained as per simulation 1. A comparison of the dissolved oxygen levels under the two simulations is presented for four positions in the Great Berg Estuary in Figure 20.

Near the mouth, DO levels decline more rapidly under the stronger influx of seawater at spring tide and remain below the 2 $mg.l^{-1}$ level for seven days compared with six days under the neap tide scenario. However, the recovery under the spring tide scenario is generally slower than under the neap tide scenario. At 5 km from the mouth under the spring tide scenario, the minimum DO levels exhibited are less than under the neap tide scenario, but do not remain below 2 $mg.l^{-1}$ on any particular day. Instead, the DO concentrations increase above this level every flood tide. However, DO levels decrease below 3 $mg.l^{-1}$ and remain this low for about 11,5 days. At 10 km from the mouth, the DO concentrations decrease further under the neap tide scenario than under the spring tide scenario, but also recover more quickly owing to the enhanced tidal circulation in the last week of March (spring tide). Under the spring tide scenario, DO levels remain at about 4 $mg.l^{-1}$ for approximately 8,5 days. No significant differences occur at a distance of 20 km from the mouth. Thus the 'black tide' event is less severe under the spring tide scenario, but persists for longer. This is most noticeable at a distance of 5 km from the mouth. The extent of intrusion of the low oxygen water is considered to be less than 10 km under both the neap tide and spring tide scenarios.

4.1.7 Summary

Calibrations of the Mike 11 water quality module were undertaken for both the high flow winter and the low flow summer conditions on the Great Berg Estuary using data collected in September 1989, January/February 1990 and March 1990. The winter calibration was subsequently tested against data collected in August 1995 and the summer calibration against data collected in February 1996. Reasonable agreement between simulated and measured temperatures and DO levels was achieved, although the range of variation in the lower, more saline reaches of the estuary was difficult to simulate accurately. In all cases, except that of August 1995 when a freshette entered the estuary, the simulated variations were more limited than the measured variations (that is during situations of low freshwater flows and more saline influence). This was primarily ascribed to the paucity of the seaward boundary data. However, it may also be a result of the de-coupling of the transport-dispersion and water quality modules of Mike 11 which means that the effect of salinity on oxygen solubility is ignored. The good agreement obtained between measured and simulated temperatures and DO levels in August 1995 when the marine influence was more limited (less saline influence) appears to bear this out.

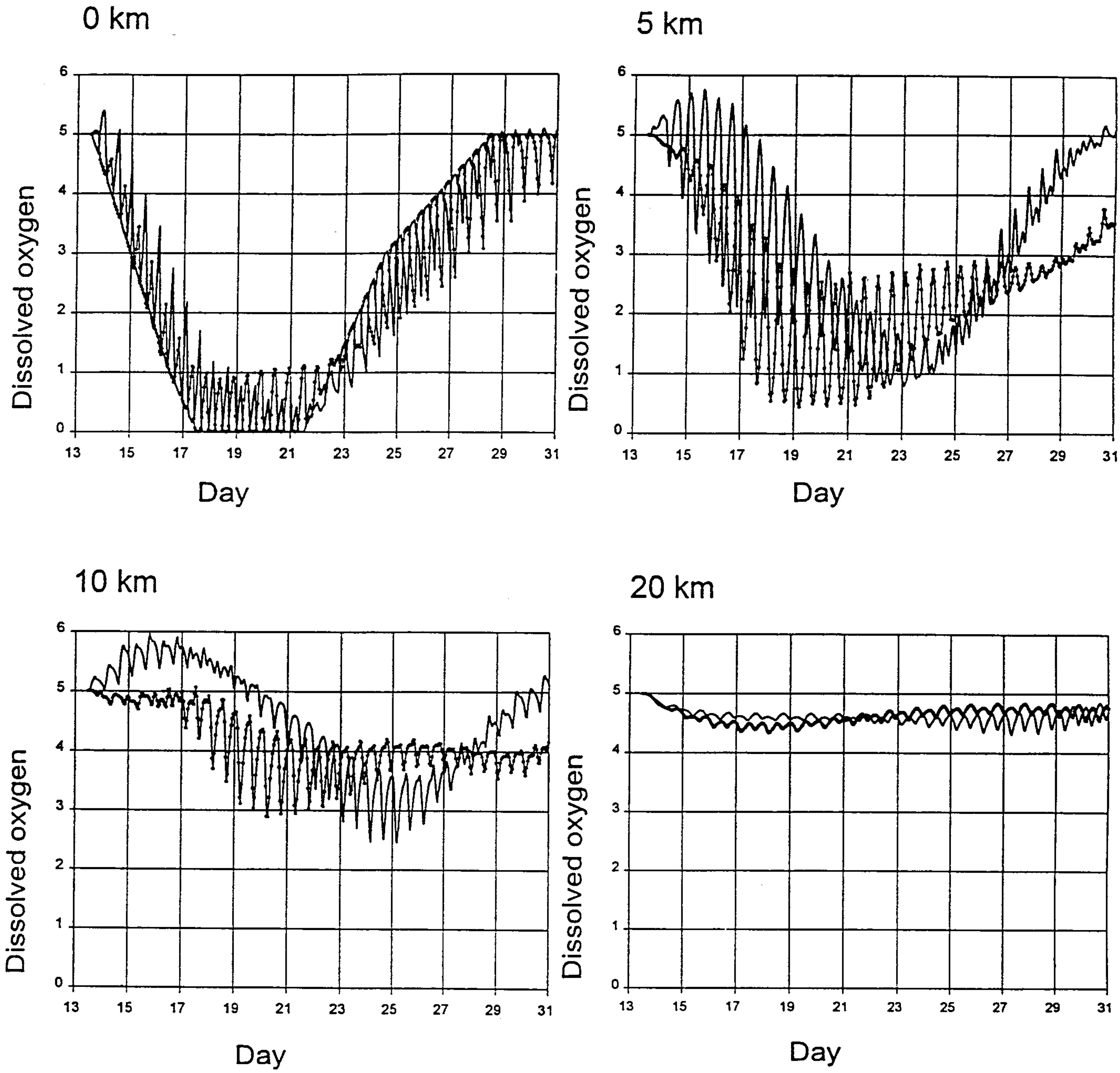

Figure 20. **Simulated dissolved oxygen concentrations (mg.l^{-1}) at six positions in the Great Berg Estuary from 13 to 31 March 1994 (low oxygen event) under Simulation 1 - neap tide (straight line) and under a spring tide simulation (dots)**

A further feature of the Mike 11 module which became evident during the calibration and testing process is that only inflowing water is assigned an associated biological demand. Once this is depleted, no additional BOD is assumed to enter the water body except by means of waste discharges. This is a realistic assumption in a highly urbanised environment, where the natural biological oxygen demand of the estuarine system is often orders of magnitude smaller than that induced by the introduction of organic waste to the system by humans. However, this assumption is certainly not true in the case of the Great Berg Estuary which receives little anthropogenic waste material (apart from fish factory effluent in the lower tidal reaches) yet meanders through a highly productive floodplain. This inadequacy in the model formulation for relatively natural systems was overcome by implementing the sediment oxygen demand component of the model (Mike 11 water quality Level 2). The sediment oxygen demand is defined as the oxygen consumption/production of the sediment within the estuary as well as the natural biological oxygen demand of the estuarine water. This parameter therefore represents the net oxygen demand within the estuary apart from the requirements for biotic respiration. This proved an effective solution in the simulation of the water quality of the relatively natural Great Berg Estuary.

A noticeable feature of the resulting DO simulations included the significantly lower dissolved oxygen concentrations in the upper and upper-middle reaches of the Great Berg Estuary in summer than in winter. This is ascribed to the warmer water temperatures, the less vigorous summer circulation and the lower DO levels of the inflowing river water. In initial studies, the first two factors were viewed as the primary causes, when the existence of a plug of typically estuarine water, which remained resident in these reaches of the estuary over the summer months and exhibited relatively low DO levels, was indicated (Taljaard & Slinger 1992, Slinger & Taljaard 1994),. This study has revealed that the character of the inflowing water (the upstream boundary condition) probably exerts the most significant influence in the formation of this feature. Our understanding of the water quality in the estuary and its relation to the circulation dynamics of the system has thus improved considerably during this study. While we cannot simulate temperatures and DO levels accurately, we can simulate envelopes of variation provided that the flow rate and the character of the inflowing water are known. Thus, a strong indication from the implementation of the model on the first case study is the need for water quality monitoring, particularly at the system boundaries (sea and river).

A further indication of the need for appropriate measurements and monitoring of boundary conditions arose from the analysis of nutrient data. While nutrient distributions in winter can be predicted with a fair degree of confidence from salinity measurements in the estuary and knowledge of concentrations at the boundaries, this is not true in the summer situation.

Moreover, little conceptual understanding exists of the bio-geochemical processes affecting nutrient concentrations, and this demonstrates the necessity for further investigation and the associated development of improved predictive capabilities. In the long term, monitoring and modelling in combination will enable researchers to judge the consequences of alterations in flow (quantity and quality) to the highly productive Great Berg ecosystem.

An interesting exploration of the utility of the Mike 11 water quality module was undertaken in its application to the 'black tide' event which occurred in St Helena Bay in March 1994 and caused massive mortalities of marine and estuarine biota. Despite very limited data, a model implementation was undertaken to explore the sequence of events and the likely DO levels in the water column as first hypoxic and then anoxic water entered the estuary on successive flood tides. The progressive decline in oxygen levels in the lower reaches was simulated as was the slow recovery from this event. Owing to considerable uncertainty in the conditions in the marine environment (the downstream boundary) at the time, a number of simulations were undertaken. The time period during which the lower estuary (up to 5 km from the mouth) experienced DO levels below 2 $mg.l^{-1}$ ranged from 4 to 6 days. A further feature investigated was the longitudinal extent of the intrusion of oxygen-depleted water. Dissolved oxygen levels below 4 $mg.l^{-1}$ appeared to extend 9 to 10 km up the estuary (to the Sishen-Saldanha railway bridge), which is the natural limit of spring flood tidal intrusion in summer. Although the height of the 'black tide' event occurred over neap tide, the oxygen-depleted water extended this distance upstream because the event persisted over a number of high tides and so intruded progressively up the estuary. Although the simulated salinities in the lower reaches are slightly lower than those measured in the system on 21 March 1994, it is extremely doubtful that the effects of the black tide extended further upstream than the Sishen-Saldahna railway bridge. Thus the Mike 11 modelling system proved useful in simulating the likely changes in water quality parameters which occurred in the Great Berg Estuary over the 'black tide' event. These results are considered extremely useful in the study of the recovery of the benthos of the system from this event (Bickerton pers. comm.) and have assisted in relating the water column changes to the observed effects on biota.

4.2 Implementation of Mike 11 on Second Case Study, Swartkops Estuary

4.2.1 Hydrodynamics

In contrast to the situation for the Great Berg Estuary, where the hydrodynamic module of the Mike 11 modelling system had been set-up and calibrated for a previous study (CSIR 1993), this was not the case for the Swartkops Estuary (Figure 21). However, a one-dimensional model had been applied to the system previously (Huizinga 1985) and the same schematisation of the 14,5 km long estuary could be used in the Mike 11 application. The cross-sectional data from surveys undertaken in 1973 and 1986 were used in defining the channel configuration (CSIR 1984).

The water level variations measured in the estuary on 6 and 7 March 1984 (CSIR 1984) were used in the calibration of the hydrodynamic module. The downstream boundary condition comprised the water levels recorded at Port Elizabeth harbour (that is the tidal variation in the sea off the mouth of the Swartkops Estuary) over the time period from 30 September to 10 October 1993. The upstream boundary condition used in the calibration was the river inflow at Perseverance, which was known to be minimal over this time period (Ms U Scharler pers. comm.) and was set at 0,1 $m^3.s^{-1}$. In view of the fact that the cross-sectional data and the recorded water levels (Figure 22) were not measured at the same time, nor was the hydrodynamic simulation conducted over either measurement period, only reasonable concurrence rather than stringent agreement can be demanded between recorded and simulated water levels for this calibration.

With the Chezy bottom roughness co-efficient for the Swartkops Estuary set at 40 $m^{1/2}.s^{-1}$ in the lower reaches (from the mouth to Brickfields) and 50 $m^{1/2}.s^{-1}$ further upstream, the behavioural agreement between recorded and simulated water levels is considered reasonable (Figures 22 & 23) and the hydrodynamic module of Mike 11 is deemed calibrated for the Swartkops Estuary.

4.2.2 Salinity

The transport-dispersion module of the Mike 11 modelling system was calibrated on the Swartkops Estuary using salinity data collected in June and July 1994 (Ms U Scharler pers. comm.). These data were selected for use in the calibration because the measurements were taken at neap tides under low river flow conditions. The initial conditions in the estuary were set at the measured salinities on 26 June 1994. The river inflow (zero salinity) over the simulation period varied from 0,3 $m^3.s^{-1}$ to 0,6 $m^3.s^{-1}$, while the salinity of the seawater entering the system at the mouth was set at 35 ppt.

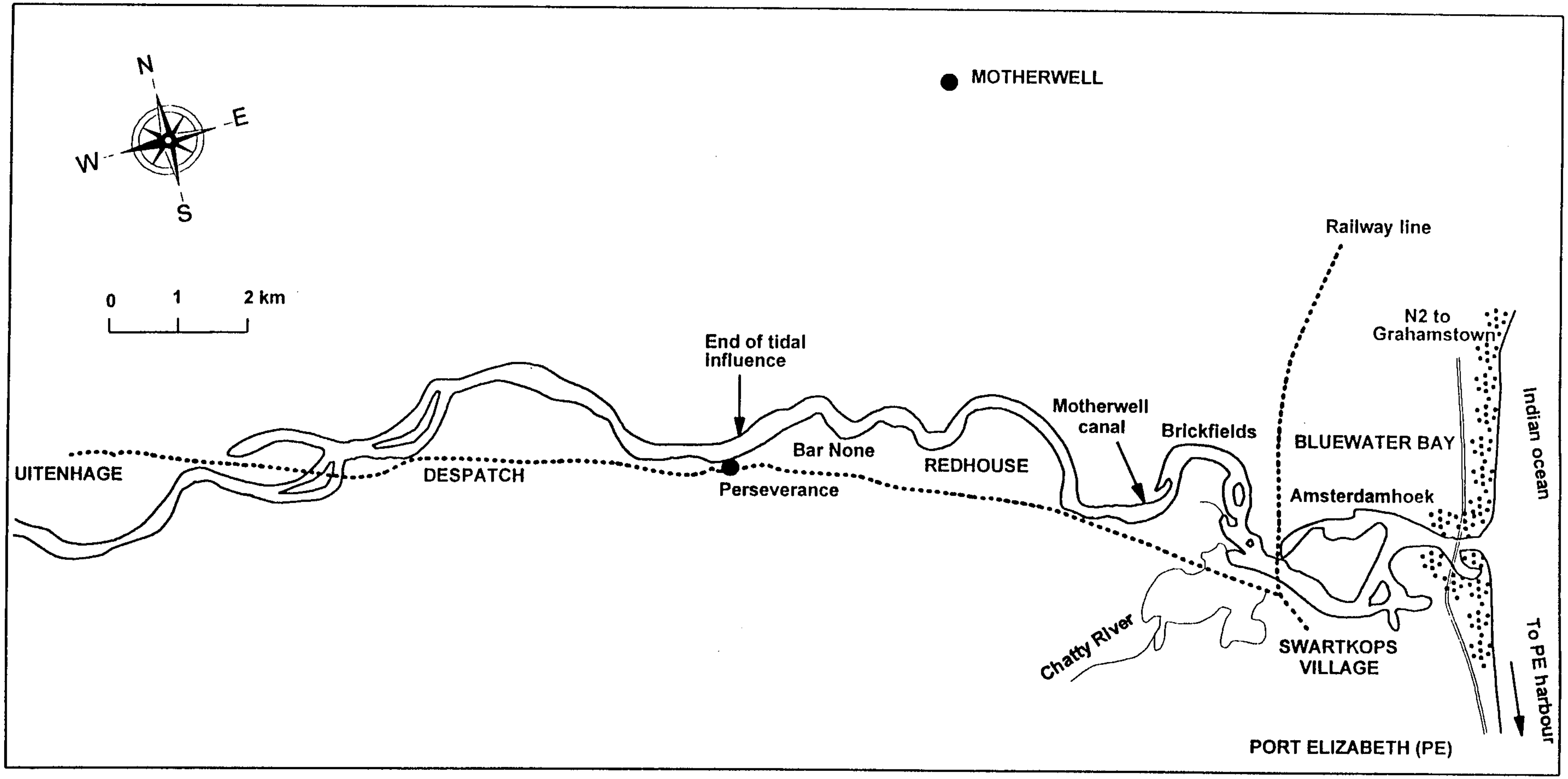

Figure 21. **Map of the Swartkops River Estuary**

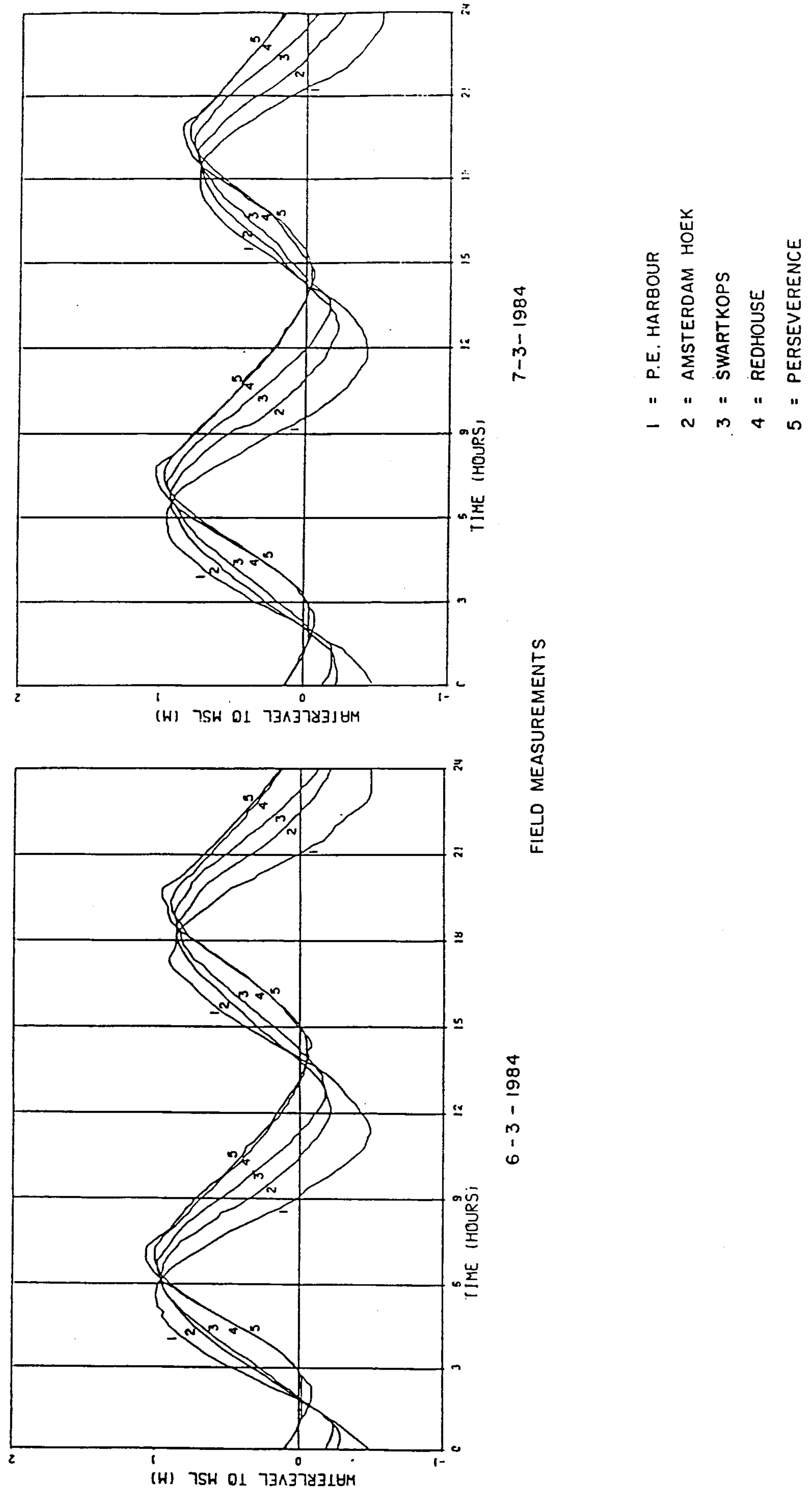

Figure 22. **Recorded variation in water level (m to MSL) in the Swartkops Estuary on 6 and 7 March 1984 (1 = Port Elizabeth harbour; 2 = Amsterdamhoek; 3 = Swartkops village; 4 = Redhouse; 5 = Perseverance)**

