THE EFFECT OF LAND USE ON GAMTOOS

ESTUARY WATER QUALITY

by

M W PEARCE and E H SCHUMANN

Report to the Water Research Commission

by the

Department of Geology (Oceanography)

University of Port Elizabeth, P O Box 1600, Port Elizabeth, 6000

South Africa

Head of Department: Dr R W Shone

Project Leader: Dr E H Schumann

This report originated from a PhD thesis entitled

Surface- and Ground-Water Hydrology of the Gamtoos Estuary and Floodplain

as submitted in the Faculty of Science at the University of Port Elizabeth

WRC Report No. 503/1/97

ISBN 1 86845 264 6

EXECUTIVE SUMMARY

Introduction

This project was initiated because of the reduction in freshwater flow in most South African rivers as a result of dams being built in the catchment regions, and abstraction of water for agricultural, industrial and domestic purposes. In particular, it was recognised that estuaries could be severely affected by this limitation in freshwater inflow in three different but interrelated ways, namely:

- The limitation of floods means that, particularly in flood-dominant estuaries, marine sediments can enter the mouth and eventually cause closure. This would limit ocean-estuary exchanges, or in the extreme case, completely stop such exchanges.
- The runoff from agricultural lands, including pesticides and fertilizers, may be accumulating in rivers and estuaries as a result of limited flushing, both because of reduction in freshwater flow and because of reduced ocean-estuary exchanges.
- It is known that many of the estuarine biota are dependent on freshwater inflow, and are particularly sensitive to the freshwater-saltwater interface.

There have been investigations, albeit limited, into the sedimentation of estuaries. Moreover, several institutions have ongoing projects analysing various aspects of the implications of a reduction of freshwater flow on estuarine biota. This project was therefore designed to investigate specifically the input of potential pollutants in agricultural runoff into an estuary, and whether the flushing processes in the estuary itself were sufficient to inhibit any build-up of such pollutants. Groundwater processes are important in these exchanges, and monitoring rest water levels and water analyses formed an important component of the project.

Objectives

In detail, the aims of the project were to:

a) Investigate the groundwater dynamics and subsurface drainage patterns, in relation to geological structures and the effects of rain, irrigation water and tidal variations in the adjacent estuary, as well as aspects such as vegetation cover, wind and solar radiation (temperature).

b) Estimate the input of fertilisers, herbicides and pesticides on the agricultural lands, and assess the resultant input of chemical products into the estuary.

c) Investigate estuarine processes such as tidal action, water stratification and the influence of freshwater input, particularly in terms of mixing and removal of chemical products, and possible effects on water quality and biota.

The Gamtoos estuary was chosen for the following reasons:

- It has an established intensive agriculture area (in newly founded irrigation areas a period of stabilisation should be allowed before the details of such processes can be determined).
- The tidal reach is substantial for South African estuarine systems (about 20 km upstream of the mouth), and this part of the estuary is an important recreational and angling facility.
- Ease of access.

Methodology

The Gamtoos estuary has an extensive floodplain, and a section was chosen for assessing the groundwater component; within this section several groups of boreholes were established. Moreover, an agricultural drainage system provided an easily sampled runoff to the estuary. Salinity, temperature and dissolved oxygen (DO) could be sampled along the whole tidal region of the estuary using a small boat, while continuously recording instrumentation allowed time series of water levels and some current structures to be obtained. Borehole monitoring involved measuring rest water levels (RWLs) and taking water samples on a regular basis. Water analyses included the determination of electrical conductivity, cations: calcium (Ca^{++}), magnesium (Mg^{++}), sodium (Na^{+}), potassium (K^{+}) and anions: chloride (Cl^{-}), and sulphate (SO_4), as well as nitrate (as N), nitrite (as N) and total phosphorus (as P). Pesticide analysis is very expensive, and only six samples were analysed using a broad screening method - the cholinesterase inhibition test.

The experimental programme started in August 1992, and continued through to February 1994, though some monitoring extended to the end of 1994. Monthly measurements were made at the boreholes, and there were intensive measurement periods involving estuarine and borehole sampling in November/December 1992, March/April 1993, and in June and November 1993; essentially these periods were chosen to cover all the seasons. Because of the known differences in estuarine structures over spring and neap tides, these intensive measurements were designed to cover a spring-neap tidal cycle.

Summary of Findings

Quite clearly it was not possible in such a short experimental period to accommodate longer-period variability, particularly in terms of sedimentation of the Gamtoos estuary mouth, and any drought and wet years. As it happened, the Gamtoos mouth was severely constricted by flood-tidal deltas, and tidal heights within the estuary were only about 30% of those in the adjacent ocean. In terms of rainfall, the seven months March to September 1992, prior to the first intensive measurement period, were particularly dry and the farmers were on reduced irrigation quotas. During the main study period in 1993 rainfall was above average, and on 12 June a record maximum quantity of rain fell in a 24 hour period. In 1994 conditions were closer to the long-term mean. The results presented here therefore need to be appraised in the context of these interannual constraints.

Rainfall in the region is bimodal, with maxima generally occurring in the autumn and spring. Evaporation losses are high in the Gamtoos valley and during the summer months there are moisture deficits with consequent decreases in groundwater recharge.