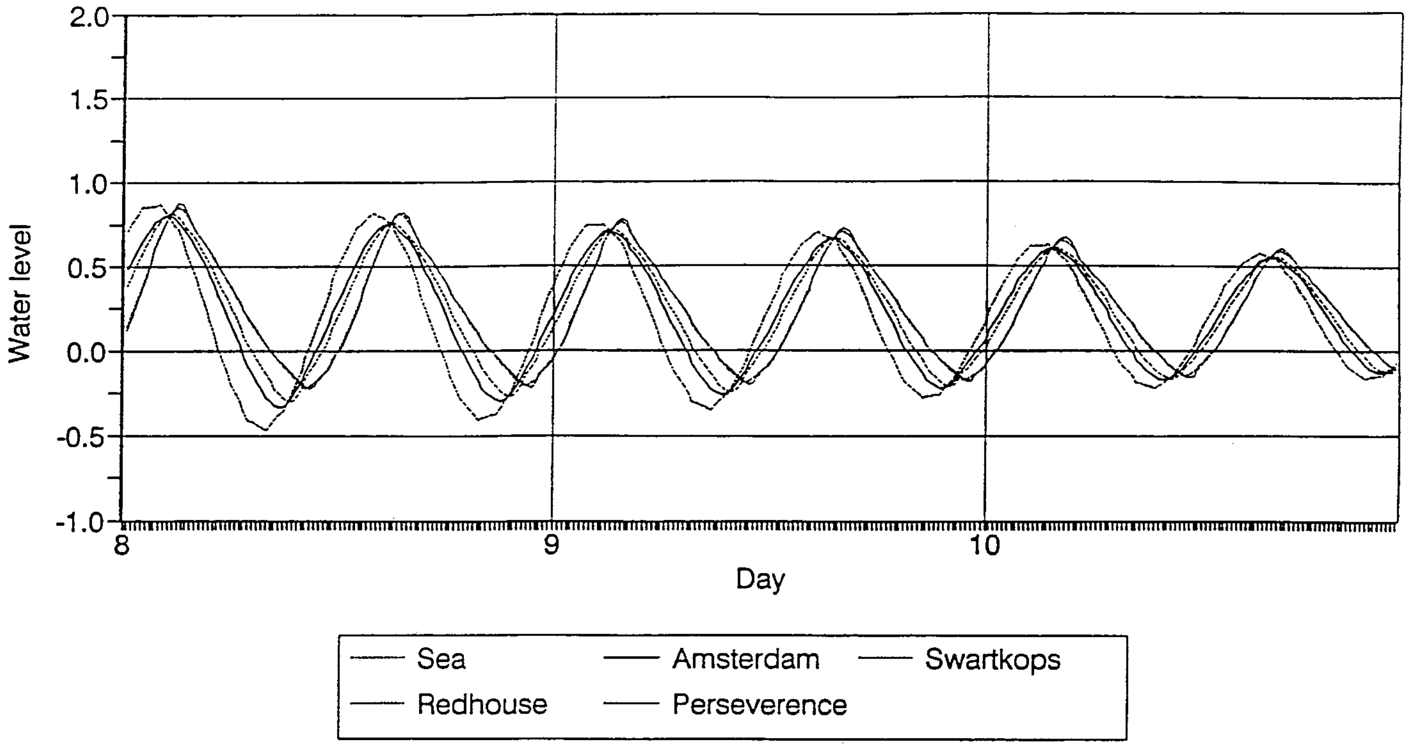

Figure 23. **Simulated variations in water level (m to MSL) in the Swartkops Estuary using the Mike 11 hydrodynamic module**

With a dispersion co-efficient of 30 $m^2.s^{-1}$, a comparison between the simulated salinities along the length of the estuary and the values measured on 29 July 1994 are presented in Figure 24. The agreement between measured and simulated values is at best reasonable, but is accepted as sufficient for calibration purposes owing to the following limitations in the data:

- the river inflow during the calibration period is not accurately known and may have varied somewhat owing to variations in waster water discharges further upstream;
- rainfall may have influenced salinities along the estuary, but the extent of influence is also unknown.

Because the Swartkops Estuary lies in a biogeographic region which experiences almost uniform rainfall throughout the year, it is not appropriate to consider seasonal differences in water inflow to the system. Instead, conditions of high and low river flow, which can occur at any season, were selected as providing an indication of the variability of the hydrodynamics and salinities in the system. Simulations under river inflow conditions of 0,5 $m^3.s^{-1}$ and 5 $m^3.s^{-1}$ over a full spring-neap-spring tidal cycle (typical of the Port Elizabeth region) were undertaken. The results are presented in Figure 25 in the form of longitudinal salinity graphs at spring high, spring low, neap high and neap low tides.

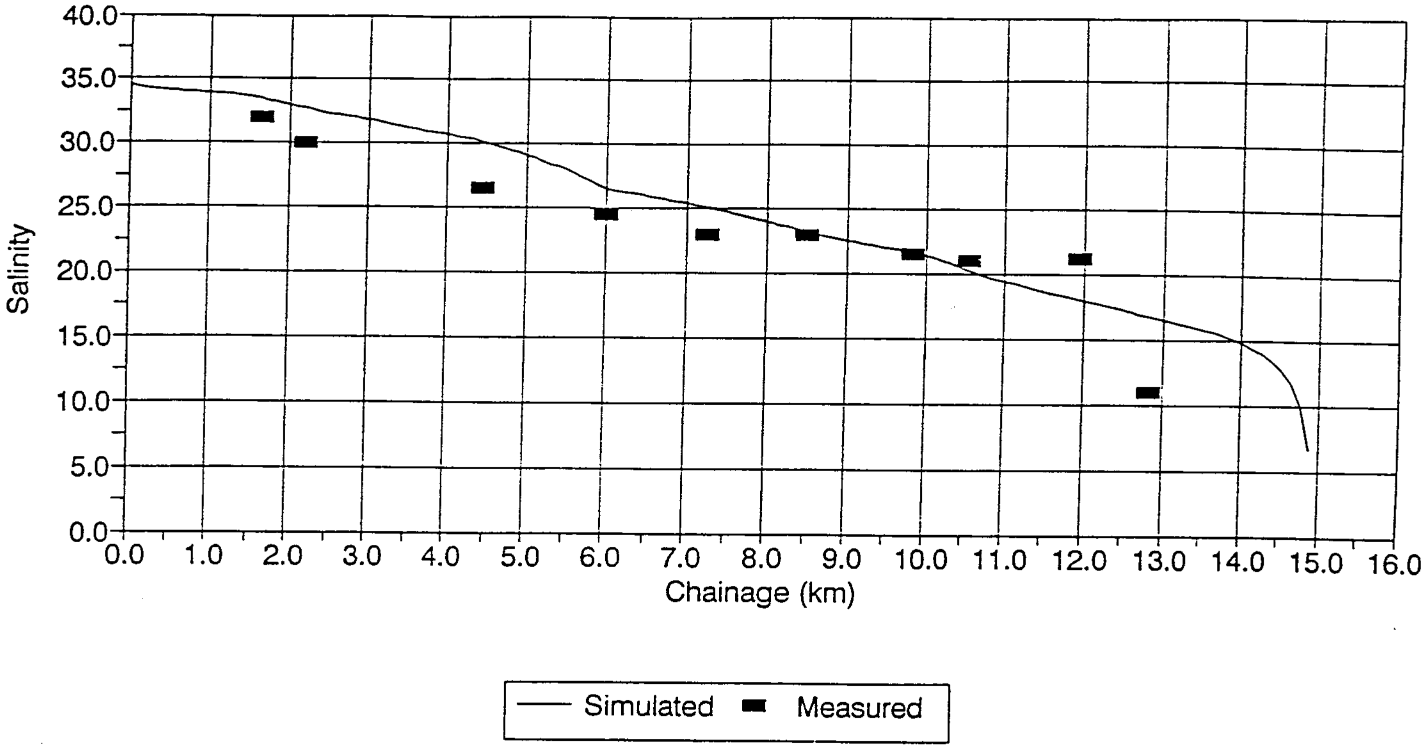

Figure 24. **Comparison between simulated salinities (ppt) in the Swartkops Estuary and those measured on 29 July 1994**

Under low river inflows, higher salinities extend further upstream in the estuary (16 - 25 ppt at 10 km from the mouth) and the envelope of variation is smaller than under high river inflows. When the river inflow is high, salinities are less than 1 ppt at 10 km from the mouth and only at spring high tide do salinities of 15 ppt extend as far upstream as 6 km from the mouth. The envelope of variation is substantial, especially at about 4 km from the mouth, where the difference between salinities at spring low tide and spring high tide is about 28 ppt. This contrasts strongly with the variation in salinities under low river inflows, which is greatest at a distance of 6 km from the mouth, when the difference between salinities at spring low tide and spring high tide is about 10 ppt. These simulations clearly indicate that more uniform salinity conditions prevail in the Swartkops Estuary under low river flows than under high flows. The available salinity data confirm this finding.

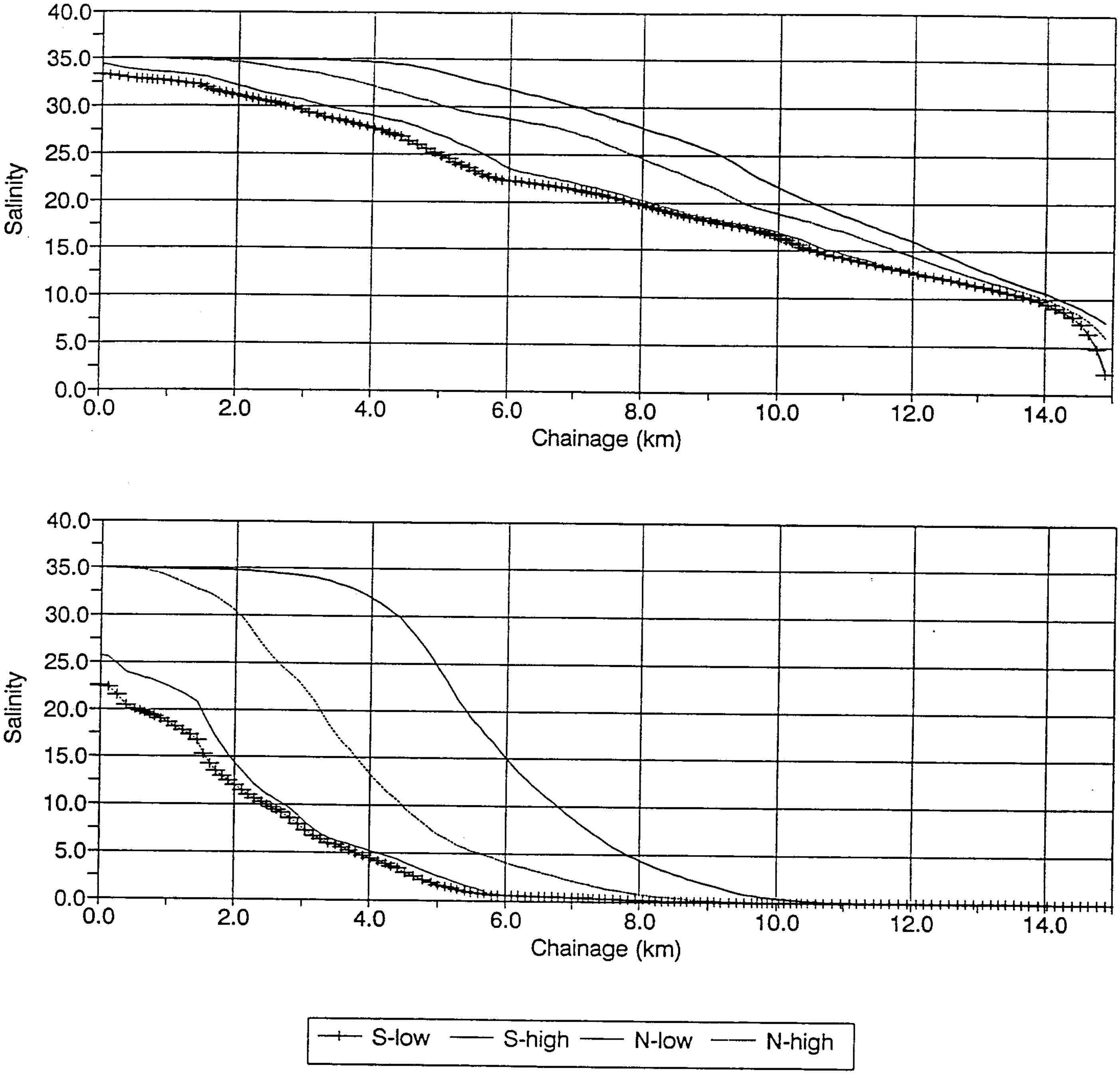

Figure 25. **Simulated longitudinal salinities (ppt) in the Swartkops Estuary at spring low (S-low), spring high (S-high) , neap low (N-low) and neap high (N-high) tides under low flow conditions (top) and high flow conditions (bottom)**

4.2.3 Temperature and dissolved oxygen (including BOD)

The available data for the Swartkops Estuary were analysed in terms of their applicability in calibrating and implementing the water quality module of Mike 11. APPENDIX A contains the salinity, temperature and dissolved oxygen measurements taken in the surface and bottom waters at spring tides along the channel of the estuary and in the river and sea on five occasions.

From these data and the knowledge gained in the application of Mike 11 to the Great Berg Estuary, parameter values and boundary conditions suitable for implementing the water quality module on the Swartkops Estuary could be deduced. Two river flow scenarios were deemed meaningful for consideration, a low flow (0,5 $m^3.s^{-1}$) and a high flow (5 $m^3.s^{-1}$) condition, and different thermal situations were also included. Under low river flows, the thermal situations represent the seasonal differences in the temperature of the river and sea. In summer, both water masses are about 4 °C to 5 °C warmer than in winter. As these differences in temperature can influence the dissolved oxygen concentrations in the system, it was considered important to model the 'cold' and 'warm' situations. In contrast, under high flows the river water is not resident in the river system long enough to be warmed substantially and even in summer the temperatures are likely to be lower than the usual river water temperatures. Thus, only one high flow thermal situation is considered, in which the temperatures of the sea and the river water are more or less equal.

The available data also indicate that concentrations of dissolved oxygen in the river and sea water were generally higher under cold, low flow conditions than when the water masses were warmer or flowing more strongly. Typical values for low flow, cold conditions as opposed to warmer or high flow conditions were derived from the data and used as upstream and downstream boundary conditions for the different simulation scenarios (Table 7).

In the absence of measured data on the biological oxygen demand of the estuarine, riverine or marine water and the respiration or photosynthetic rates in the Swartkops Estuary, the values pertaining to the summer calibration for the Great Berg Estuary were selected as most likely to concur with conditions in the Swartkops Estuary. Additionally, a sediment oxygen demand of 0,2 g $O_2.m^{-2}.day^{-1}$ was deemed appropriate over the whole Swartkops Estuary. The boundary conditions used in the simulation of the water quality of the Swartkops Estuary are listed in Table 7. The initial conditions for temperature, DO and BOD comprised a linear variation between the downstream (sea) and upstream (river) boundary conditions.

Table 7. **Boundary conditions applied in the simulation of different flow and temperature scenarios for the Swartkops Estuary**

WQ MODEL COMPONENT	**SIMULATION SCENARIOS**		
	Low Flow, Warm	**Low Flow, Cold**	**High Flow**
River inflow ($m^3.s^{-1}$)	0,5	0,5	5,0
Temperature (°C): seawater (0 km) freshwater (14,9 km)	19,6 25,0	15,2 20,0	19,6 20,0
Dissolved oxygen (mg/l): seawater freshwater	7,31 7,75	8,3 9,3	7,31 7,75
Biological oxygen demand (mg/l): seawater freshwater	1,0 1,0	1,0 1,0	1,0 1,0

Temperatures

The results of the simulations of temperatures over a twenty-eight day spring-neap-spring tidal cycle are depicted in Figure 26. Under the low flow, warm scenario the temperatures vary by about 5 °C between the head and the mouth regions of the estuary throughout the simulation period. As in the salinity simulation, the greatest variability is evident at about 6 km from the mouth, as this distance marks the extent of tidal intrusion of new seawater at spring tide. The envelope of variation of salinities is not large as the greatest variation is of the order of 3 °C. The simulation results for the low flow, cold scenario are very similar apart from the fact that the envelope of variation is shifted down by about 5 °C. Again, the greatest variability (about 3 °C) is evident 6 km upstream of the mouth.

Under high river flows, temperatures are almost uniform throughout the system. The greatest variation occurs at between 2 and 4 km from the mouth. This variation amounts to only about 1 °C, indicating the confined nature of the envelope of variation in temperatures.

Dissolved oxygen (including BOD)

Under the low flow, warm scenario (Figure 27), the DO levels in the estuary initially vary from 7,3 $mg.l^{-1}$ in the sea to 7,75 $mg.l^{-1}$ at the head of the estuary. As the simulation progresses, the dissolved oxygen concentrations in the upper reaches decrease further until a value of 6,3 $mg.l^{-1}$ is attained. Thereafter the variations in DO level remain within the envelope created. Thus the greatest variation in concentrations (about 1,5 °C) occurs at a distance of 12 km from the mouth.

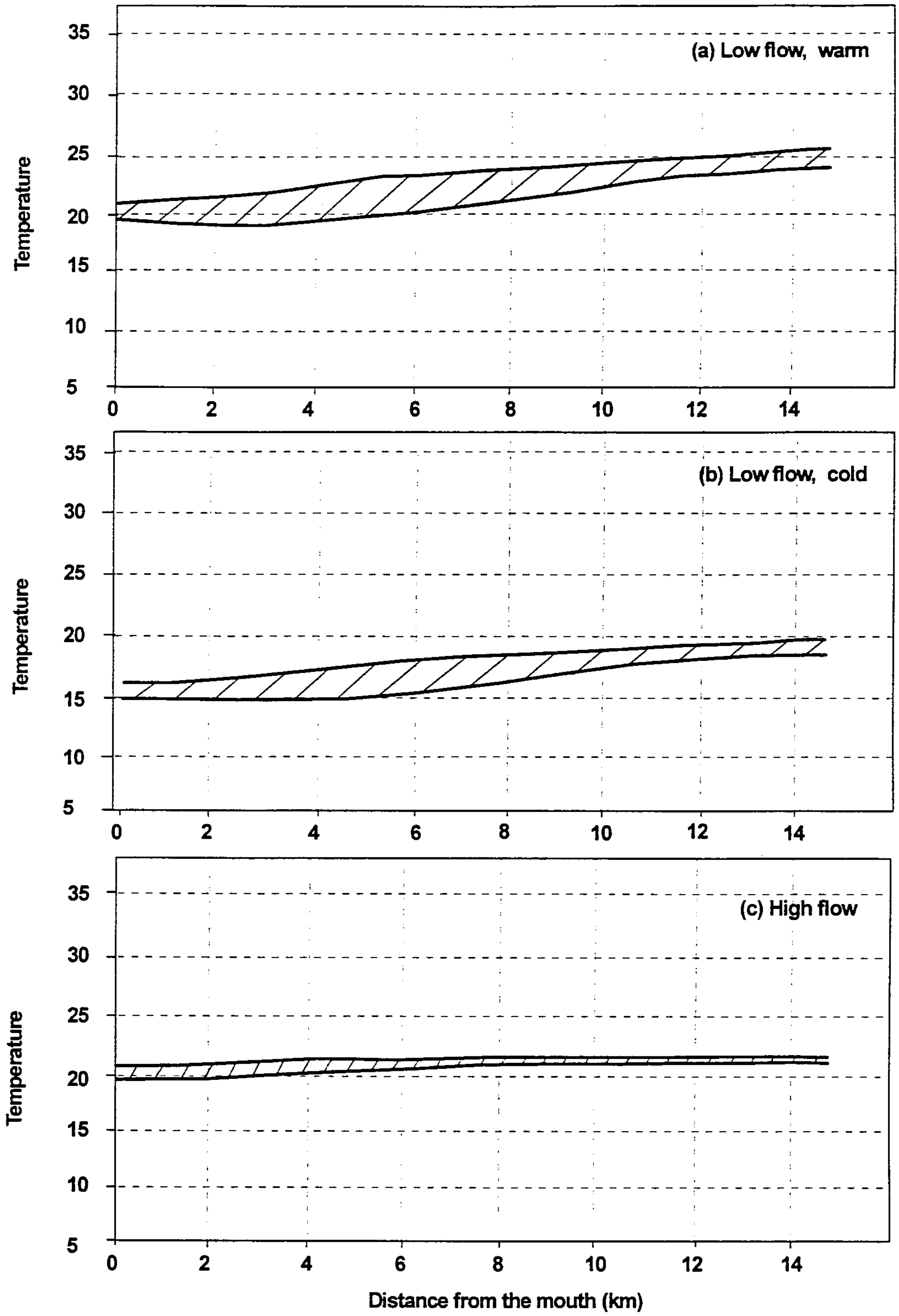

Figure 26. **The band of temperatures (°C) simulated by Mike 11 (shaded area) under (a) the low flow, warm, (b) the low flow, cold and (c) the high flow simulation scenarios over a 28 day period in the Swartkops Estuary**

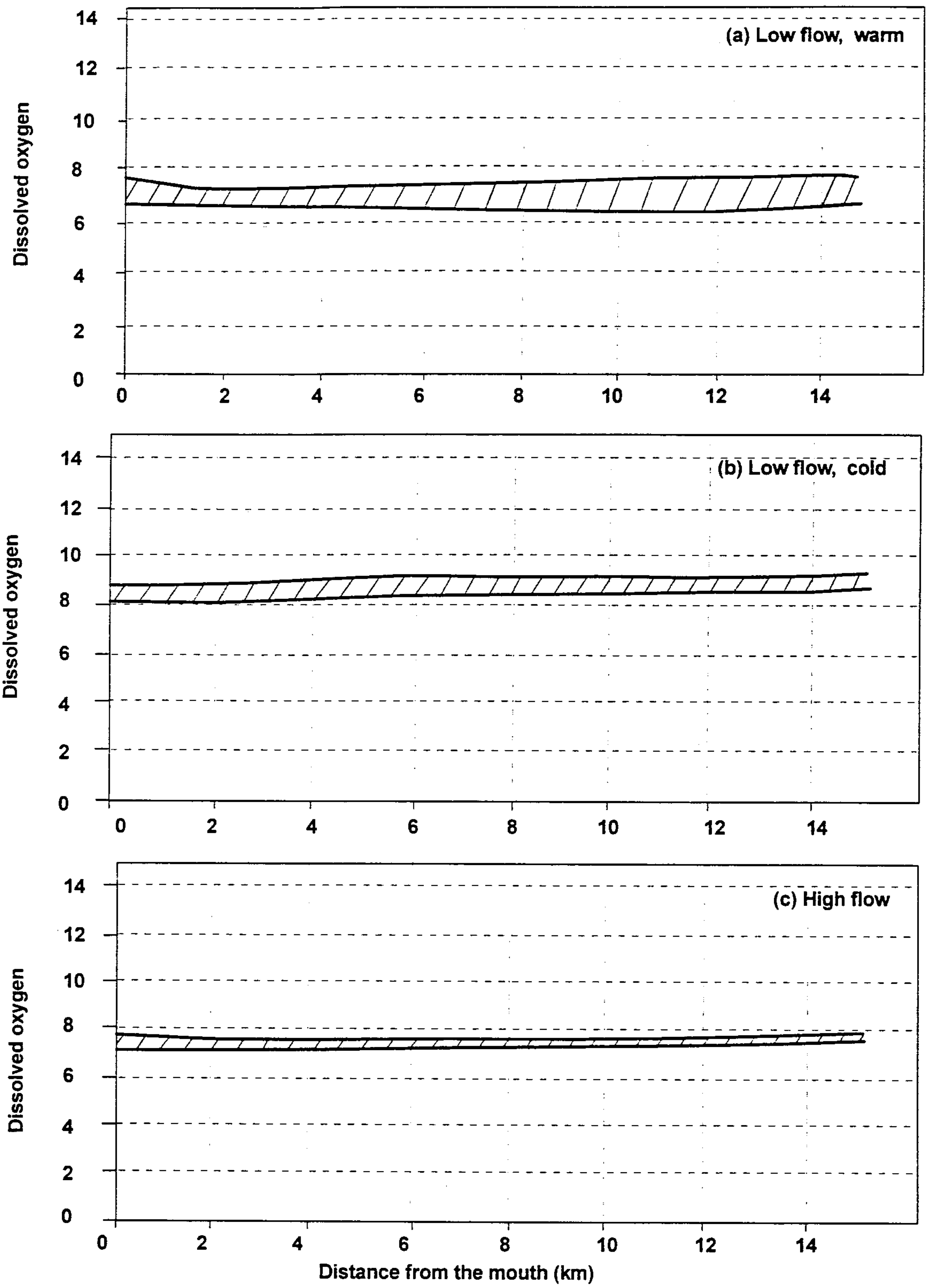

Figure 27. **The band of dissolved oxygen concentrations (mg.l^{-1}) simulated by Mike 11 (shaded area) under (a) the low flow, warm, (b) the low flow, cold and (c) the high flow simulation scenarios over a 28 day period in the Swartkops Estuary**

In contrast, under the low flow, cold scenario (Figure 27), the DO levels exhibit an almost uniform range of variation of about 1,25 °C throughout the estuary. The maximum value achieved is 9,3 $mg.l^{-1}$ (the inflowing river water) and the minimum value is 8,1 $mg.l^{1}$ in the mouth region. Under the high flow condition (Figure 27), the DO levels form a narrow band with the most variation exhibited at the mouth (about 0,7 $mg.l^{-1}$). The inert behaviour of the Swartkops Estuary in terms of oxygen dynamics accords well with available field data and the strong tidal mixing which occurs in the system. However, these simulations merely highlight features of the water quality of the Swartkops Estuary in relation to the hydrodynamics and salinity distributions. They cannot be considered validated simulations in view of the inadequacy of the available data for model testing.

4.2.4 Nutrients

The relationships between salinity and dissolved nutrient concentrations (nitrite-N, nitrate-N, total ammonia-N, reactive phosphate-P, reactive silicate-Si) were investigated for the data collected in August 1996 by the CSIR and on seven other occasions by the University of Port Elizabeth (Ms U Scharler pers. comm.). The results are presented in APPENDIX A and those for total inorganic nitrogen-N, dissolved reactive phosphate-P and dissolved reactive silicate-Si are repeated in Figures 28 to 30.

Dissolved reactive phosphate-P concentrations, and to a lesser extent dissolved reactive silicate-Si concentrations, show a strong linear relationship to salinity (average r^2 values of 0,89 and 0,82, respectively). Thus these dissolved nutrients may be treated as 'conservative' in the Swartkops Estuary. In contrast, no clear relationship was discernible for total inorganic nitrogen, nor could regions of depletion, generation or influx be distinguished. Thus a conceptual picture of the processes modifying concentrations could not be obtained from the available data. This suggests that the bio-geochemical processes in the estuary which modify concentrations e.g. nitrification, de-nitrification, need to be investigated further before the dynamics of the total inorganic nitrogen-N of the Swartkops Estuary can be modelled. The apparent decoupling between dissolved reactive phosphate-P and total inorganic nitrogen-N is an interesting feature, emphasising the need for further investigations of the nutrient dynamics of the Swartkops Estuary.

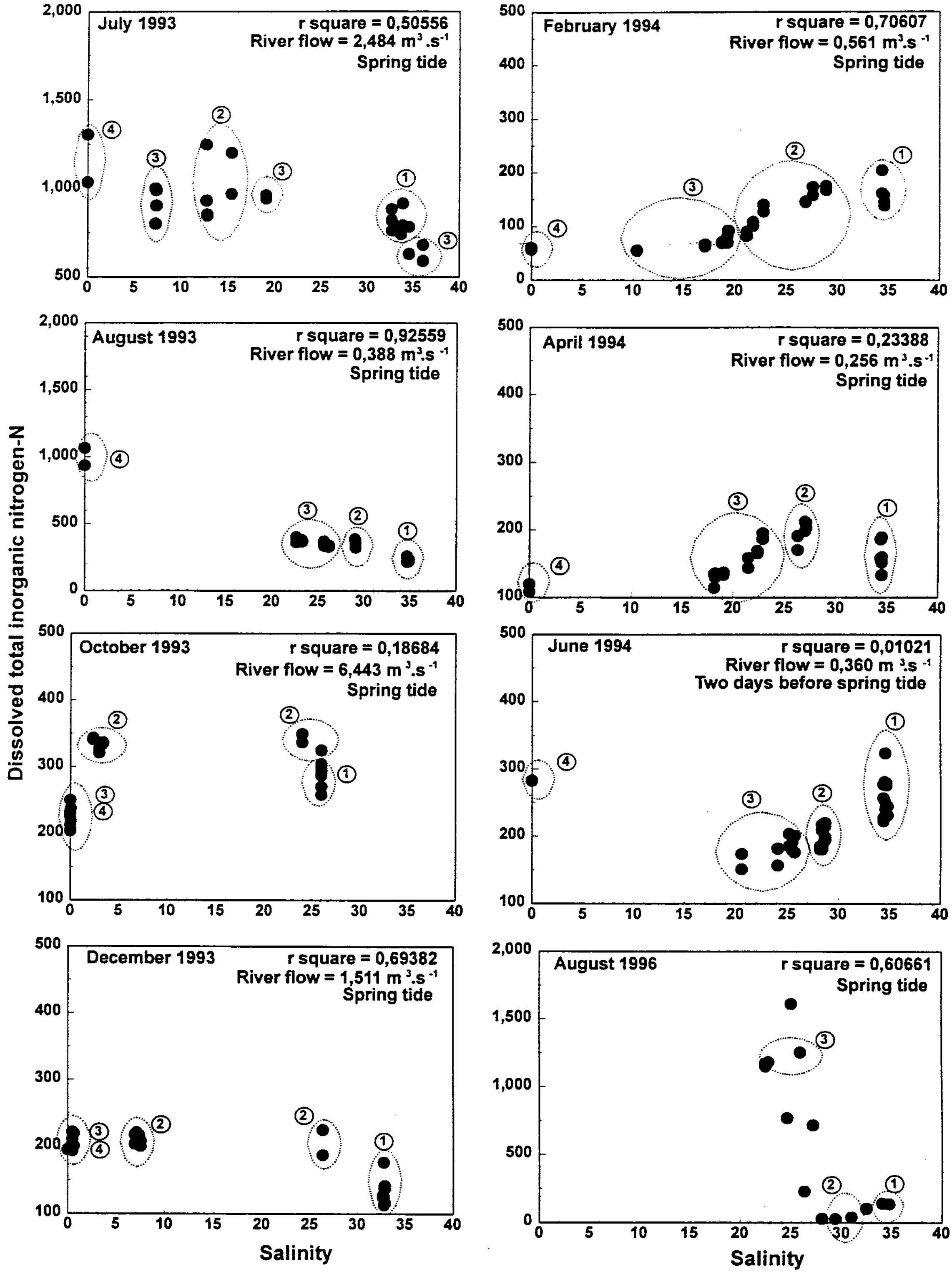

Figure 28. **The relationship between salinity (ppt) and dissolved total inorganic nitrogen-N concentrations (µg.l^{-1}) in the Swartkops Estuary (1 = Amsterdamhoek, 2 = Brickfields, 3 = Bar None and 4 = Perseverance)**

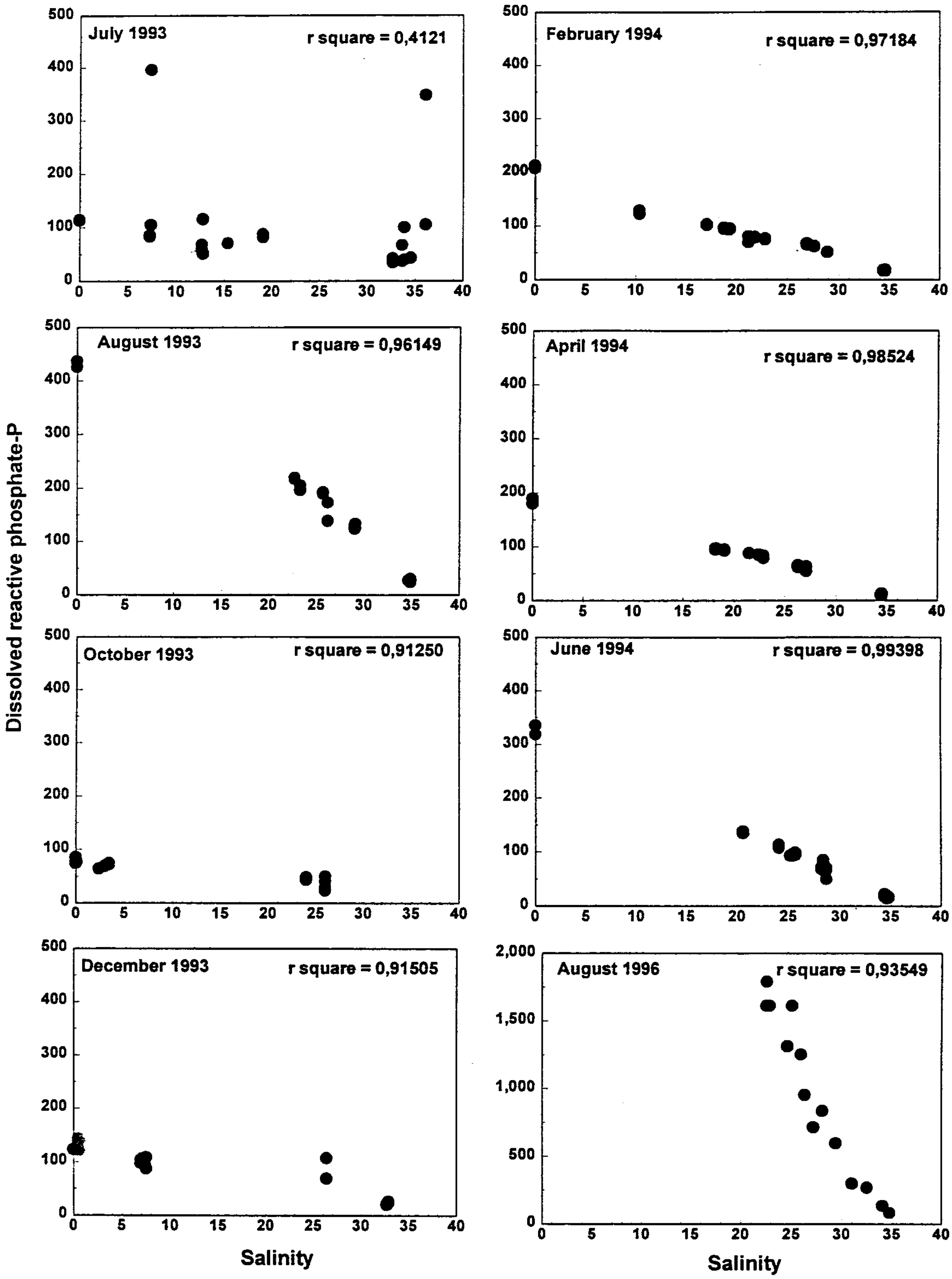

Figure 29. **The relationship between salinity (ppt) and dissolved reactive phosphate-P concentrations ($\mu g.l^{-1}$) in the Swartkops Estuary**

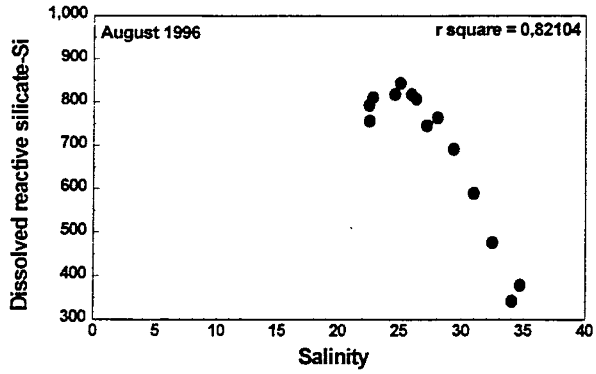

Figure 30. **The relationship between salinity (ppt) and dissolved reactive silicate-Si concentrations (µg.l^{-1}) in the Swartkops Estuary**

The concentrations of the 'conservative' nutrients in the Swartkops Estuary may be derived from salinity values by using the following formula:

$$C(x) \;=\; C_{riv} + [\,(C_{sea} - C_{riv}) / (S_{sea} - S_{riv})]\,.\,x$$

where $C(x)$ = concentration at salinity x

x = salinity

C_{riv} = concentration in the river

C_{sea} = concentration in the sea

S_{sea} = salinity of the sea

S_{riv} = salinity of the river.