RWLs at all the borehole site were well above the water levels in the estuary, and no detectable variations could be ascertained in response to tidal fluctuations in the estuary. In the experimental area borehole RWLs generally responded within a few days to both rainfall events and irrigation. The hydraulic gradient between most of the boreholes was very slight, and the constantly changing surface inputs coupled with variable recharge rates meant that the groundwater exhibited highly dynamic and variable subsurface flow. On a longer term basis, the water table throughout the study area was at its lowest at the end of 1992. With the increased rainfall in 1993 the level of the water table rose, reaching a maximum immediately after the heavy June rainfall. During the latter six months of 1993 the water table was generally higher than before June, but by early 1994 a declining trend was evident.

It was not possible to obtain information on the application of fertilizers, herbicides and pesticides from the farmers in the area because they feared reprisals. However, inputs to the estuary from surface runoff, effluent from the surface agricultural system, discharge from the subsurface drainage system and groundwater inputs could be analysed. In particular, the discharge from the drainage pipe gave an indication of the chemical inputs to the estuary. Results from this discharge show that the concentrations of nitrate-N, sulphate and potassium were frequently present at very high levels, while the nitrite-N content often exceeded the recommended limit of 0.06 $mg.l^{-1}$, and all samples had total phosphorus levels above that recommended for the protection of aquatic life. In addition to these nutrient inputs, the pesticide screening test yielded positive results in all of the samples analysed.

Groundwater samples obtained from the boreholes varied in quality, though patterns emerged with certain areas having higher nutrient concentrations; consequently the groundwater discharge to the estuary can also be expected to vary spatially and temporally. Similarities were found with the drainage discharge and the quality of water from a seepage into the estuary, and this enabled rough estimates to be obtained within broad ranges of the amount of nutrients entering the estuary on an annual basis.

The nutrient content of the water in the Gamtoos estuary was generally low, but from time to time certain nutrient levels, in particular phosphorus, exceeded the limits recommended for the protection of marine life. Such elevated values were found more frequently in the upper estuary than in the lower reaches. Hypoxic conditions were also observed on occasion in the bottom waters, in particular after heavy rainfall, and were attributed to decomposition of organic debris washed into the estuary by the increased freshwater inflow. Salinity stratification inhibits mixing with more oxygenated surface water, and therefore favours such low dissolved oxygen conditions in the bottom waters. Deeper scour holes in the upper estuary were also found to act as sinks for decomposing organic matter, with more frequent occurrence of hypoxic conditions.

As expected, substantial differences were found in the estuarine salinity structures during spring and neap tides. Thus the lower reaches were mostly well mixed during spring high tides by the input of sea water through the tidal inlet, with the concurrent flushing out occurring at the following low tides. The input of sea water at neap tides was limited, and well-stratified conditions were found. Because of the narrow estuary channel wind was generally not an important mixing agent, through in certain sections orientated parallel to the wind reasonable waves could be generated at times.

Solar radiation caused marked changes in temperature of the upper layers of the water in the estuary, particularly in summer. On occasion cold water was also observed entering from the sea, probably as a result of wind-generated upwelling events. However, the dominant influences on water density were the salinity differences between freshwater and saltwater.

The average freshwater input at the head of the Gamtoos estuary was estimated at less than 1 $m^3.s^{-1}$. Pulses of freshwater emanating from rainfall events caused substantial increases in this flow, but measurements of volume flow were difficult to make. Inflows in excess of 1 $m^3.s^{-1}$ effectively flushed out much of the upper reaches of the estuary, but in the lower 14 km the cross-section broadens and deepens and the freshwater tended to exit in the surface layers.

Conclusions

The conclusion reached in this short investigation is that at present the Gamtoos estuary is in a healthy condition, and while there were times when water quality exceeded specified limits, the system is resilient and water quality was restored within a few days. However, there are indications that problems could be experienced in future and that there is the potential for the shorter periods of low quality to persist, with adverse implications for the management of the system.

There is a substantial input of pollutants from the land into the estuary. It will be difficult to reduce the application of fertilizers and pesticides to the agricultural lands, and nor can the discharge of irrigation return flow easily be avoided. It is therefore essential that the flushing capability of the estuary be maintained, and this is effected by both the input of freshwater at the head of the estuary and the tidal exchanges at the mouth. It should be noted that many biota are dependent on a freshwater input, and that the Gamtoos is an important recreational and angling region where diverse species should be maintained.

At present there is no means of monitoring the freshwater inflow, since there is no suitable gauging weir. Most of the problem conditions occurred in the upper reaches where tidal flushing was limited, with the implication that freshwater inflow is necessary to remove pollutants further downstream to where tidal action can abstract them into the ocean. To further facilitate the process, future drainage systems should discharge as far downstream in the estuary as possible.

Finally, the ocean-estuary exchanges through the tidal inlet must not be allowed to decrease further. There are indications that the flood-tidal deltas are closing the estuary mouth, and it is likely that eutrophication problems could arise if nutrients cannot be exported from the system; with the limited pesticide sampling it is at this stage not possible to give an assessment of this input. In the past the mouth has been scoured open by regular floods, but in 1993 not even the heavy rains in June were sufficient to cause any substantial opening of the mouth. It may then be that the mouth will have to be opened mechanically, but with due cognisance of the biological requirements of such an event.