4.2.5 Bacterial Contamination

Simulations of bacterial contamination and die-off in the Swartkops Estuary could not be conducted as the requisite formulae are not contained in the water quality module of the Mike 11 system, but in a separate module not in the possession of the CSIR. However, the data relating to bacterial contamination were collated and analysed in terms of scenarios and loads which could be simulated in any future modelling studies. The details of these data sources and the analyses conducted are included in APPENDIX A. The suggested simulation scenarios for future modelling of bacterial contamination effects are listed in Table 8.

Table 8. Suggested simulation scenarios for modelling bacterial contamination in the Swartkops Estuary (after MacKay 1993)

INFLOW SOURCE	FLOW ($m^3.s^{-1}$)	*E. coli* (count per 100 ml)
River (low flow)	0,5	100
River (high flow)	5,0	10 000
Motherwell canal (base flow)	0,002	1 x 10^6
Motherwell canal (rain event - 9,5 mm)	0,0015 (at outset) 0,056 (after 8 hrs) 0,113 (after 10 hrs) 0,09 (after 11 hrs) 0,72 (after 12 hrs) 0,63 (after 13 hrs) 0,39 (after 14 hrs) 0,37 (after 15 hrs) 0,35 (after 16 hrs) 0,0025 (after 23 hrs)	1 x 10^6 throughout

According to MacKay (pers. comm.), the inflow from the Motherwell Canal tends to become confined to a narrow ribbon along the eastern bank of the estuary in the region of the canal outlet. It is doubtful whether a one-dimensional modelling approach would be suitable in such a situation as the width- and depth-averaged *E. coli* numbers would be simulated, probably underestimating the level of bacterial contamination in the locality of the canal.

4.2.6 Summary

Rather than calibrating the Mike 11 model for specific measurement periods and simulating measured events, the approach adopted in the Swartkops case study was that of applying a number of representative flow rates over twenty-eight day periods (a full spring-neap-spring cycle) and evaluating the resulting variations in salinities, temperatures and dissolved oxygen levels. This approach reflects the fact that while considerable information is available on the water quality of the Swartkops Estuary, these data were not collected with model application purposes in mind and hence generally proved unsuitable for this use. Thus the simulation results provide an indication of the dynamics of the estuary rather than accurate pictures of specific measured conditions.

The Swartkops Estuary exhibits relatively uniform salinities along its length under low river flows. In contrast, under high river flows, longitudinal variations in salinity from 0 to 35 ppt occur, while variations in salinity of 28 ppt occur about four km upstream of the mouth over the spring tidal cycle. The simulated variations in temperature are strongest under low river flow conditions when warm river water enters the estuary, but are comparatively weak under high flows primarily because the inflowing river water is assumed to be comparatively colder and nearer in temperature to the seawater. The simulations of dissolved oxygen concentrations reveal that the estuary is relatively inert in terms of the assumed parameter set applied to the system. The only scenario in which some depletion of oxygen occurs is that of low river flows in summer (warm river and sea temperatures). Even then, dissolved oxygen concentrations in excess of 6,3 $mg.l^{-1}$ are maintained throughout the estuary. This is ascribed primarily to the strong tidal circulation in the system. Adequate measurements are required to determine model parameters sufficiently accurately in order to draw general conclusions or simulate measured conditions.

Interesting results were obtained from an analysis of nutrient data in the Swartkops Estuary. Both dissolved reactive phosphate-P and dissolved reactive silicate-Si were found to exhibit linear relationships to salinity to the extent that concentrations at a given salinity could be predicted based on knowledge of the concentrations of the relevant parameters in the sea and the river and the salinities of the sea and the river at the time. No clear relationship to salinity was obtained for total dissolved inorganic nitrogen-N, nor could a conceptual picture of the processes causing depletion or generation be obtained from the available data. It is considered advisable that further measurements be undertaken in this system with relation to the determination of these processes and aimed at explaining the apparent decoupling between reactive dissolved phosphate-P and total dissolved inorganic nitrogen-N.

Additionally, scenarios for future simulation of bacterial contamination in the Swartkops Estuary are suggested. No such simulations were undertaken in this study as the formulae for modelling bacterial contamination and die-off are not contained in the Mike 11 water quality module presently in the possession of the CSIR.

5. DISCUSSION AND CONCLUSIONS

5.1 The Context of Water Quality Problems

Substantial progress has been made in addressing the context of water quality problems in South African estuaries and the information and decision support requirements against which the utility and performance of Mike 11 can be measured. The literature survey and the consultations with international scientists and key South Africans have contributed to the development of an understanding of the decision support requirements for water quality management both at present and in the future. This the framework within which such evaluation can occur.

In assessing the capacity and suitability of Mike 11 for water quality modelling of South African estuaries from the literature, particular attention was paid to the applicability of a Mike 11-type modelling system to the types of water quality problems and management requirements that occur in South African estuaries. The long, narrow form of many estuaries is particularly suited to a branched one-dimensional modelling system such as Mike 11. Such models are best applied to systems with relatively stable mouth configurations e.g. permanently open systems, marine bays. However, in association with other modelling approaches such as expert systems and systems modelling, models such as Mike 11 can usefully be applied to temporarily closed systems for short time periods. Many effective one-dimensional model applications have been undertaken to date. These include implementations on the Knysna Estuary, the Keurbooms system, the Great Brak Estuary and the Kromme Estuary, amongst others. So, while not being ideally suited to the rapidly changing morphodynamic environment of many South African estuaries, Mike 11 can be used effectively to model the abiotic environment.

Other clear indications from the literature and the discussions with scientists relate to the readily discernible effects on many estuaries internationally of human activities such as waste disposal or nutrient enrichment of the inflowing river water owing to catchment land-use. In comparison, South African systems are still very natural and anthropogenically-induced effects are often difficult to distinguish from ambient variability. It is for the purposes of this distinction that the value of long term sequences of monitoring data are demonstrated as these sequences are useful in determining the envelope of natural variability, but also in indicating increasing or decreasing trends. A serious limitation to effective water quality management of estuaries is the general lack of data on estuarine responses to catchment and local activities. Prevailing scientific opinion on the type of response exhibited in South African estuaries is that water column effects are seldom evinced, but that benthic productivity increases considerably. This contrasts with the types of responses commonly observed internationally. The benthically-orientated responses of South

African systems are of interest internationally as it is only in recent years that scientists have realised that many biotic responses have been neglected by focusing on the prediction of water column responses. In fact, these effects manifest relatively quickly (short term responses), whereas over years other effects may be more significant e.g increased or reduced macrophyte growth, more persistent or extensive anoxia and hence more prolonged stress to biota (long term responses). It was strongly recommended that work on the prediction of the longer term biological responses associated with the physical and chemical changes in the estuarine environment is pursued further.

An area receiving considerable attention internationally, particularly in recent years, relates to the effects of catchment land-use practices on the water quality of downstream rivers and estuaries. Much monitoring of effects is being done, while technical developments are focused on the synthesis of information for management through the development of decision support systems. These systems integrate monitoring, models of water flow, waste loading and water quality as well as ecological responses to provide measures of system performance useful in testing management practices and supporting decision making. South Africa could benefit by adopting some of these recent technological developments.

Information on the acknowledged strengths and weaknesses of the Mike 11 modelling system was also obtained from the literature and other scientists. In brief, the numerical stability and reliability of the model is deemed its most outstanding feature. Weaknesses include an older file management structure and the inclusion of fewer alternative formulae in the water quality modules than other comparable systems.

Lastly, the relationships between human activities, the associated changes in water quality and the existing or potential water quality problems in South Africa were explored and reported. The current understanding of the context of estuarine water quality in South Africa is captured most succinctly in the final document. Herein are described matchings between anthropogenic activities and developments, existing and potential water quality problems, the highly dynamic environment of South African estuaries and the information and modelling required to support the management decision process. In particular, numerical modelling comprises one of a number of techniques supporting water quality management decision making in South Africa. Within the management framework, numerical modelling is most appropriately used in compliance testing, determining the fate of different water quality constituents in the environment and establishing the natural or ambient variability of different water quality parameters. In these applications, numerical models play the role of associating variable physical forcing and development scenarios with the abiotic responses of the estuary and of establishing the efficacy of management practices by comparing these responses directly with abiotic compliance measures or indirectly, through biotic indicators or prediction techniques, with biotic compliance measures. The numerical models explore the matching between likely and required behaviour under variable environmental conditions, so enabling more informed decisions.

It is advisable to refer to the document entitled "*Decision Support Requirements for Estuarine Water Quality Management*" by Taljaard and Slinger (1997) for a more comprehensive analysis of the context of water quality management in South African estuaries, prior to reading the subsequent discussion on the utility and performance of the Mike 11 modelling system.

5.2 The Utility and Performance of Mike 11 in Modelling Estuarine Water Quality

Information on the utility and performance of Mike 11 in addressing water quality issues in South Africa can be obtained by examining its application to the two case studies, the Great Berg and Swartkops estuaries. The systems chosen are both relatively long, narrow estuaries with permanently open mouths for which a one-dimensional approach is applicable. In addition, a reasonable amount of data existed on each of these systems; in fact, more data than is commonly available on most South African estuaries. In the implementation of the Mike 11 hydrodynamic, transport-dispersion and water quality modules on the Great Berg Estuary, both the high flow conditions typical of the winter season and the low flow conditions typical of the summer season were considered. The water quality module with sediment-water column interaction was implemented, and temperatures, dissolved oxygen concentrations and biological oxygen demand levels were simulated in addition to the simulations of water levels, flows and salinities. Representative ranges of thermal variation and dissolved oxygen concentrations were simulated for the winter 1989, summer 1990, winter 1995 and summer 1996 situations. Generally, DO levels at distances greater than 20 km from the mouth were characteristically low compared with those nearer the mouth. A higher degree of variability was also evident in the more saline, lower reaches both in the measured and the simulated values. The simulated range of variation of DO in this region of the estuary was less than the measured range apart from under the high flow conditions of August 1995 when the system was freshwater dominated. This more limited variation of the simulations is ascribed primarily to limited data for the seaward boundary, but may also arise from the fact that the effect of salinity on the solubility of oxygen is not incorporated in the Mike 11 model (owing to the de-coupling of the transport-dispersion and water quality modules). More extensive information on variations in the DO and BOD concentrations of the inflowing seawater would be required before the reasons for this feature could be determined absolutely.

The sediment oxygen demand, incorporating the natural or ambient biological oxygen demand within the estuary, was discovered to play a critical role in the calibration of the DO component of the model for the Great Berg Estuary. This proved to be true in the Swartkops Estuary as well, although intensive calibration of this application was not undertaken owing primarily to data limitations. While many data exist for this system, they were collected with purposes other than model calibration in mind and so are not necessarily appropriate to this use. Despite these limitations, a number of characteristic run-off situations and thermal conditions were simulated and the thermal and dissolved oxygen dynamics of the estuary were simulated. Time periods of twenty-eight days covering a full spring-neap-spring cycle were selected for simulation. The range of variation of the water quality parameters with position along the estuary and variations in inflow conditions could then be examined. Features of the estuary such as the almost uniform salinities under low flow conditions and the relatively well-oxygenated conditions under strong tidal circulation were evident. Clearly, the effects of altered inflow conditions and waste loading could be investigated using the model as it stands at present, but it would be preferable to undertake the necessary calibration studies to ensure the validity of the simulation results.

Dissolved nutrient levels have not been modelled using the Mike 11 water quality module for either the Great Berg or the Swartkops case studies as field investigations for the winter condition in the Great Berg have indicated that dissolved nutrients are strongly linearly correlated with salinity, while dissolved reactive phosphate-P concentrations and dissolved reactive silicate-Si levels in the Swartkops exhibit similarly strong relationships to salinity. Owing to the decoupling between salinity modelling and the water quality module of Mike 11, it seemed superfluous to attempt to model the nutrient dynamics in relation to temperature and dissolved oxygen processes when they can already be predicted fairly accurately from salinity simulations. Under summer conditions in the Great Berg such relationships do not exist, particularly in regard to total dissolved inorganic nitrogen-N or its components (dissolved nitrite-N, dissolved nitrate-N and total dissolved ammonia-N). Instead, depletion of total inorganic nitrogen appears to occur in the middle reaches of the estuary. A similar situation exists for the Swartkops Estuary where the total dissolved inorganic nitrogen levels show no clear relationship to salinity (note that they are thus also decoupled from dissolved reactive phosphate-P concentrations) and there are no clear patterns of generation or depletion in the different areas of the estuary. As no conceptual model of the biogeochemical processes operative in the estuaries could be derived from the available data, no modelling of the complex behaviour of dissolved nutrient levels was attempted.

As mentioned previously, the summer 1996 field data indicate that dissolved oxygen levels can decrease substantially over the summer period. Previous work on the Great Berg Estuary indicated that the resident nature of the water in the middle reaches over summer led to depletion of dissolved oxygen in the estuarine water. While this is definitely a strong contributory factor, this preliminary modelling work indicates that the quality of the inflowing water (particularly the river water) plays a major role in determining the DO and nutrient levels of the middle reaches of the estuary towards the end of summer. Evidently, the water entering the system is low in oxygen over the summer period. The reasons for this should be established as this may be a natural feature or may be aggravated by the severe infestation of the upper estuary and river by water hyacinth. The need to establish these reasons emphasises the need for monitoring, particularly of the boundary conditions.

An additional test of the utility and performance of the Mike 11 water quality module was undertaken by applying it to the low oxygen event or 'black tide' which occurred in St Helena Bay in March 1994. This event was marked by the intrusion of water entirely depleted in oxygen into the estuary. Massive fish and invertebrate kills were observed. Information on the possible limit of intrusion and the duration of low oxygen conditions in the estuary would contribute to current studies of the fish and benthic invertebrates of the Great Berg Estuary (Mr S Lamberth pers. comm., Mr I Bickerton pers. comm.). Thus all available data on the event were collated and despite little scientific data at the onset of the event and slightly better data only well into the event, simulations of the probable conditions were undertaken. Results indicated that DO levels below 2 $mg.l^{-1}$ persisted in the lower estuary for six to seven days, but that such low dissolved oxygen concentrations did not extend 10 km up the estuary. This means that the Great Berg Estuary, which is highly saline in summer, could act as a refuge to any mobile animal able to make its way this distance upstream. This application also demonstrates the efficacy of the Mike 11 model in enabling the association of water quality to biological assessment and prediction, a desirable development in terms of the needs of water quality management in South Africa.

Thus the Mike 11 system performs well in modelling the salinity distributions, thermal variations and dissolved oxygen concentrations in both the Great Berg and Swartkops Estuaries and seems well suited to applications on permanently open South African estuaries. While accurate simulations are primarily dependent on appropriate calibration and monitoring data, the water quality dynamics of these systems were reasonably well represented. The utility of the Mike 11 system in addressing the water quality issues of South African estuaries relates both to the type of estuary and the anticipated water quality changes. It has already been seen that it is applicable to estuaries where the mouth configuration is fairly stable and from previous studies it is known that, in association with other predictive techniques, it is also applicable to temporarily closed systems. However, at issue here is whether Mike 11 is applicable in matching performance measures and the response of the abiotic or chemical environment. This was not specifically tested, instead the variation of water quality constituents in the estuarine environement were simulated. The step from there to analysing compliance e.g. what percentage of time and under what conditions certain values are exceeded, is not difficult provided that the compliance limits or guidelines are specified. Thus compliance testing is certainly achievable for parameters such as salinity, temperature and dissolved oxygen using the Mike 11 system, but aspects such as biological contamination cannot be addressed at present using this system as CSIR does not have this module. For all water quality constituents, however, the development of estuarine specific water quality guidelines is a necessary interim step if model applications are to become routine decision support tools in water quality management. Additionally, skills in associating water quality changes and biotic responses are necessary. This is an area of interest worldwide and South African scientists are positioned to contribute substantially in this field as much groundwork has already been undertaken in predicting the effects of freshwater limitation. Mike 11 can play a role in linking the human activities and developments to water quality changes and hence to the prediction of biological responses, so assisting in ensuring the continued existence of healthy estuarine environments.

In applying Mike 11 to the case studies and analysing its capabilities, we have also seen that model performance and utility are intimately linked with data availability. The design of appropriate monitoring methods is thus an area requiring further attention if adequate decision support for management is to be provided in the long term.

Techniques other than modelling can prove useful in addressing some management needs. For example, the association of dissolved nutrient distributions to salinity distributions under certain flow conditions (the data analysis techniques of the document by Taljaard and Slinger (1997)) using simple regression techniques provides more effective and accurate predictions than a full model application. Similarly, well designed field investigations can provide the information from which new understanding and insights can be developed. What should never be overlooked is the interaction between monitoring, investigative field work, appropriate analysis techniques and successful model implementations and predictions.

5.3 Necessary Future Activities

Limitations in the current modelling activities relate to the prediction of bacterial contamination and the simulation of dissolved nutrient distributions under all flow conditions, that is, both when bio-geochemical regeneration or depletion processes become significant as well as when the distributions are circulation-dependent. In the case of the former, further mathematical modelling skills need to be developed or acquired particularly as this is an acknowledged problem in the KwaZulu -Natal area and on other South African estuaries e.g. the Swartkops system. In the latter case, investigative field research is necessary to establish and quantify a conceptual model of nutrient dynamics before full scale predictive modelling can be undertaken, and to establish whether this is even necessary.

The necessity for the development of the estuarine water quality guidelines is also evident, particularly where compliance testing needs to be undertaken. This would also provide a framework within which the development of appropriate water quality indices and biological assessment and prediction techniques could be developed and would assist in ensuring that the focus remained on linking the predicted or measured water quality changes to the observed effects and not on the study of the biological response in isolation of the causative effects.

A strong indication from this study is the necessity for the design of appropriate monitoring strategies, so that modelling investigations can be made possible in future and so contribute effectively to water quality management decision support. Additionally, the link between the river and estuary monitoring programmes needs to be well established so that the long term response of an estuary can be linked to specific sources and not just to some isolated change in the quality and quantity of the inflowing river water.

Last, but not least, it is important that the modelling tools applicable to the South African estuarine environment are known and their individual limitations appreciated. This will obviate the impression that a model can answer all questions and will also initiate to using simpler models at times or even coupling different types of models to generate the required simulations. For instance, it may be possible to use the stable and reliable hydrodynamic and transport-dispersion modules of Mike 11 in association with more comprehensive water quality models, or even just with particular formulae, to simulate the variables of interest rather than having to work through a complex water quality module. A robust and pragmatic approach to water quality modelling is thus advocated in the South African situation.

6. RECOMMENDATIONS

Water quality modelling of South African estuaries is one of the components of holistic prediction in estuaries which requires further development in order that management of water quality can take place from the perspective of preventing the occurrence of problems rather than allowing them to develop and then attempting to solve them. The human activities affecting estuaries will continue to exert more pressure on these natural systems as South Africa develops and water quality issues will play a more dominant role in decision making as time proceeds. To be able to tackle the present and future water quality concerns in South African estuaries the following strategy is recommended:

1. The **formulation of a pragmatic decision support framework for estuarine management** integrating information on aspects such as water quality monitoring and prediction, water quantity management and development options and the effects of catchment land-use practices on downstream estuaries, in such a way as to enable the implications of different management measures to be explored realistically. Inputs to such a system were formulated in the current study (and are described in the document entitled "*Decision Support Requirements for Estuarine Water Quality Management*" (Taljaard & Slinger 1997)) and some components have already been developed under the auspices of the Co-ordinated Research Programme on Decision Support for the Conservation and Management of Estuaries undertaken for the Water Research Commission by the Consortium for Estuarine Research and Management (Slinger 1996), but the overall management decision support system has yet to be developed;

2. The **establishment of South African water quality guidelines for estuaries**. The guidelines for the freshwater and marine environments have been revised recently, but a similar process has not yet been initiated for estuaries. Such guidelines are necessary for linking the prediction of water quality constituent concentrations to the effects on the users of a system and for testing compliance with system or environmental objectives;

3. The **development of a comprehensive monitoring strategy for South African estuaries** and the establishment of appropriate procedures for processing and analysing monitoring data. It is important that this monitoring strategy provides data useful for present and future estuarine investigations and niches with the monitoring programmes for rivers and catchments so that the long term effects of specific changes in human activities in the catchments as well as in the estuaries themselves can be established. The role of estuarine scientists in the development of an appropriate monitoring programme cannot be underestimated;

4. The **initiation of investigative studies to address deficiencies in the understanding of nutrient dynamics in South African estuaries**. Both field activities, data analysis procedures and model implementations would be required for such research. Examples of such deficiencies in understanding include the apparent decoupling of dissolved reactive phosphate-P and total inorganic nitrogen-N in the Swartkops Estuary, a matter deserving of further investigation, and the lack of even a conceptual understanding of the relative importance of the various bio-geochemical processes modifying the total inorganic nitrogen-N concentrations in both the Great Berg and Swartkops Estuaries, a necessary precursor to any modelling attempts;

5. The **acquisition and/or development of a capability to model bacterial contamination in South African estuaries**. This is considered important as bacterial contamination is presently an issue of concern in certain estuaries and will probably become more prevalent in future.

6. **Encouragement of the inclusion of water quality aspects in existing or developing biological response models** so that the prediction of the response of biota to water quantity and quality changes can be addressed more comprehensively in future.

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PERSONAL COMMUNICATIONS

Dr J Adams	Botany Department, University of Port Elizabeth.
Dr G Bailey	Sea Fisheries Research Institute, Cape Town.
Prof D Baird	Zoology Department, University of Port Elizabeth.
Ms A Belcher	Institute for Water Quality Studies, Department of Water Affairs and Forestry, Pretoria.
Mr I Bickerton	CSIR, Stellenbosch.
Ms P Brown	Wallingford Hydraulics Institute, Wallingford, UK
Mr T Harrison	CSIR, Durban.
Dr Z Ibrahim	University of Malaysia, Penang, Malasia
Dr H Los	Delft Hydraulics, Delft, The Netherlands
Dr H MacKay	Institute for Water Quality Studies, Department of Water Affairs and Forestry, Pretoria
Mr S Lamberth	Zoology Department, University of Cape Town.
Dr M Lingby	Danish Hydraulic Institute, Horsholm, Denmark.
Ms U Scharler	Zoology Department, University of Port Elizabeth.
Prof G Stelling	Delft University of Technology and Delft Hydraulics, Delft, The Netherlands
Mr R Thomas	Applied Science Associates, London, UK
Mr J van Gils	Delft Hydraulics, Delft, The Netherlands
Ms N Villars	Delft Hydraulics, Delft, The Netherlands
Mr Von der Molen	Zoology Department, University of Port Elizabeth.
Dr R Vos	Delft Hydraulics, Delft, The Netherlands
Mr D Walstra	Delft Hydraulics, Delft, The Netherlands

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APPENDIX A

MIKE 11 WATER QUALITY MODELLING APPLICATIONS.
FIELD DATA FOR SWARTKOPS RIVER ESTUARY

December 1996

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A.1 SALINITY, TEMPERATURE AND DISSOLVED OXYGEN

A.1.1 Field Data

Field data on salinity, temperature and dissolved oxygen measurements in the Swartkops River Estuary was available from the following sources:

- MacKay (1993)
- MacKay (pers. comm.)
- Scharler (pers. comm.)
- CSIR (personal observation on 16 August 1996)
- Emmerson (1985).

For the purpose of this study the data of Scharler, UPE (pers. comm.), CSIR (observations on 16 August 1996) were used to depict typical conditions in the estuary.

Figure A.1(a) and A.1(b) show the salinity and temperature profiles measured in the estuary on 16 August 1996 on the ebb tide. Although the river inflow was not measured at the time, it was considered to be typical of low flow conditions, i.e. less than 0,5 m^3 s^{-1}.

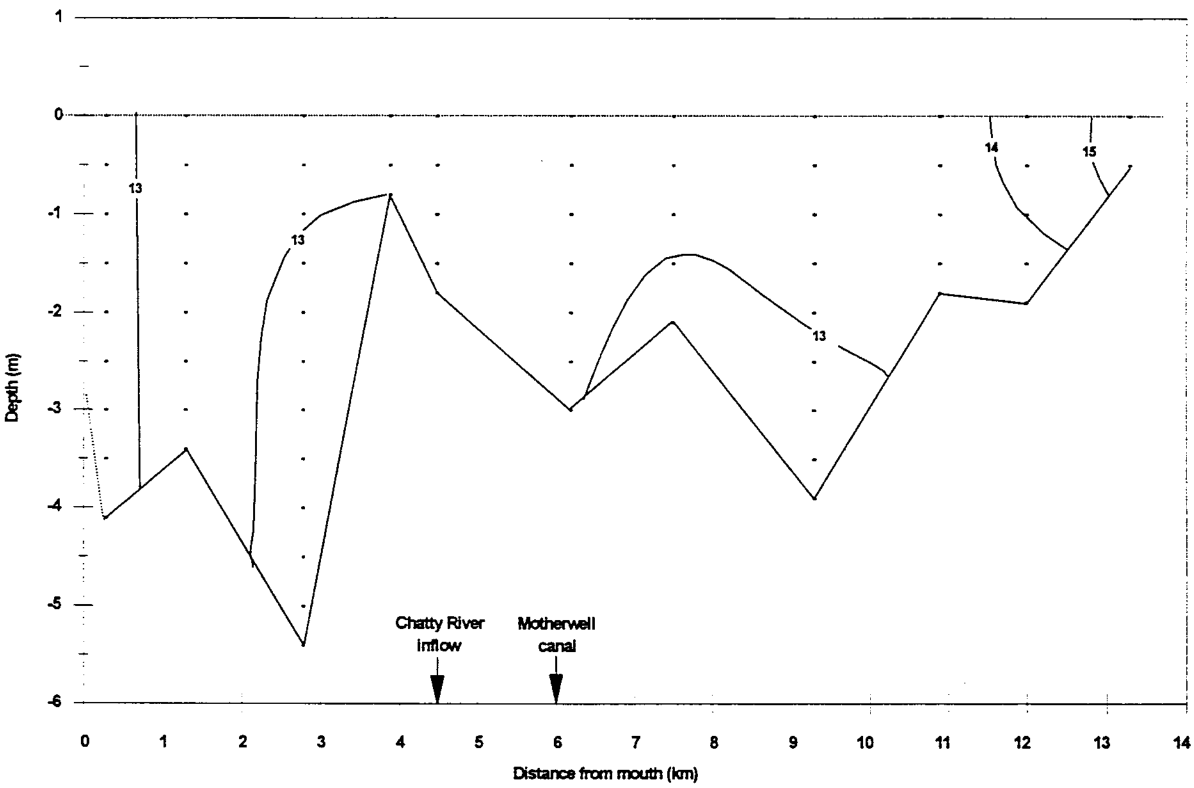

Figure A.1(a). Temperature profile (°C) measured in the Swartkops River Estuary during ebb tide on 16 August 1996

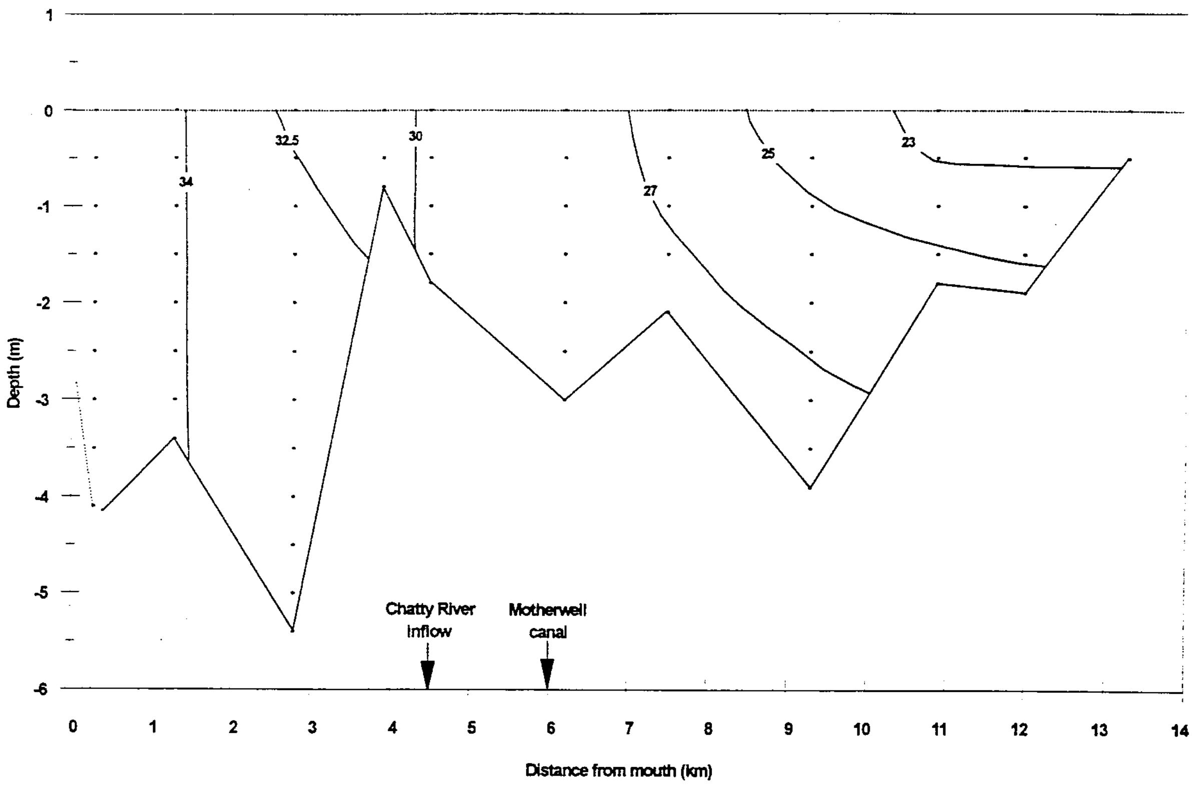

Figure A.1(b). Salinity (ppt) profile measured in the Swartkops River Estuary during ebb tide on 16 August 1996

The data collected by Scharler, UPE (pers. comm.) and CSIR (observations on 16 August 1996) were used to illustrate typical conditions under *low* and *high* river inflow conditions. Because the catchment of the Swartkops River Estuary does not lie in a rainfall area which is strongly seasonal, it was considered more appropriate to refer to low and high inflow conditions rather than summer or winter conditions. Typical temperature, salinity and DO measurements in the estuary during low and high inflow conditions are presented in Figures A.2(a), A.2(b) & A.2(c) and A.3(a) & A.3(b), respectively. All measurements were taken on spring tides.

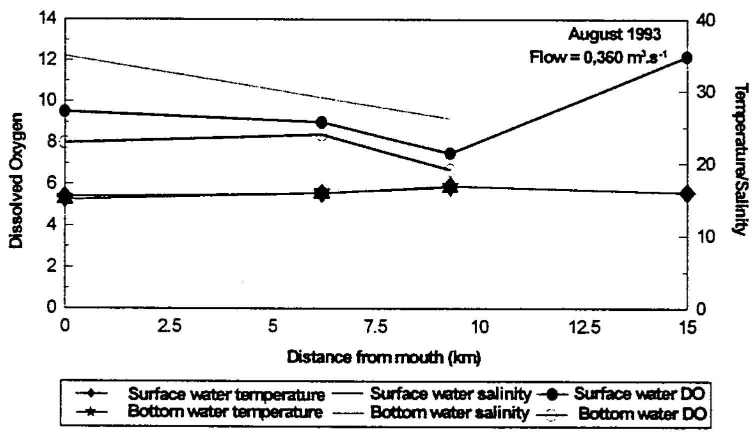

Figure A.2(a). Temperature (°C), salinity (ppt) and DO concentrations (mg.l^{-1}) measured in surface and bottom waters in the Swartkops river estuary during August 1993

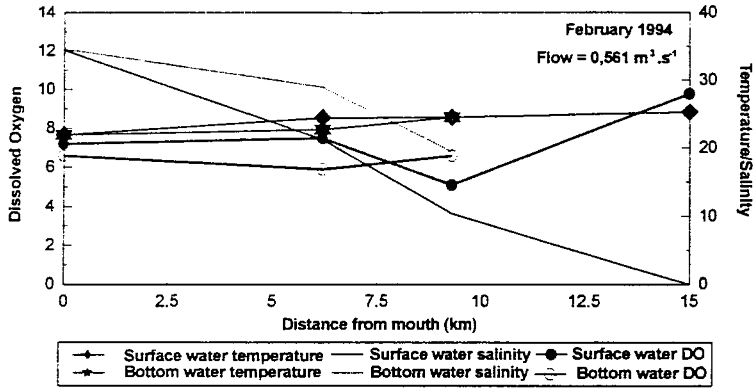

Figure A.2(b). **Temperature (°C), salinity (ppt) and DO concentrations (mg.l^{-1}) measured in surface and bottom waters in the Swartkops river estuary during February 1994**

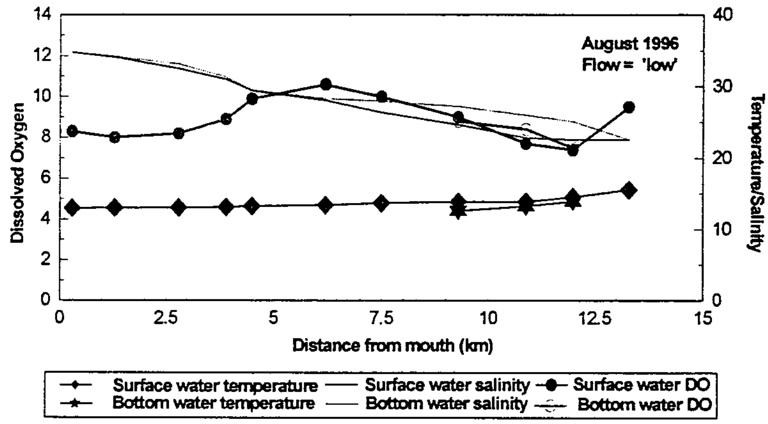

Figure A.2(c). **Temperature (°C), salinity (ppt) and DO concentrations (mg.l^{-1}) measured in surface and bottom waters in the Swartkops river estuary during August 1996**

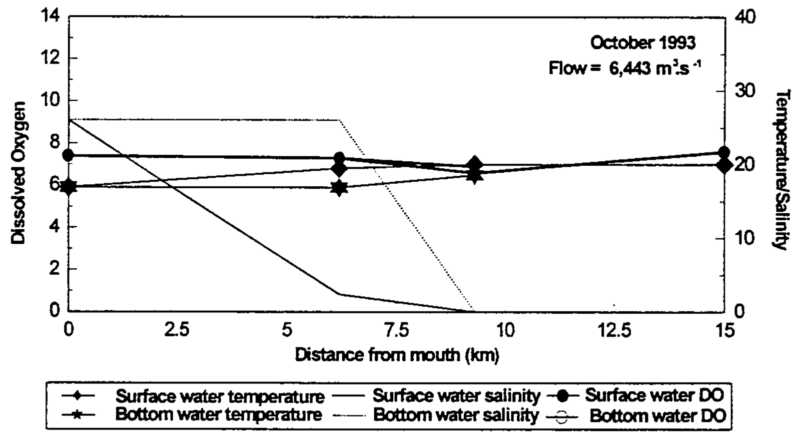

Figure A.3(a). **Temperature (°C), salinity (ppt) and DO concentrations (mg.l^{-1}) measured in surface and bottom waters in the Swartkops river estuary during October 1993**

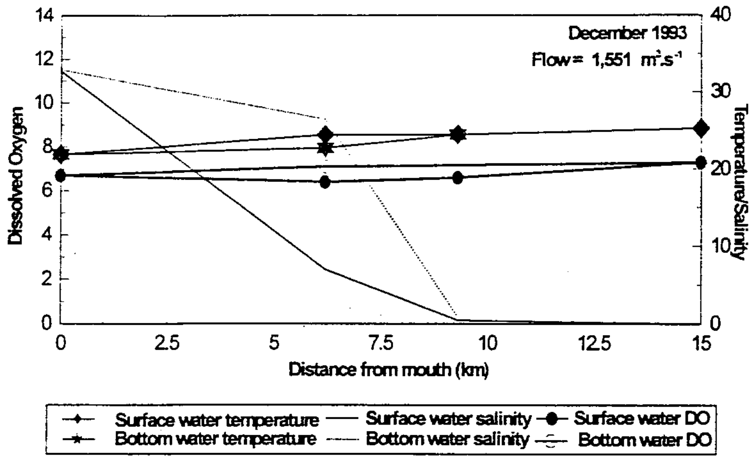

Figure A.3(b). Temperature (°C), salinity (ppt) and DO concentrations (mg.l^{-1}) measured in surface and bottom waters in the Swartkops river estuary during December 1993

The conditions in the seawater and river water at the time of these measurements are provided in Table A.1.

TABLE A.1. Seawater and river water conditions

DATE	RIVER FLOW (m^3.s^{-1})	SEA			RIVER		
		Temp (°C)	Salinity (ppt)	DO (mg.l^{-1})	Temp (°C)	Salinity (ppt)	DO (mg.l^{-1})
August 1993	0,360	15,2	34,8	8,3	16	0	9,3
February 1994	0,561	23	34,6	7,3	25,3	0	9,8
August 1996	'low'	13	34,8	8,3	-	-	-
October 1993	6,443	-	-	-	20	0	7,6
December 1993	1,551				25,3	0	7,3

From these results it appears as if DO concentrations in the river water are generally higher during low flow than during high inflow conditions (Table A.1).