The general results presented here should also be applicable to other estuaries where intensive agriculture is practised in the floodplain with a concomitant input of fertilizers and pesticides. However, it is important to realise the differences that do exist, and each system must be considered on its own merits.

ACKNOWLEDGEMENTS

The research in this report emanated from a project funded by the Water Research Commission and entitled:

'The effect of land use on Gamtoos estuary water quality'.

The financing of the project by the Water Research Commission and the contribution of Mr M Du Plessis, who was responsible for this project, is acknowledged gratefully.

The project was only possible with the co-operation of many individuals and institutions. The authors therefore wish to record their sincere thanks to the following:

The farmers in the area being investigated proved to be very co-operative in allowing boreholes to be installed and measurements to be made in their agricultural fields. Thanks are due to Messrs M Snyman, A Moolman, I du Preez, J Viljoen, W Williams, S Clark and J Reyneker.

Mr P Joubert of the Gamtoos Irrigation Board and Mr A Botha of the Department of Agriculture, Port Elizabeth for access to literature and maps.

The students who went out in a small boat at all times and in all sorts of weather conditions to make some of the measurements.

Dr R W Shone, Department of Geology, University of Port Elizabeth for valuable comments on the document.

Mr G Devey and the Scientific Services Division, City Engineers Department, of the Port Elizabeth Municipality, for the analysis of water samples. Dr A de Kock and the PETCRU staff for the pesticide analyses.

Mr J Churchill, Department of Computer Science, Vista University for assistance with computing problems.

Ms W Grundlingh for a portion of the cartographic work and Mrs J Pearce of the Department of Geography, University of Port Elizabeth.

The Weather Bureau, Pretoria for weather data.

Whilst the Water Research Commission provided the bulk of the funding for this project. Some funds were also forthcoming from the Foundation for Research Development core funding programme.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1. INTRODUCTION

The presence of the Kouga dam on the Gamtoos River has greatly reduced the natural freshwater flow downstream in the river. Such a reduction in flow, together with the limited possibility of floods flushing out the flood-tidal deltas in the estuary mouth, may have severe consequences for the structure and functioning of the river and estuary. Moreover, intensive cultivation is characteristic of the upper reaches of the estuarine floodplain, with water for irrigation supplied directly from the dam. With the natural flow thus limited, anthropogenic influences will be exacerbated, and it was these factors that prompted this broadly-based hydrological investigation into the structure and functioning of the Gamtoos estuary. This project was aimed specifically at examining the influence of agricultural practices in a small section of the floodplain on the estuary and examines the land-based inputs to the estuary via surface runoff or subsurface throughflow.

Whilst certain aspects of this project have been investigated in other rivers and estuaries throughout South Africa, a project of this nature with particular reference to agricultural inputs has not been undertaken before in the Gamtoos estuary.

In detail, the aims of the project were to:

a) Investigate the groundwater dynamics and subsurface drainage patterns, in relation to geological structures and the effects of rain, irrigation water and tidal variations in the adjacent estuary, as well as aspects such as vegetation cover, wind and solar radiation (temperature).

b) Estimate the input of fertilisers, herbicides and pesticides on the agricultural lands, and assess the resultant input of chemical products into the estuary.

c) Investigate estuarine processes such as tidal action, water stratification and the influence of freshwater input, particularly in terms of mixing and removal of chemical products, and possible effects on water quality and biota.

The Gamtoos estuary was chosen for the following reasons:

- It has an established intensive agriculture area (in newly founded irrigation areas a period of stabilisation should be allowed before the details of such processes can be determined).

- The tidal reach is substantial for South African estuarine systems (about 20 km upstream of the mouth), and the estuary is a recreational and angling facility.
- Ease of access.

Report Structure

The contents of this report follow accepted structures. The literature survey analyses the results of relevant investigations elsewhere, and assesses the conclusions applicable to this study. Moreover, the experimental region of the Gamtoos estuary is described, and details of climate, ocean-estuary exchanges, geology and irrigation practices are given.

The sampling programme, field and laboratory techniques are given in chapter 3. The latter part of chapter 3 outlines data processing and data presentation techniques.

A large portion of the investigation examined the physical properties and tidal dynamics of the estuary. These results are presented in chapter 4 together with that of the irrigation water and the surface agricultural drainage system.

In chapter 5 the groundwater quality and subsurface drainage patterns in terms of flow paths and response to irrigation and rainfall are presented. While there is some discussion and clarification interjected in the results sections of chapters 4 and 5, the main findings and trends are dealt with in chapter 6. The results obtained are assessed in terms of the study objectives given in chapter 1. Management implications are presented, and recommendations are made for future research.

2. LITERATURE SURVEY AND THE STUDY AREA

2.1 Literature Survey

The literature review to follow makes reference to both international and South African sources, however it is important to appreciate that, compared to most systems analysed elsewhere in the world, the input of freshwater into South African estuaries is very limited, and has been further reduced by the building of dams in many river catchment areas. It is also important to recognise that marked interannual variability occurs in rainfall (Tyson, 1986), and any investigation needs to be put into this longer-term perspective. Nonetheless, most estuaries in the Eastern Cape can be considered as low-inflow systems (Reddering, 1988; Whitfield and Bruton, 1989), with typically a mean freshwater input of 1 $m^3.s^{-1}$ or less (MacKay, 1993; Jerling and Wooldridge, 1994). As the Gamtoos study was a multi-disciplinary investigation a broad, comprehensive literature review was conducted. A similar international study is discussed first. It was found that there was no single similar study conducted elsewhere in South Africa. Thus the remainder of the literature review focuses on the various aspects which form components of this study, separately under sub-headings. In the section on estuarine hydrodynamics (circulation, stratification and mixing), the literature review is presented in a theoretical context.

An investigation with some aspects in common with the Gamtoos study was conducted by Staver *et al.*, (1996). They investigated the spatial and temporal patterns of nutrient inputs into the Choptank River estuary. It is a well-mixed, shallow estuary (less than 3 m deep), with 29% of the catchment area forested and 66% agricultural. Agricultural practices were identified as the main source of groundwater contamination (high nitrate content). Nitrate inputs to the estuary occurred primarily through groundwater discharge whereas phosphorus inputs occurred via surface runoff during rainfall. A higher input of nitrogen occurred from well-drained soils as opposed to the poorly-drained regions. Furthermore, they found that the effect of reduced freshwater flow to the estuary during summer, coupled with a high rate of evaporation and a decline in the flushing rate caused an accumulation of nutrients in the upper estuary. Where there was little salinity stratification this reduced the possibility of downstream transport of the nutrients via a well developed two-layered flow system. Consequently the water quality of the lower estuary was not adversely affected by the nutrient build-up in the upper estuary. During periods of high freshwater discharge, the nutrients entering the upper estuary, were however transported downstream.

Agriculture and Groundwater Quality

The influence of agricultural irrigation on groundwater quality was the subject of extensive research in the USA in the 1980s, and many of these studies examined both nutrients and pesticides in groundwater. Schmidt and Sherman (1987) analysed irrigated areas at seventy-one sites in coastal counties and thirty-six sites in inland counties of the San Joaquin valley representing a variety of soil and crop types, and corresponding fertilizer and water use. Their results showed that irrigation return flow exerted a substantial impact on groundwater quality (high nitrate content) and there was extensive pollution of groundwater caused by the pesticide DBCP (di-bromochloropropane). Where soils were sandy with the water table near the ground surface, DBCP and other pesticides were detected in the groundwater.

Sabol *et al.*, (1987) studied irrigation effects in Arizona and New Mexico and concluded that the extent and time factor of changes in groundwater quality was a function of irrigation practice, quality of the irrigation water, fertilizer and pesticide applications, rate of change in groundwater levels and soil salinity. Hallberg (1986) examined nitrates and pesticides in groundwater and found that nitrates moving into groundwater were in relative equilibrium with water flux whereas the concentration of certain pesticides increased regardless of water flux, and were thus not an artifact of climatic conditions. Tredoux (1993) provides a comprehensive literature review with regard to nitrate in groundwater. Amongst other factors, fertilizer and manure application to land is cited as a problem. A study undertaken by Konikow and Person (1985) on the Arkansas River, Texas, examined changes in salinity of groundwater and surface water in response to irrigation but had no estuarine component.

A study undertaken in South Africa by a team of researchers at the Hydrological Research Unit (now the Institute for Water Research at Rhodes University) from 1987 to 1992 (Herold, 1996) investigated groundwater quality and salinization of the Coerney River (a tributary of the Sundays River) as a result of intensive agriculture. Herolds research differed from this Gamtoos study in that little attention was given to the estuary.

Pesticides

International literature on pesticide case studies shows both positive and negative results. Following a national (USA) assessment of groundwater contamination by fertilizers and pesticides, Fairchild (1987) concluded that twenty-six states experienced some degree of groundwater contamination by pesticides despite adequate legislative awareness. Mayers and Elkins (1990) tested for fourteen pesticides in water and sediments in Padilla Bay, Washington, and found no ecologically significant levels of any pesticide. In Wye River, a tributary of Chesapeake Bay, Glotfelty *et al.*, (1984) found that 2 to 3% of atrazine applied to fields moved into the estuary. This value did however, decrease during low runoff periods. During the eighteen month sampling of rain and river water in the Granta River chalk catchment in the UK, Gomme *et al.*, (1991) found a higher concentration of pesticides in river water during flood and high flow conditions although pesticides were persistently found during low flow conditions. Pesticides were also found in rain water samples as a result of aerial spraying.

In the analysis of sediments and soils associated with two estuaries on Oahu, Green *et al.*, (1977) found that two herbicides, atrazine and ametryn underwent degradation in the soil and thus did not enter the coastal waters in detectable quantities. A third herbicide (diuran), however, did not undergo such degradation and was present in almost all the estuarine samples taken in West Loch and Kaiaka Bay as well as their influent streams. The authors cited storm runoff from fields, irrigation tail waters, excess irrigation ditch water and contamination from herbicide mixing areas as the means by which the herbicide could have entered the estuaries.