A.1.2 Recommended input data for Mike 11

Based on the above-mentioned information and understanding of South African estuarine systems, the following input parameters for Mike 11 are recommended:

i. Low flow conditions

PARAMETER		CONDITION 1	CONDITION 2
River inflow		0,5 $m^3.s^{-1}$	0,5 $m^3.s^{-1}$
Sediment oxygen demand (SOD)		0,2 g $O_2.m^{-2}.day^{-1}$	0,2 g $O_2.m^{-2}.day^{-1}$
BOD:	seawater (0 km) upstream (14,9 km)	1 $mg.l^{-1}$ 1 $mg.l^{-1}$	1 $mg.l^{-1}$ 1 $mg.l^{-1}$
Temperature:	seawater upstream	19,6 °C 25 °C	15,2 °C 20 °C
Salinity:	seawater upstream	35 ppt 0 ppt	35 ppt 0 ppt
DO:	seawater upstream	7,31 $mg.l^{-1}$ 7,75 $mg.l^{-1}$	8,3 $mg.l^{-1}$ 9,3 $mg.l^{-1}$

ii. High flow conditions

PARAMETER		CONDITION
River inflow		5 m^3 s^{-1}
Sediment oxygen demand (SOD)		0,2 g $O_2.m^{-2}.day^{-1}$
BOD:	seawater (0 km) upstream (14,9 km)	1 $mg.l^{-1}$ 1 $mg.l^{-1}$
Temperature:	seawater upstream	19,6 °C 20 °C
Salinity:	seawater upstream	35 ppt 0 ppt
DO:	seawater upstream	7,3 $mg.l^{-1}$ 7,75 $mg.l^{-1}$

A.2 DISSOLVED NUTRIENTS

A.2.1 Field Data

The nutrient data collected by Scharler, UPE (pers. comm.) and CSIR (observations on 16 August 1996) were used. The relationship between salinity and nutrients is presented in Figures A.6 to A.11.

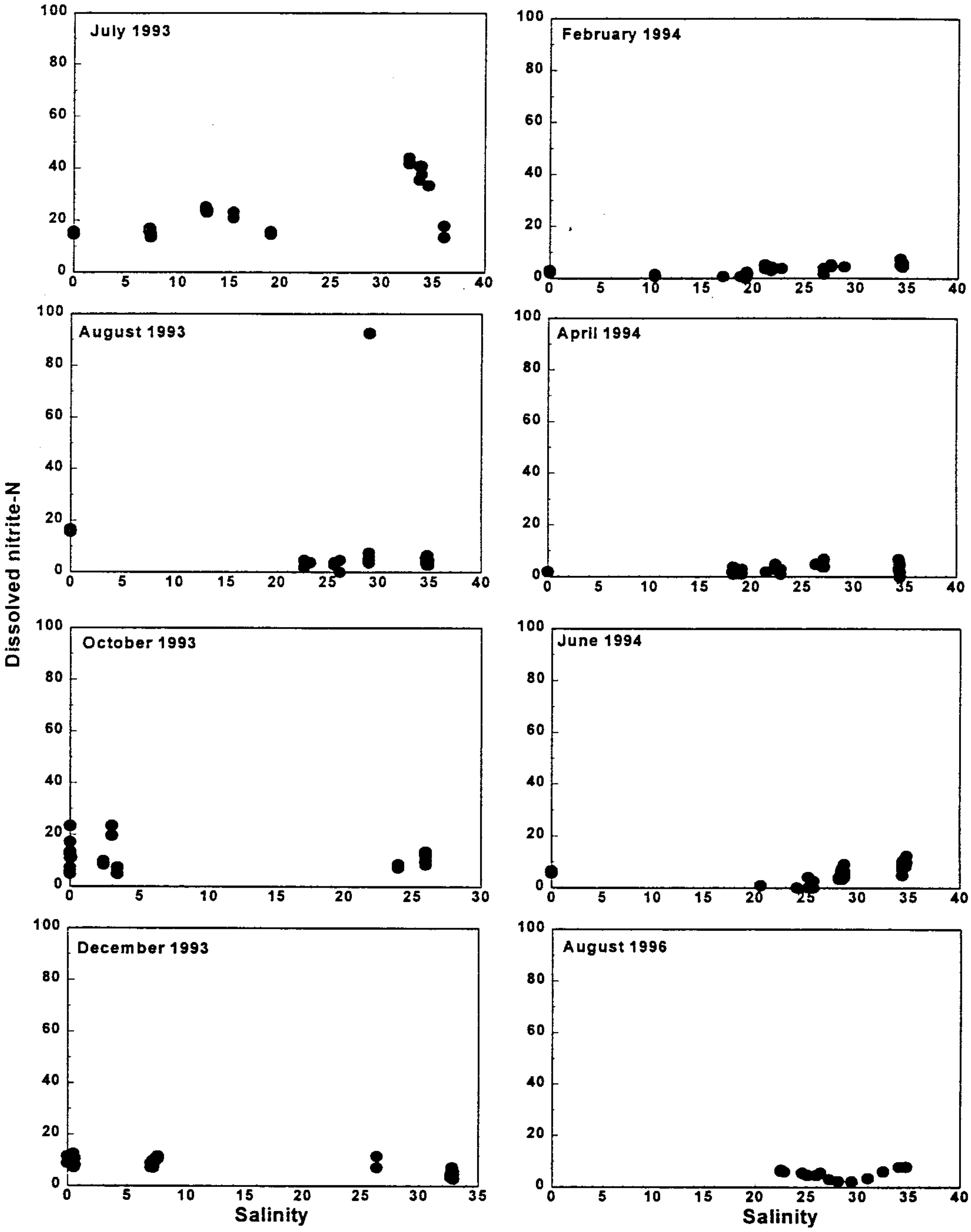

Figure A.6 Relationship between salinity (ppt) and dissolved nitrite-N concentrations (μg.l^{-1})

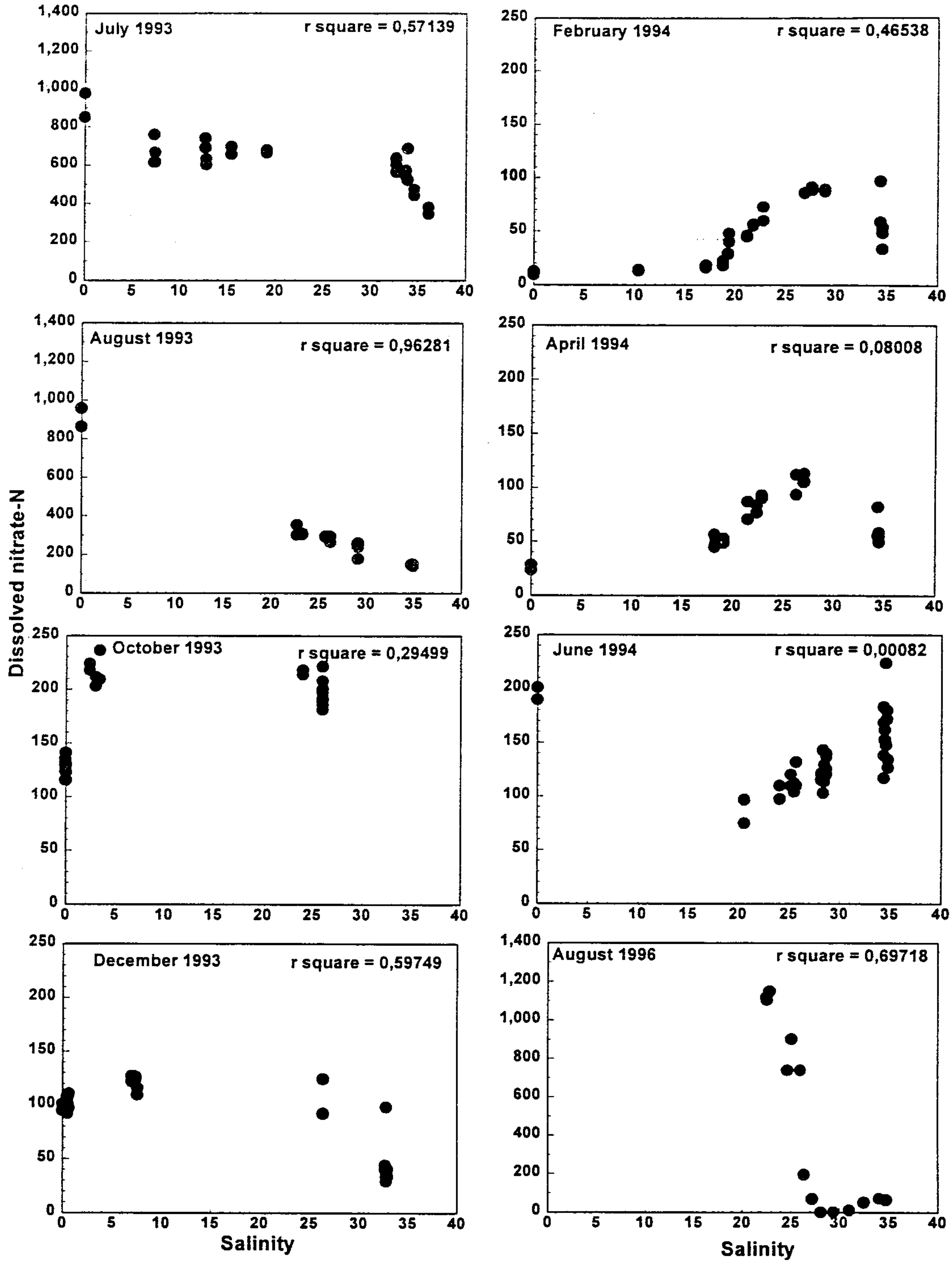

Figure A.7 **Relationship between salinity (ppt) and dissolved nitrate-N concentrations ($\mu g.l^{-1}$)**

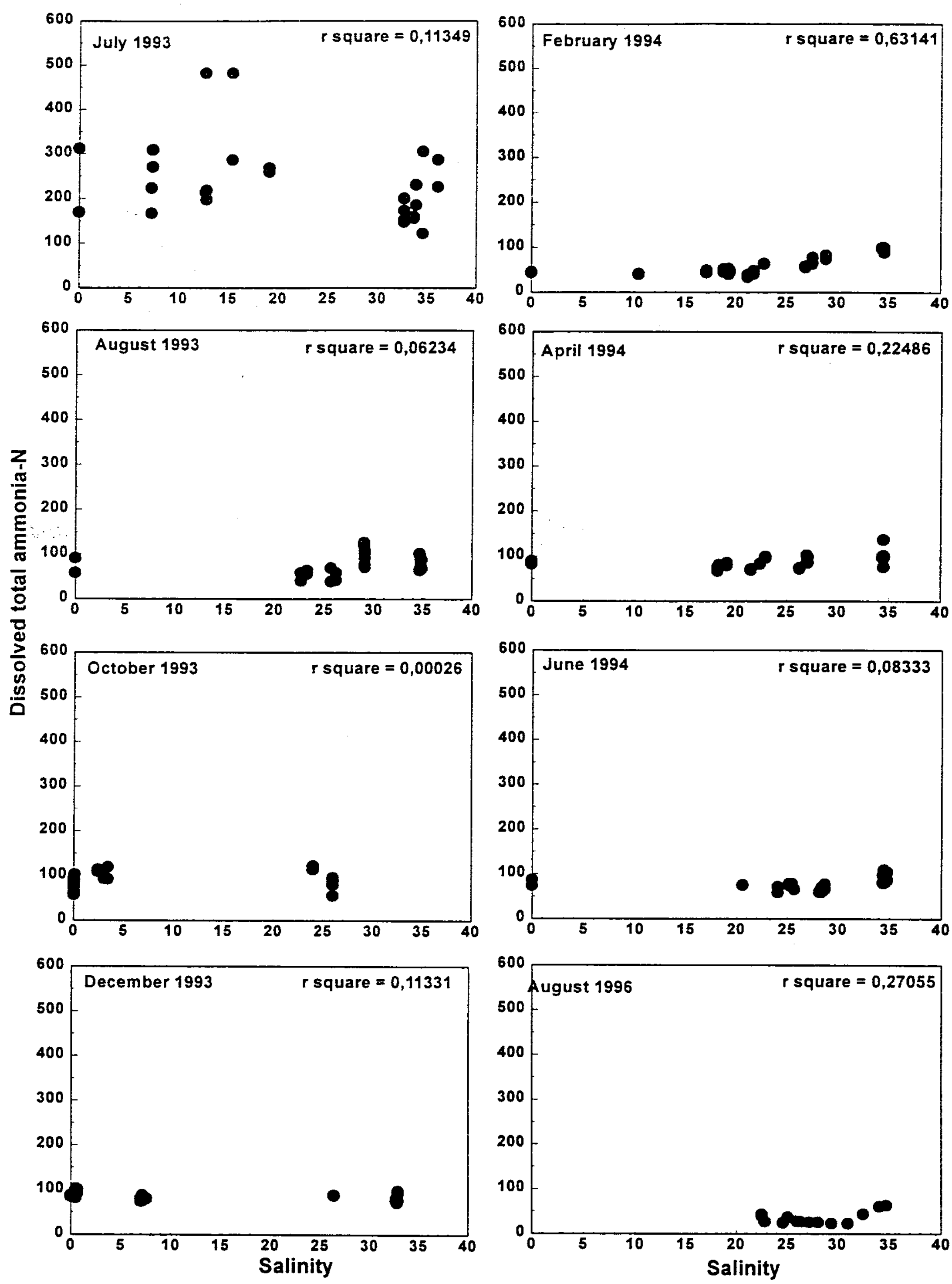

Figure A.8 Relationship between salinity (ppt) and dissolved total ammonia-N concentrations ($\mu g.l^{-1}$)

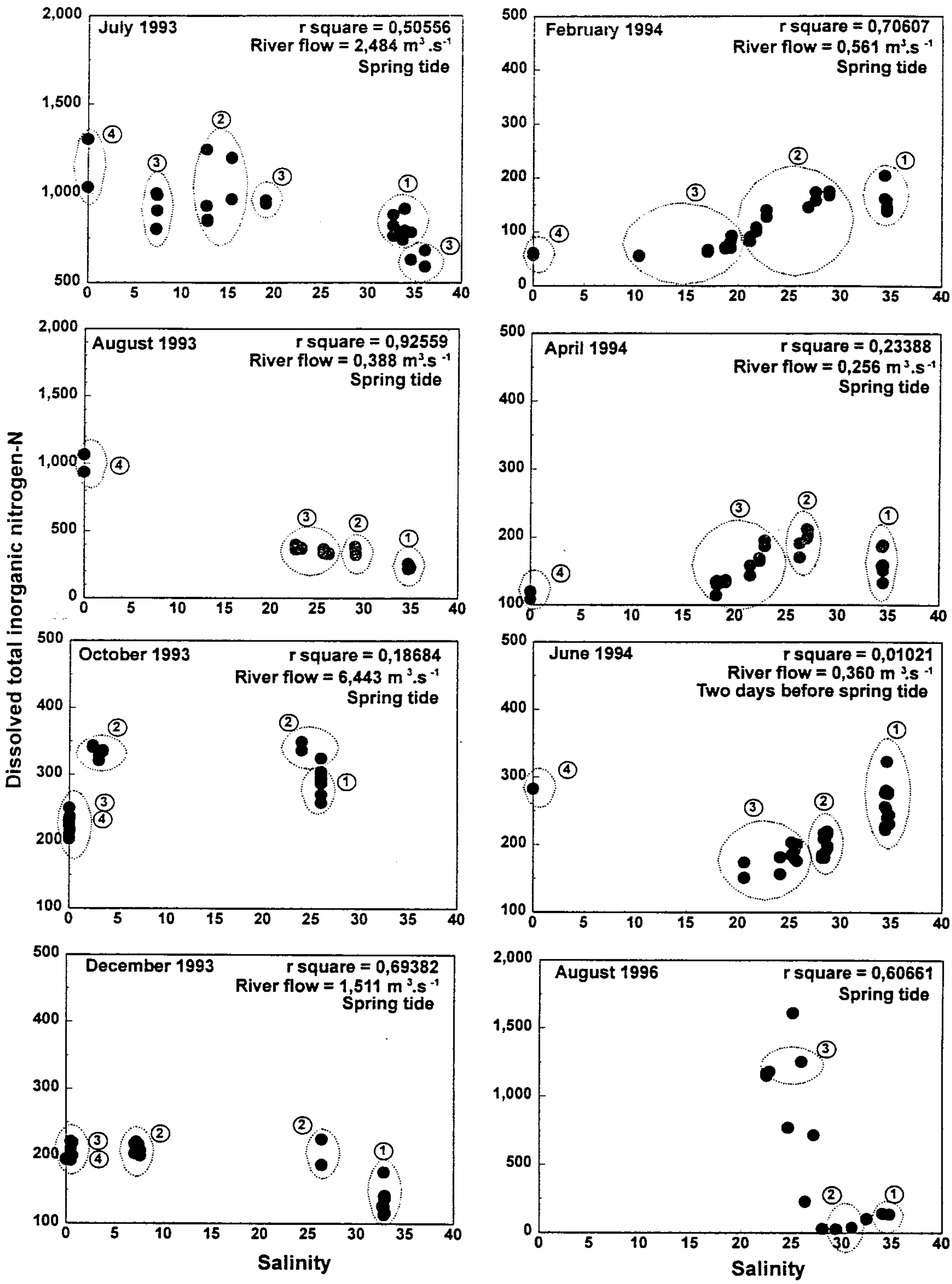

Figure A.9 Relationship between salinity (ppt) and dissolved total inorganic nitrogen-N concentrations ($\mu g.l^{-1}$) (1 = Amsterdamhoek; 2 = Brickfields; 3 = Bar None; 4 = Perseverance)