Gilliom *et al.*,(1985) discussed the general characteristics and trends of pesticide use and occurrence over the period 1975 to 1980 at one hundred and fifty river sites across the USA. They found that although chlorinated hydrocarbons had decreased in both water and bed material of the rivers monitored, there were no clear trends with regard to organophosphates. They also noted that the types of pesticides used were constantly changing and new chemicals were being introduced which created sampling and analysis problems.

In South Africa, Weaver (1993) investigated pesticide levels in groundwater in the Hex River valley, and showed that none of the fourteen pesticides tested for were detected. Work done by Weaver also analysed the concentration of atrazine in the soil profile and groundwater in the Vaalharts irrigation area; this also yielded no detectable levels of the pesticide, and it was concluded that either the atrazine had degraded, or had been flushed out of the soil profile by percolating irrigation water (White, 1989).

Groundwater Inputs into Estuaries

Depending on the volumes and chemical characteristics, surface and subsurface discharges can impact on the structure and functioning of the receiving aquatic environment (Baker and Horton, 1990; Reay *et al.*, 1992; Nuttle and Harvey, 1995). Johannes (1980) examined the ecological significance of subsurface groundwater discharge from an unconfined aquifer between Swan River and Gingin Brook, western Australia. The data collected by Johannes indicated that subsurface groundwater discharge delivered several times as much nitrate to adjacent waters as did river runoff. Bokuniewicz (1980) states that groundwater discharge accounted for between 10 and 20% of the freshwater input to Great South Bay, New York. Capone and Bautista (1985) working in the same area maintained that the rate of subsurface discharge was a function of the hydraulic gradient as determined by groundwater recharge from rainfall. Furthermore, because a good correlation was obtained between interstitial nitrate content and rainfall (when considered over a thirty day period), they concluded that the nitrate-charged groundwater input was an important source of nitrate to the adjacent aquatic environment. Similarly, Staver and Brinsfield (1996) found that the subsurface discharge of groundwater with a high nitrate content, as caused by agricultural activities, accounted for the high rates of nitrogen delivery to the Wye River estuary. Short-term fluctuations in groundwater discharge were attributed to tidal fluctuations, with seasonal variations a function of groundwater recharge.

Estuarine Circulation

To determine the impact which land-based activities and inputs have on an estuary, it is necessary to first identify the physical properties, mixing characteristics and residence times in the estuary. When ocean water enters an estuary, it moves as a wedge of dense, saltwater cutting beneath the less dense freshwater flowing seaward. This movement of seawater inland along the bottom, whilst freshwater leaves the estuary as surface flow is known as estuarine circulation. Some mixing occurs when the outward motion of surface waters results in the entrainment of underlying saline water with the simultaneous entrainment of less saline water by the intruding bottom water (Schumann *et al.*, 1997). In a system such as the Gamtoos estuary, which receives agricultural inputs, it is important that nutrients do not accumulate. The extent of estuarine circulation determines residence times in the estuary.

Ocean-Estuary Exchanges

Estuarine stratification and circulation are dependent on the nature of the exchanges that occur with the ocean through the mouth. Ocean-estuary exchanges are essentially dependent on the configuration of the estuary mouth, and the relative water level variations in both the estuary and the ocean. In South Africa estuaries are mainly flood-dominant in nature (Schumann *et al.*, 1997). The flood-dominant nature of estuaries may result in the build-up of flood-tidal deltas in the mouth region, which in extreme situations can lead to complete closure of the tidal inlet (Reddering and Esterhuizen, 1984). The restriction of the tidal influence to the inlet channels of the Nichupte Lagoon system, Mexico, with little tidal mixing within the system led to a long residence time. The model applied by Merino *et al.*, (1990) computed a flushing time of 1.3 years.

Coastal trapped waves (CTWs), generated by the alongshore component of wind stress, can play a very important part in the water exchanges with the ocean through the estuary mouth. It is known that they can have amplitudes in excess of 0.5 m (Schumann and Brink, 1990), and with typical periods of the order of ten days, they can influence the background water level for the tidal variations.

Estuarine Stratification and Mixing

Stratification is set up when the lower density freshwater flows over the saline water entering the estuary through the tidal inlet or mouth. Where a sharp vertical density gradient exists, which dominates the longitudinal gradient conditions are described as highly stratified. Vertical stratification initiates subsequent circulation patterns and mixing depends on the current shear set up between the two layers, and on winds (Schumann *et al.*, 1997). Partially mixed estuaries exhibit differences in density between surface and bottom waters but the change is less structured, and these systems have relatively weak vertical mixing (Jay and Smith, 1990).

The salinity, temperature and density structures set up within an estuary are dependent on the freshwater input, the tidal input, and the mixing processes that take place (Dardeau *et al.*, 1992). Thus the stratification within an estuary may vary depending on the dominant influences at that time. For example, Largier (1986) and Largier and Slinger (1991) analysed conditions in the Palmiet estuary, an estuary less than 2 km long in the Western Cape, where stratified conditions were found, though during the winter rainy season the freshwater input was enough to flush out all the saltwater. Schroeder *et al.*, (1990a) found that stratified conditions within Mobile Bay were broken down by strong tidal currents in conjunction with wave activity. MacKay and Schumann (1990) found that the Sundays estuary, Eastern Cape was partially stratified with a strong spring-neap tidal dependence, primarily because of the stronger currents at spring tides. The tidal flow was identified as the dominant cause of mixing in the estuary.

In well-mixed estuaries, vertically homogeneous conditions exist, with the longitudinal density gradient dominating the vertical gradient. Well-mixed conditions may arise as a result of intense tidal mixing coupled with a lack of freshwater input or as a result of wind and wave induced mixing. The Delaware estuary can be categorised as a weakly stratified or well mixed estuary, with a vertical salinity gradient generally below 3. The estuary exhibits a weak response to river discharge, which is caused by both the action of vertical shear flow in a tidally driven system and lateral shear coupled to the strong lateral salinity gradient (Garvine *et al.*, 1992; Wong, 1995). Reddy and Rao (1994) found that during the hot, dry season (March to May) the Gautami-Godavari estuary exhibited a high salinity with little vertical gradient as a result of intense tidal mixing and a decreasing freshwater discharge.

Wind-Induced Mixing

Wind-induced circulation or circulation resulting from wind stress may result in vertically homogeneous conditions, particularly in shallow estuaries. Schroeder *et al.*, (1990b) draw attention to the impact of wind forcing on subtidal exchanges within Weeks Bay, Alabama. During certain periods wind stress exerted a dominant influence in the exchange of water between the estuary and the adjacent shelf. During periods of strong river discharge and weak winds, however, runoff was the controlling factor in estuarine structures. Goodrich *et al.*, (1987) described the wind-induced mixing that frequently occurred over large areas of Chesapeake Bay. The resultant destratification favoured reoxygenation of bottom waters and a vertical flux of nutrients. Griffin and LeBlond (1990) state that although tidal mixing was the main factor controlling estuary-ocean exchanges in the Strait of Georgia-Juan de Fuca Strait estuarine system, there were also periods of wind-induced mixing at neap tides.

Nutrient Build-up in Estuaries

Where there is limited mixing or long residence times, pollutants or nutrients entering an estuary may persist for some time in the aquatic environment. A resultant build-up in nutrients could culminate in the excessive growth of nuisance plants. Fichez *et al.*, (1992) report on an algal bloom in the shallow, well-mixed Great Ouse estuary. Baker and Horton (1990) cited agricultural runoff as the main culprit introducing large quantities of nitrogen and phosphorus into Chesapeake Bay, causing excessive growth of floating plants, which in turn led to low oxygen in the bottom waters when the plants eventually decomposed. Similarly Birch (1982) reports that the excessive growth of benthic algae in the shallow Peel-Harvey estuary, western Australia, was positively correlated with both the rate of fertilizer application in neighbouring fields and soil type.

Hypoxic Conditions in Rivers and Estuaries

The low oxygen conditions and algal blooms caused by excessive inputs of agricultural related nutrients into the Chowan River, a tidal river in north-east Carolina was highlighted in the investigation by Duda (1983). Agricultural drainage improvements including tile drains and surface drainage systems were identified as being responsible for the increased nutrient input to the coastal waters. Reyes and Merino (1991) report on the low oxygen conditions in the shallow, well-mixed Borjorquez Lagoon, Mexico. They found normal diurnal fluctuations in oxygen levels related to biological activity but the lagoon experienced periods of low dissolved oxygen following the introduction and resuspension of organic matter. Kuo and Neilson (1987) provide a conceptual model of a dissolved oxygen budget. Among those factors thought to replenish oxygen levels in a partially mixed estuary are vertical transport and longitudinal advective transport.

Temperature Structures

Temperature structures (insolation and radiation losses or gains) within an estuary are determined by the characteristics of the influent water and also conduction, condensation, evaporation and rain. It may happen that the temperature of the riverine input to an estuary is not much different from that of the estuary or the adjacent tidal input as occurred during the 1973 flooding of Mobile Bay (Schroeder, 1977). Upwelling is known to be a regular occurrence along the Eastern Cape coast, particularly in summer when easterly winds are more common (Schumann *et al.*, 1988). Temperature drops of 8 to 10^{0}C can occur in periods of hours, and such cold water can then also penetrate into the estuaries: Schumann and de Meillon (1993) traced such cold water input into the St. Francis Bay marina, off the Kromme estuary some 30 km west of the Gamtoos River mouth.

Specific aspects of some studies mentioned in the literature survey, together with additional references are discussed in the appropriate sections of subsequent chapters.

2.2 Experimental Area

2.2.1 Location

The Gamtoos estuary and floodplain are situated 55 km west of Port Elizabeth on the Eastern Cape coast between latitude 33^0 58' south and longitude 25^0 01' east (Figure 2.1). The Kouga dam is located at the confluence of the Kouga and Baviaanskloof Rivers. Below the dam the Kouga River flows for approximately 1 km where it meets the Groot River, thereafter the river is called the Gamtoos River. The Gamtoos River extends approximately 75 km to the point of discharge into the sea. The Gamtoos estuary is tidal for approximately 20 km inland of the inlet. The total catchment area is 34 438 km^2 (Midgley *et al.*, 1994c). A minor tributary, the Loeriespruit enters the Gamtoos estuary 8.5 km from the mouth.