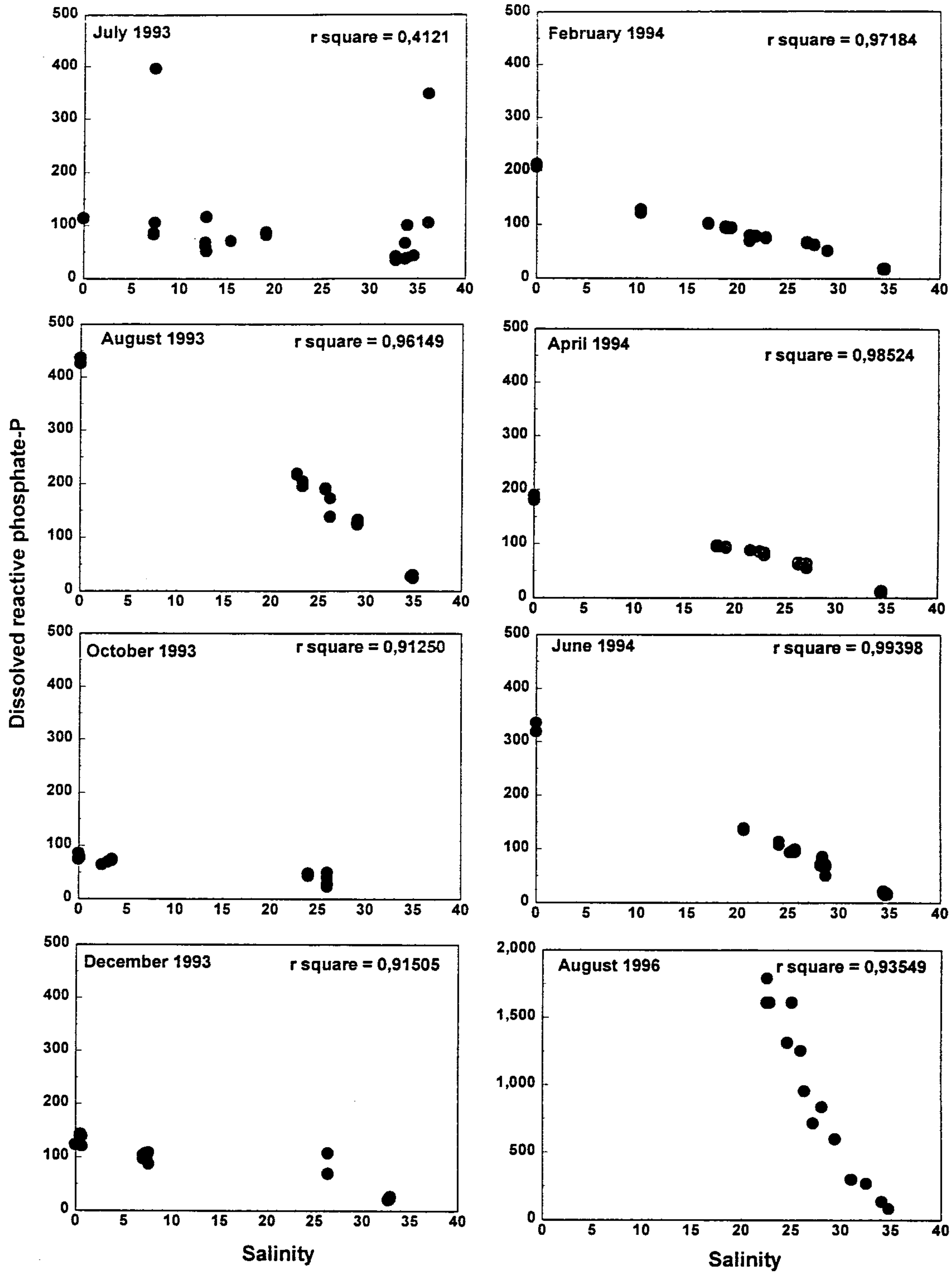

Figure A.10 Relationship between salinity (ppt) and dissolved reactive phosphate-P concentrations (μg.l[-1])

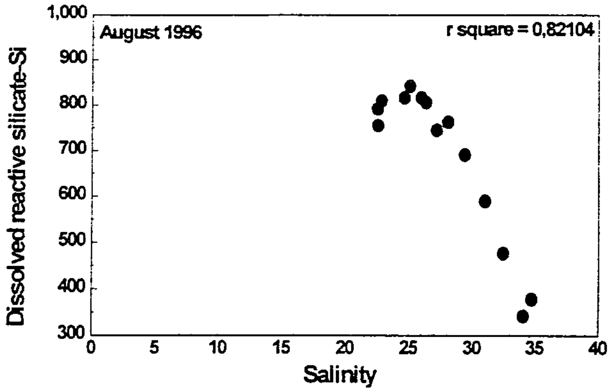

Figure A.11 Relationship between salinity (ppt) and dissolved reactive silicate-Si concentrations ($\mu g.l^{-1}$)

A.2.2 Recommendations

Dissolved reactive phosphate-P (Figure A.10) and to a lesser extent reactive silicate-Si (Figure A.11) showed a strong linear relationship with salinity, i.e. they are 'conservative'. The relationship between the other dissolved nutrients and salinity was less straightforward. This suggested that these nutrient concentrations in the estuary were probably subjected to biochemical processes within the estuary which modified concentrations, for example photosynthesis, denitrification, nitrification, etc.

Knowledge of these processes in the estuary is not sufficient both in terms of the type of processes and their different contributions. It is therefore recommended these be investigated further, so as to provide a better conceptual picture, prior to application of numerical modelling techniques

The concentration of conservative parameters can be linked to salinity outputs by the following equation:

$$f(x) = \frac{C_{sea} - C_{riv}}{S_{sea} - S_{riv}} x + C_{riv}$$

Where	f(x)	= concentrations at salinity x
	x	= salinity
	C_{sea}	= concentration in sea
	C_{riv}	= concentration in river
	S_{sea}	= salinity of sea
	S_{riv}	= salinity of river

A.3 *E. coli*

A.3.1 Field Data

i. River inflow

Data on the *E coli* counts in river inflow was taken from the DWAF data set (Van der Molen, UPE, pers comm.), Emmerson (1985) and MacKay (1993).

The DWAF data set runs from 28 June 1993 to 28 September 1995. On two occasions during this period, *E. coli* counts measured at the Perseverance Drift were listed as *'innumerable counts'*, i.e. 6 October 1993 and 20 January 1995. These were assumed to correspond to periods of high river flow (river flow measurements taken by UPE during October 1993 were about 6 m^3 s^{-1}).

Excluding the data points of 6 October 1996 and 20 January 1995, the average *E. coli* count in river inflow was about *50 counts per 100 ml*, which is considered to be typical of low inflow conditions. MacKay (1993) gave *100 counts per 100 ml* as a typical low flow concentration (0,2 to 0,4 m^3 s^{-1}).

Emmerson (1985), stated that during high flow conditions, *E. coli* counts in the river inflow increased, owing to the Uitenhage sewage works not being able to cope with the larger volumes and then overflowing into the river prior to proper treatment. Although the exact measurements were not given, a range of concentrations was provided. Knowing that the highest counts were measured during high flow conditions, it was assumed that the highest value in the range at the topmost station, i.e. Perseverance, should be representative of *E. coli* counts under high river inflow conditions. The highest value in the range is *3 000 counts per 100 ml*. MacKay (1993) gave *10 000 counts per 100 ml* as typical of counts during higher flows.

ii. Motherwell canal

Data on the *E. coli* counts and flows in the Motherwell canal were obtained from MacKay (pers. comm.) and MacKay (1993).

The Motherwell canal drains an area of about 13,5 km^2 (MacKay, 1993).

Typical dry weather flow *(base flow)* was given as 0,002 $m^3.s^{-1}$, with little variability (MacKay, 1993).

During March 1990, the flow pattern in the Motherwell canal was recorded over a rain event (5 dry days just prior to the measurements, with about 9,5 mm rain in the catchment over 24 hours) (MacKay, 1993; MacKay, pers. comm.). Results are tabulated below:

TABLE A.2 Flows measured in the Motherwell canal on 14/15 March 1990 (MacKay, pers. comm)

TIME AFTER 0:00 ON 14 MARCH 1990 (hrs)	FLOW ($m^3.s^{-1}$)
9	0,0015
17	0,056
19	0,113
20	0,09
21	0,72
22	0,63
23	0,39
24	0,37
25 (0:00 on 15 March)	0,35
32	0,0025

Unlike parameters like COD, which decreased during wet weather flows (i.e. rain events), *E coli* counts in the Motherwell canal remained high during both wet and dry weather flows (MacKay, 1993). Typical *E. coli* counts for Motherwell were given as *1 x 10^6 counts per 100 ml* (MacKay, 1993).

A.3.2 Recommended input data for Mike 11

Based on the above-mentioned information, the following input data are recommended for application of Mike 11 in the Swartkops River Estuary:

INFLOW SOURCE	FLOW ($m^3.s^{-1}$)	*E. coli* (count per 100 ml)
River inflow (low inflow)	0,5	100
River inflow (high inflow)	5,0	10 000
Motherwell canal (base flow)	0,002	1 x 10^6
Motherwell canal (rain event - 9,5 mm)	Refer to Table 1	1 x 10^6 (throughout)

REFERENCES FOR APPENDIX A

CSIR (observation on 16 August 1996) Water quality data collected on 16 August 1996 at the Swartkops River Estuary.

EMMERSON, W D (1985) The nutrient status of the Swartkops River Estuary, eastern Cape. *Water SA* **11**(4): 189-198.

MACKAY, H M (1993) The impact of urban runoff on the water quality of the Swartkops estuary: Implications for water quality management. Water Research Commission report KV 45/93. Pretoria 217 pp.

PERSONAL COMMUNICATION LIST FOR APPENDIX A

MACKAY, H M (pers. comm.) Raw data collected in the Swartkops River Estuary during 1990 to 1992. DWA&F, Pretoria.

SCHARLER U (pers. comm.) Raw data collected in the Swartkops River Estuary from July 1993 to April 1994. University of Port Elizabeth, Port Elizabeth.

VON DER MOLEN (pers. comm.). Data collected from selected sites along the Swartkops River Estuary from 1990 to 1995 by DWA&F, Eastern Cape. University of Port Elizabeth, Port Elizabeth.

APPENDIX B

RAW DATA

July 1993

Stn	depth m	Sal ppt	Temp C	pH	O2 mg/l	O2 % Sat	PO4-P ug/l	NO3-N ug/l	NO2-N ug/l	NH4-N ug/l
1	0	32.7	15.3	8.07			34.2	638.5	43.0	200.2
	0	32.7	15.3				41.0	566.8	43.0	152.6
1	1	32.7	15.3	8.08			42.4	631.5	44.0	147.7
	1	32.7	15.3				38.3	603.7	41.9	173.3
1	2	33.7	15.3	8.04			67.0	574.1	35.6	159.9
	2	33.7	15.3				36.9	545.0	40.9	156.2
1	3	33.9	15.3	8.03			39.6	688.5	40.9	185.5
	3	33.9	15.3				99.8	524.2	37.7	230.7
2	0	12.7	14.6	7.89			61.5	741.1	24.1	482.1
	0	12.7	14.6				68.3	692.2	25.2	213.6
2	1	12.8	14.6	8.02			51.9	604.7	23.1	218.5
	1	12.8	14.6				116.2	633.5	24.1	197.7
2	2	15.4	14.4	7.87			71.1	658.5	23.1	286.8
	2	15.4	14.4				71.1	696.4	21.0	482.1
2	3	34.6	15.2	7.94			43.7	444.7	33.5	305.1
	3	34.6	15.2				43.7	474.6	33.5	122.1
3	0	7.3	14.8	7.62			83.4	616.9	16.8	167.2
	0	7.3	14.8				86.1	761.4	15.7	223.4
3	1	7.4	14.7	7.72			396.3	619.0	14.7	271.0
	1	7.4	14.7				105.2	667.9	13.6	308.8
3	2	19.1	14.7	7.66			87.5	678.8	14.7	268.5
	2	19.1	14.7				82.0	665.8	15.7	260.0
3	3	36.1	15.6	7.51			348.5	380.9	13.6	286.8
	3	36.1	15.6				105.2	346.9	17.8	225.8
4	0	0	14.1	7.55			114.8	976.7	15.7	312.5
	0	0	14.1				113.4	852.2	14.7	169.7

August 1993

Stn	depth m	Sal ppt	Temp C	pH	O2 mg/l	O2 % Sat	PO4-P ug/l	NO3-N ug/l	NO2-N ug/l	NH4-N ug/l
1	0	34.7	15.4	8.19	9.5	105	27.7	149.0	5.5	101.7
1	0	34.7	15.4				27.7	149.1	3.7	65.0
1	0.5	34.8	15.2	8.23	8.0	87	26.5	144.5	6.5	73.5
1	0.5	34.8	15.2				26.5	150.0	2.8	92.3
1	1	34.8	15.2	8.22	8.3	91	27.7	144.6	4.6	92.3
1	1	34.8	15.2				26.5	147.3	3.7	84.8
1	1.5	34.9	15.1							
1	2	34.9	15	8.14	8.0	89	29.9	149.9	4.6	68.8
1	2	34.9	15				24.3	141.1	2.8	87.6
2	0	29.04	15.9	8.14	9.0	100	123.9	253.0	4.6	126.2
2	0	29.04	15.9				128.3	256.5	4.6	118.7
2	0.5	29.1	15.9	8.13	9.0	99	126.1	252.0	7.4	101.7
2	0.5	29.1	15.9				130.5	238.8	4.6	78.2
2	1	29.1	15.9	8.14	7.4	85	127.2	256.4	6.5	91.4
2	1	29.1	15.9				127.2	259.1	5.5	106.4
2	1.5	29.1	15.9							
2	2	29.1	15.9	7.96	8.4	97	130.5	257.4	3.7	109.3
2	2	29.1	15.9				132.7	177.6	92.5	71.6
3	0	22.7	16.9	7.97	7.5	86	216.8	301.9	1.8	58.4
3	0	22.7	16.9				219.0	354.2	4.6	41.4
3	0.5	23.3	16.8	7.95	9.1	101	195.8	305.4	3.7	56.5
3	0.5	23.3	16.8				204.6	309.0	3.7	64.0
3	1	25.7	16.8	7.97	7.5	84	189.2	293.0	3.7	69.7
3	1	25.7	16.8				191.4	293.9	2.8	39.6
3	1.5	26	16.8							
3	2	26.2	16.7							
3	2.5	26.2	16.7	7.85	6.7	80	138.3	293.1	0.0	42.4
3	2.5	26.2	16.7				172.6	265.4	4.6	58.4
4	0	0	16	9.34	12.2	124	425.9	863.6	15.7	58.4
4	0	0	16				437.0	958.6	16.6	91.4

October 1993

Stn	depth m	Sal ppt	Temp C	pH	O2 mg/l	O2 % Sat	PO4-P ug/l	NO3-N ug/l	NO2-N ug/l	NH4-N ug/l
1	0	26	16.9	8.11	7.4	87	25.0	192.4	12.3	82.1
1	0	26	16.9				25.0	190.0	12.3	55.8
1	0.5	26	16.9	7.95	7.9	100	24.1	186.3	13.6	94.2
1	0.5	26	16.9				27.5	188.6	11.1	96.3
1	1	26	16.9	8.05	6.6	81	26.7	197.3	12.3	79.9
1	1	26	16.9				30.1	201.0	11.1	89.8
1	2	26	16.9	8.09	7.4	85	25.0	181.5	8.6	79.9
1	2	26	16.9				26.7	190.0	9.9	93.1
2	0	2.4	19.5	8.13	7.3	86	64.5	224.1	9.9	109.5
2	0	2.4	19.5				64.5	218.1	8.6	113.9
2	0.5	3	18.9	8.06	7.7	86	69.7	203.3	23.4	94.2
2	0.5	3	18.9				68.8	211.9	19.7	96.3
2	1	3.4	19	8.07	7.1	86	74.9	236.4	4.9	93.1
2	1	3.4	19				72.3	209.5	7.4	119.3
2	1.5	4.5	18.6							
2	2	24	17.2	7.99	7.2	86	43.9	218.1	8.6	122.6
2	2	24	17.2				48.2	214.4	7.4	115.0
2	3	26	16.9	8.08	7.3	88	40.4	221.7	9.9	93.1
2	3	26	16.9				49.9	208.3	8.6	87.6
3	0	0	20	7.7	6.6	82	74.9	131.6	4.9	74.5
3	0	0	20				86.0	131.5	7.4	89.8
3	0.5	0	19.3	7.55	6.7	78	77.4	127.9	6.2	89.8
3	0.5	0	19.3				77.4	115.7	11.1	85.4
3	1	0.02	18.7	7.52	6.9	82	77.4	141.2	12.3	78.8
3	1	0.02	18.7				76.6	130.2	23.4	96.3
3	1.5	0.05	18.6							
3	2	0.07	18.6	7.59	7.4	86	77.4	123.0	11.1	102.9
3	2	0.07	18.6				78.3	115.7	11.1	90.9
3	2.5	0.08	18.5							
4	0	0	20	7.66	7.6	90	78.3	135.1	13.6	
4	0	0	20				84.3	124.1	17.2	

December 1993

Stn	depth m	Sal ppt	Temp C	pH	O2 mg/l	O2 % Sat	PO4-P ug/l	NO3-N ug/l	NO2-N ug/l	NH4-N ug/l
1	0	32.7	21.9	8.05	6.7	87	20.5	44.4	4.4	78.4
1	0	32.7	21.9				21.0	40.1	3.5	81.0
1	0.5	32.8	21.9	7.88	7.9	99	22.3	29.6	7.1	76.6
1	0.5	32.8	21.9				21.9	98.5	6.2	72.1
1	1	32.8	21.9	8	6.5	82	21.4	39.2	6.2	80.2
1	1	32.8	21.9				24.1	33.1	7.1	83.7
1	1.5	32.9	21.9	8.02	6.4	82	26.8	34.0	4.4	77.5
1	1.5	32.9	21.9				26.4	40.1	5.3	91.7
1	2	32.9	21.9	8.02	6.7	90	23.2	40.1	5.3	94.4
1	2	32.9	21.9				26.8	41.0	2.7	98.0
2	0	7.02	24.4	7.95	6.4	75	97.8	127.3	8.8	81.9
2	0	7.02	24.4				104.1	122.1	7.1	74.8
2	0.5	7.19	24.4	7.99	7.7	94	103.6	127.3	7.1	75.7
2	0.5	7.19	24.4				106.3	122.9	9.7	88.2
2	1	7.39	24.3	7.95	6.3	75	96.0	126.4	9.7	81.0
2	1	7.39	24.3				106.3	123.8	10.6	81.0
2	1.5	7.56	24.3	7.95	6.7	79	88.0	115.9	11.5	81.0
2	1.5	7.56	24.3				108.6	109.8	10.6	80.2
2	2	26.4	22.7	8.03	7.1	88	69.7	92.4	7.1	88.2
2	2	26.4	22.7				108.1	124.6	11.5	89.1
3	0	0.47	24.5	7.55	6.6	77	138.9	104.6	12.4	81.9
3	0	0.47	24.5				138.9	107.2	11.5	84.6
3	0.5	0.48	24.5	7.75	6.5	79	137.1	92.4	10.6	90.8
3	0.5	0.48	24.5				142.5	94.1	10.6	95.3
3	1	0.49	24.5	7.63	8.5	103	143.4	102.8	10.6	98.0
3	1	0.49	24.5				136.7	108.9	11.5	101.5
3	1.5	0.52	24.5	7.57	8.9	105	142.5	108.1	7.1	103.3
3	1.5	0.52	24.5				140.7	95.9	10.6	94.4
3	2	0.62	24.5		0.0		139.4	97.6	10.6	92.6
3	2	0.62	24.5				121.1	110.7	8.0	101.5
3	2.5	0.92	24.5							
4	0	0	25.3	7.11	7.3	87	123.3	95.0	11.5	
4	0	0	25.3				123.7	101.1	8.8	