The main study area (Figure 2.2) comprised a 18 km stretch of the estuary extending from inside the flood-tidal deltas to Boschhoek railway bridge (the upper tidal limit). On occasions readings were extended up the Loeriespruit to give a broader understanding of the functioning of the estuary and its surrounds.

2.2.2 Climate

The Gamtoos Region has a moderate climate with rainfall occurring mainly in spring and autumn (Heydorn and Tinley, 1980). The annual average rainfall is approximately 400 mm and annual evaporation approximately 1 400 mm.

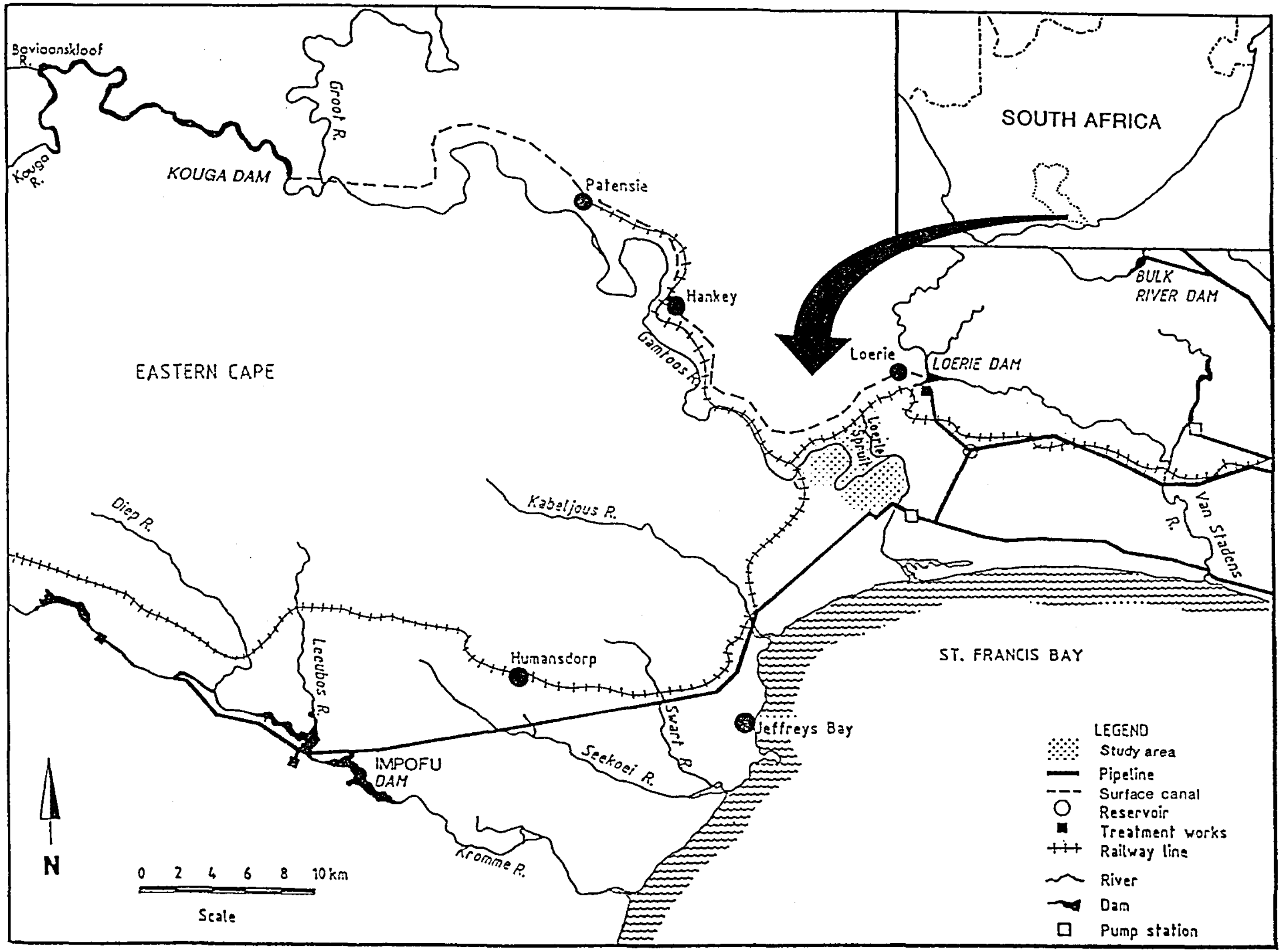

Figure 2.1 River impoundments, the Gamtoos irrigation canal and study area in the Eastern Cape region. Literature throughout the report makes reference to various rivers in this region (Port Elizabeth Municipality).