[illegible]

Stn	depth m	Sal ppt	Temp C	pH	O2 mg/l	O2 % Sat	PO4-P ug/l	NO3-N ug/l	NO2-N ug/l	NH4-N ug/l
1	0	34.4	23	7.79	7.2	101	18.0	97.0	7.4	100.3
1	0	34.4	23				17.1	58.8	5.2	97.5
1	0.5	34.6	23	7.85	7.3	99	17.6	33.9	4.4	100.3
1	0.5	34.6	23				16.7	48.8	4.4	90.2
1	1	34.6	23	7.88	6.6	99	18.4	50.2	5.2	90.2
1	1	34.6	23				17.1	53.8	5.9	98.4
2	0	21.2	25	7.73	7.5	101	80.5	46.0	5.2	39.2
2	0	21.2	25				69.8	45.3	3.7	33.7
2	0.5	21.8	24.9	7.75	6.1	85	78.4	56.7	3.0	41.0
2	0.5	21.8	24.9				80.1	55.2	4.4	48.3
2	1	22.8	24.8	7.73	6.9	96	76.6	60.2	3.7	63.8
2	1	22.8	24.8				74.9	73.0	3.7	63.8
2	1.5	26.9	24.4	7.72	5.7	76	67.7	85.9	1.5	58.3
2	1.5	26.9	24.4				65.1	85.8	3.7	55.6
2	2	27.6	24.4	7.74	7.4	100	62.9	88.6	5.2	63.8
2	2	27.6	24.4				61.7	91.5	4.4	77.5
2	3	28.9	24.3	7.78	5.9	81	51.0	89.3	4.4	73.8
2	3	28.9	24.3				51.4	87.2	4.4	83.0
3	0	10.4	26.5	7.54	5.1	68	122.0	13.4	1.5	41.0
3	0	10.4	26.5				128.0	14.2	0.7	40.1
3	0.5	17.1	25.6	7.56	5.2	75	101.9	18.4	0.7	43.8
3	0.5	17.1	25.6				102.8	16.3	0.7	49.2
3	1	18.8	25.4	7.5	7.3	102	96.8	18.4	0.7	52.0
3	1	18.8	25.4				94.2	22.7	0.7	45.6
3	1.5	19.3	25.3	7.52	7.1	104	93.3	29.1	0.7	40.1
3	1.5	19.3	25.3				94.2	29.8	0.0	52.9
3	2	19.4	25.2	7.5	6.6	88	95.1	40.4	2.2	49.2
3	2	19.4	25.2				94.6	48.3	0.7	43.8
4	0	0	25.3	8.21	9.8	123	208.1	12.7	2.2	45.6
4	0	0	25.3				213.2	9.8	3.0	43.8

April 1994

Stn	depth m	Sal ppt	Temp C	pH	O2 mg/l	O2 % Sat	PO4-P ug/l	NO3-N ug/l	NO2-N ug/l	NH4-N ug/l
1	0	34.4	19.8	7.68			10.9	82.0	6.7	97.4
1	0	34.4	19.8				10.5	55.5	2.9	99.3
1	0.5	34.5	19.9	7.97			11.3	56.4	1.9	99.3
1	0.5	34.5	19.9				10.5	58.3	0.0	101.2
1	1	34.5	19.9	7.87			11.7	51.3	4.8	98.4
1	1	34.5	19.9				10.9	56.0	0.0	98.4
1	1.5	34.5	19.8	8.05			11.7	50.4	1.0	100.2
1	1.5	34.5	19.8				10.9	55.1	1.0	96.5
1	2	34.5	19.8	7.98			13.0	49.4	1.9	137.3
1	2	34.5	19.8				12.1	55.1	1.0	77.0
2	0	26.3	21.9	7.21			62.4	111.9	4.8	74.2
2	0	26.3	21.9				65.3	93.3	4.8	72.4
2	0.5	27	21.7	7.71			61.5	108.2	3.8	86.3
2	0.5	27	21.7				62.8	104.9	4.8	102.1
2	1	27.1	21.6	7.03			55.3	105.4	6.7	98.4
2	1	27.1	21.6				63.2	112.9	3.8	86.3
3	0	18.2	23	7.53			95.4	56.4	1.9	76.1
3	0	18.2	23				96.7	44.8	1.9	67.7
3	0.5	18.3	22.9	7.54			95.4	54.5	3.8	77.0
3	0.5	18.3	22.9				97.1	48.1	1.0	80.7
3	1	19.1	23.1	7.56			92.5	48.5	2.9	85.4
3	1	19.1	23.1				94.6	52.7	1.0	79.8
3	1.5	21.5	23.1	7.5			88.3	86.8	1.9	69.6
3	1.5	21.5	23.1				87.9	70.4	1.9	71.5
3	2	22.4	22.7	7.51			85.0	83.5	2.9	82.6
3	2	22.4	22.7				86.2	76.9	4.8	83.5
3	2.5	22.9	22.6				78.7	92.8	2.9	99.3
3	2.5	22.9	22.6				83.3	90.1	1.0	95.6
3	3	22.9	22.5							
4	0	0	21.6	7.88			190.0	28.4	1.9	89.1
4	0	0	21.6				180.4	23.8	1.9	82.6

June 1994

Stn	depth m	Sal ppt	Temp C	pH	O2 mg/l	O2 % Sat	PO4-P ug/l	NO3-N ug/l	NO2-N ug/l	NH4-N ug/l
1	0	34.4	13.5				22.0	138.5	8.2	80.6
1	0	34.4	13.5				17.9	168.7	4.9	83.3
1	0.5	34.4	13.5				19.6	117.4	7.4	97.8
1	0.5	34.4	13.5				18.7	183.4	9.8	84.2
1	1	34.5	13.4				17.5	153.2	10.6	89.6
1	1	34.5	13.4				17.9	162.2	9.0	109.5
1	1.5	34.6	13.2				14.3	148.3	10.6	81.5
1	1.5	34.6	13.2				19.2	224.1	8.2	91.4
1	2	34.7	13				18.7	180.1	8.2	90.5
1	2	34.7	13				17.9	172.0	11.5	92.3
1	2.5	34.7	12.8							
1	3	34.8	12.8				15.5	127.1	12.3	105.0
1	3	34.8	12.8				16.7	134.5	9.8	86.9
2	0	28.2	13.2				69.3	121.5	3.3	59.7
2	0	28.2	13.2				71.3	115.7	4.1	60.6
2	0.5	28.4	13.2				70.5	143.5	5.7	67.9
2	0.5	28.4	13.2				82.3	103.5	6.6	70.6
2	1	28.4	13.2				71.3	114.1	5.7	60.6
2	1	28.4	13.2				85.6	142.6	6.6	59.7
2	1.5	28.5	13.2				69.7	129.6	7.4	70.6
2	1.5	28.5	13.2				65.2	114.1	3.3	71.5
2	2	28.7	13.2				69.3	120.6	9.0	68.8
2	2	28.7	13.2				70.9	140.2	6.6	68.8
2	2.5	28.7	13.2							
2	3	28.7	13.1				50.1	136.9	4.9	77.8
2	3	28.7	13.1				66.8	125.5	4.1	65.2
3	0	20.6	13.8				135.3	97.0	0.8	76.0
3	0	20.6	13.8				139.0	75.0	0.8	75.1
3	0.5	24.1	14.4				114.1	97.8	0.0	58.8
3	0.5	24.1	14.4				108.4	110.1	0.0	71.5
3	1	25.2	14.5				94.1	110.1	0.0	75.1
3	1	25.2	14.5				94.1	120.6	4.1	78.7
3	1.5	25.5	14.3				97.0	104.3	0.8	74.2
3	1.5	25.5	14.3				94.6	112.5	0.0	78.7
3	2	25.7	14.2				95.0	132.1	2.5	66.1
3	2	25.7	14.2				99.4	110.1	0.0	66.1
4	0	0	14.5				319.1	201.3	6.6	74.2
4	0	0	14.5				335.8	189.9	5.7	86.9

SWARTKOPS RIVER ESTUARY
16 August 1996
EBB TIDE

TIME	COORDINATES S	COORDINATES E	DISTANCE FROM MOUTH	STN	DEPTH m	TEMP	SAL	DO mg/l	NO2-N µg/l	NO3-N µg/l	NH3-N µg/l	PO4-P µg/l	SiO4-Si µg/l	COMMENTS
14:30:00				Sea		14.70	34.80		8.5	97.1	32.9	74.3	240.1	
11:15:00	33 51 46	25 37 47	0.3	1	4.1	12.99	34.75							Tide ebbing
					3.5	12.99	34.79							
					3.0	12.98	34.80							
					2.5	12.98	34.80							
					2.0	12.98	34.80							
					1.5	12.98	34.80							
					1.0	12.98	34.8							
					0.5	12.99	34.80							
					0.0	12.99	34.75	8.3	7.8	64.3	63.3	81.5	378.3	
11:25:00	33 51 35	25 37 25	1.3	2	3.4	13.14	34.19							
					3.0	13.12	34.30							
					2.5	13.14	34.31							
					2.0	13.07	34.18							
					1.5	13.04	34.20							
					1.0	13.06	34.18							
					0.5	13.02	34.17							
					0.0	13.03	34.08	8.0	7.8	71.8	60.8	134.9	341.9	
	33 52 01	25 36 44	2.8	3	5.4	12.84	33.17							Bait collection and
					5.0	12.84	33.17							fishing on banks
					4.5	12.85	33.15							
					4.0	12.85	33.15							
					3.5	12.86	33.17							
					3.0	12.87	33.16							
					2.0	12.88	33.13							
					1.5	12.89	33.11							
					1.0	12.92	32.96							
					0.5	13.06	32.77							
					0.0	13.07	32.50	8.2	6.0	51.2	43.8	268.7	476.5	

TIME	COORDINATES S	E	DISTANCE FROM MOUTH	STN	DEPTH m	TEMP	SAL	DO mg/l	NO2-N µg/l	NO3-N µg/l	NH3-N µg/l	PO4-P µg/l	SiO4-Si µg/l	COMMENTS
	33 51 41	25 36 13	3.9	4	1.2	12.93	31.30							
					0.8	12.95	31.27							
					0.5	13.00	31.20							
					0.0	13.13	31.01	8.9	3.4	10.3	23.1	298.5	589.3	
	33 51 18	25 35 46	4.5	5	1.8	13.38	29.52							
					1.5	13.31	29.52							
					1.0	13.30	29.51							
					0.5	13.30	29.53							
					0.0	13.31	29.40	9.9	2.1	1.0	23.1	597.0	691.1	
	33 50 23	25 35 59	6.2	6	3.0	13.03	28.31							
					2.5	13.01	28.32							
					2.0	13.03	28.27							
					1.5	13.14	28.26							
					1.0	13.22	28.19							
					0.5	13.28	28.16							
					0.0	13.40	28.08	10.6	2.3	0.1	25.6	835.8	763.9	
	33 50 21	25 35 22	7.5	7	2.1	12.82	27.96							
					1.5	12.85	27.73							
					1.0	13.59	26.64							
					0.5	13.69	26.37							
					0.0	13.71	26.36	10.0	5.4	195.9	26.8	955.2	807.5	

TIME	COORDINATES		DISTANCE FROM MOUTH	STN	DEPTH m	TEMP	SAL	DO mg/l	NO2-N µg/l	NO3-N µg/l	NH3-N µg/l	PO4-P µg/l	SiO4-Si µg/l	COMMENTS
	S	E												
	33 50 22	25 35 21	9.3	8	3.9	12.64	27.21	8.8	3.1	69.0	25.6	716.4	745.7	Benthic micro algae
					3.5	12.59	29.21							growing extensively
					3.0	12.63	28.87							on tidal flats. Good
					2.5	12.74	27.03							Zostra beds and very
					2.0	13.15	25.92							healthy salt marsh
					1.5	13.29	25.67							
					1.0	13.46	25.35							
					0.5	13.85	24.66							
					0.0	13.88	24.65	9.0	5.4	740.4	24.3	1313.4	818.4	
13:15:00	33 49 25	25 34 11	10.9	9	1.8	13.27	25.98	8.4	4.6	741.1	28.0	1253.7	818.4	
					1.5	13.39	25.29							
					1.0	13.44	24.20							
					0.5	13.90	22.90							
					0.0	13.90	22.84	7.7	6.0	1150.0	26.8	1611.9	811.2	
13:35:00	33 49 20	25 33 07	12	10	1.9	13.94	25.10	7.5	4.6	902.7	36.5	1611.9	843.9	
					1.5	13.89	24.96							
					1.0	14.00	24.67							
					0.5	14.22	23.81							
					0.0	14.58	22.55	7.4	6.2	1118.7	42.6	1611.9	793.0	
13:45:00	33 48 54	25 32 50	13.3	11	0.5	15.58	22.56							Bait collection cont.
					0.0	15.58	22.58	9.5	6.5	1106.0	38.9	1791.0	756.6	above rock sill

SWARTKOPS (UPE)

Station	depth (m)	01/06/94 time	Sal (ppt)	Temp (°C)	25/06/94 time	Sal (ppt)	Temp (°C)	29/07/94 time	Sal (ppt)	Temp (°C)
1	0	10:05	33.8	16.8	09:06	35.5	16.4	14:18	31.8	15.1
1	0.5		34.9	16.8		35.5	16.4		32.0	15.1
1	1		35.0	16.7		35.5	16.4		32.0	15.1
1	1.5		35.1	16.7		35.5	16.3		32.2	15.2
1	2		35.1	16.7		35.5	16.3		32.4	15.2
1	2.5		35.1	16.7		35.5	16.3		32.6	15.4
1	3		35.1	16.7		35.5	16.3			
1	3.5		35.1	16.7		35.5	16.3			
1	4					35.5	16.3			
2	0	10:08	35.1	16.6	09:14	35.5	15.9	14:29	29.1	15.1
2	0.5		35.1	16.6		35.5	15.9		29.3	15.0
2	1		35.1	16.6		35.5	15.9		29.4	14.8
2	1.5		33.1	16.6		35.5	15.9		30.2	14.9
2	2		35.1	16.6		35.5	15.9		30.5	14.9
2	2.5		35.1	16.6		35.5	15.9		30.9	14.9
2	3		35.1	16.6		35.5	15.9		31.5	15.2
2	3.5		35.1	16.6		35.5	15.9			
2	4					35.5	15.9			
2	4.5					35.5	15.9			
3	0	10:15	33.3	16.1	09:24	35.5	14.6	14:36	26.0	15.1
3	0.5		33.6	16.0		35.4	14.6		26.6	15.2
3	1		33.8	16.1		35.4	14.6		26.9	15.2
3	1.5		34.0	16.2		35.4	14.6			
3	2					35.4	14.6			
4	0	10:20	31.8	15.9	09:33	34.5	13.3	15:03	22.7	14.8
4	0.5		31.9	15.8		34.5	13.3		23.3	14.6
4	1		31.9	15.8		34.5	13.3		23.8	14.6
4	1.5		31.9	15.8		34.5	13.3		25.5	13.9
4	2		32.0	15.8		34.5	13.3		25.9	13.8
4	2.5		32.0	15.7		34.5	13.3		26.7	13.6
4	3								27.5	13.5
4	3.5									
5	0	10:24	29.8	15.8	09:35	31.2	12.6	15:12	22.6	14.9
5	0.5		30.7	15.5		32.2	12.5		22.8	14.8
5	1		31.1	15.5		32.5	12.9		24.2	14.3
5	1.5		31.2	15.5		32.9	12.6			
5	2		31.2	15.5		33.1	12.8			
5	2.5					33.5	13.0			
6	0	10:28	28.4	15.4	09:46	30.5	13.0	15:16	21.4	15.0
6	0.5		28.9	15.1		30.6	13.0		21.6	14.9
6	1		28.7	15.4		30.8	13.1		21.7	14.9
6	1.5		28.8	15.4		31.0	13.3		24.7	14.1
6	2		28.9	15.4		31.4	13.3		25.1	13.1
6	2.5		28.9	15.4		31.6	13.3		25.2	13.8
6	3		28.9	15.4		32.2	13.1		25.2	13.8
6	3.5					32.4	13.1			
6	4					32.5	13.1			
6	4.5					32.8	13.1			
6	5					32.6	13.1			
7	0	10:32	22.2	15.1	09:55	25.8	12.4	15:24	19.4	15.6
7	0.5		23.9	15.0		26.4	12.5		20.0	15.3
7	1		27.2	15.3		28.8	13.0		21.0	15.3
7	1.5		27.7	15.4		29.6	13.4		22.7	14.9
7	2					30.1	13.5		23.3	14.8
7	2.5					30.3	13.6			
7	3					30.3	13.6			
8	0	10:38	22.2	15.7	10:00	24.4	12.1	15:27	18.7	16.0
8	0.5		23.9	15.2		25.1	12.6		19.4	15.7
8	1		26.1	15.2		28.2	13.5		22.5	14.9
8	1.5		26.9	15.2		28.3	13.3		23.3	14.9
8	2		27.1	15.2		28.6	13.3		23.5	14.8
8	2.5		27.2	15.2		29.3	13.4			
8	3		27.2	15.2						

SWARTKOPS (UPE)

Station	depth (m)	01/06/94 time	Sal (ppt)	Temp (°C)	28/06/94 time	Sal (ppt)	Temp (°C)	29/07/94 time	Sal (ppt)	Temp (°C)
9	0	10:44	24.1	15.9	10:07	23.9	12.8	15:37	15.1	17.1
9	0.5		25.4	15.4		24.3	12.9		20.9	15.6
9	1		25.7	15.2		26.5	13.6		23.0	15.3
9	1.5		26.0	15.1		26.9	13.7		23.3	15.2
9	2		26.3	15.1		27.6	13.9		23.7	15.2
9	2.5					27.7	13.9			
9	3					27.9	14.0			
10	0	11:40	5.6	17.1	10:23	21.2	13.5	16:40	4.1	16.5
10	0.5		22.3	16.4		25.4	14.0		18.5	16.5
10	1		22.7	16.2		26.0	13.9			
10	1.5					26.4	13.9			
10	2					26.4	13.9			
Tide			neap high slack			neap-2 high slack			neap-1 low slack	
Freshwater inflow (m3/sec)			0.264			0.290			0.562	

E. coli data collected by the DWA&F, Eastern Cape at Perseverance Drift
(Von der Molen, pers. comm.)

DATE	COUNTS per 100 ml
28 June 1993	100
28 July 1993	0
31 August 1993	16
14 September 1993	0
6 October 1993	1000000
23 November 1993	26
8 December 1993	184
18 January 1994	26
25 February 1994	34
23 March 1994	3
8 April 1994	21
17 May 1994	34
30 June 1994	25
29 July 1994	14
12 August 1994	0
22 September 1994	22
21 October 1994	78
25 November 1994	47
9 December 1994	105
20 January 1995	1000000
17 Feb 1995	40
23 March 1995	10
6 April 1995	10
26 May 1995	117
30 June 1995	230
31 July 1995	-
30 August 1995	6
28 September 1995	30