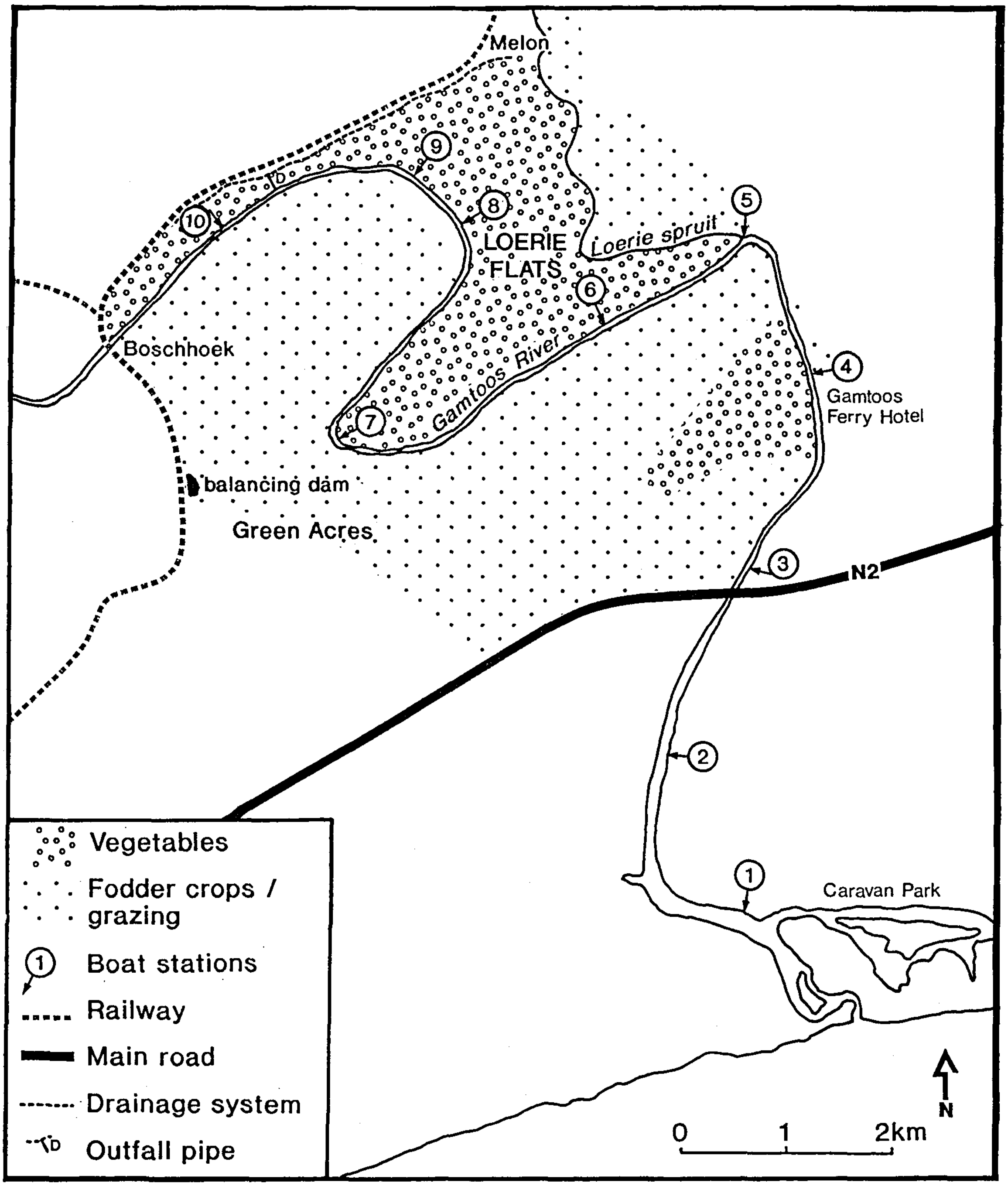

Figure 2.2 Details of the Gamtoos estuary investigation area, including sampling stations (encircled), agricultural drainage system, the drainage pipe, land use, and names referred to in the text.

Average maximum air temperatures in the region are 26^{0}C in January and 19^{0}C in July, with extremes reaching 42^{0}C and 32^{0}C respectively. Average minimum air temperatures are 15^{0}C in January and 7^{0}C in July (Schulze, 1986).

Defining a flood as an event during which the river waters overtopped their banks or the stream channel, records show that flooding of the Gamtoos River occurred in October 1847, October 1867, October 1905, October 1916, December 1931, March 1961, August 1970, March 1981 (Heydorn and Grindley, 1981), July 1982 (Midgley *et al.*, 1994b) and also occurred in 1993. A number of these floods pre-date the building of the Kouga dam, which was completed in 1964.

2.2.3 Estuary Mouth Dynamics and Exchanges

Figure 2.3 shows a substantial constriction by the flood-tidal deltas at the beginning of the project in August 1992. The periodic floods play an important part in the mouth dynamics of the Gamtoos estuary. It is a flood-dominant system, which means that marine sediments are brought into the estuary through tidal and wave action, and can build up extensive flood-tidal deltas. These deltas constrict the mouth area, and can lead to closure. Moreover, a longshore drift is responsible at times for an eastward migration of the mouth of up to 4 km (Reddering and Scarr, 1990a). Floods then scour out the marine sediments, and can breach an established

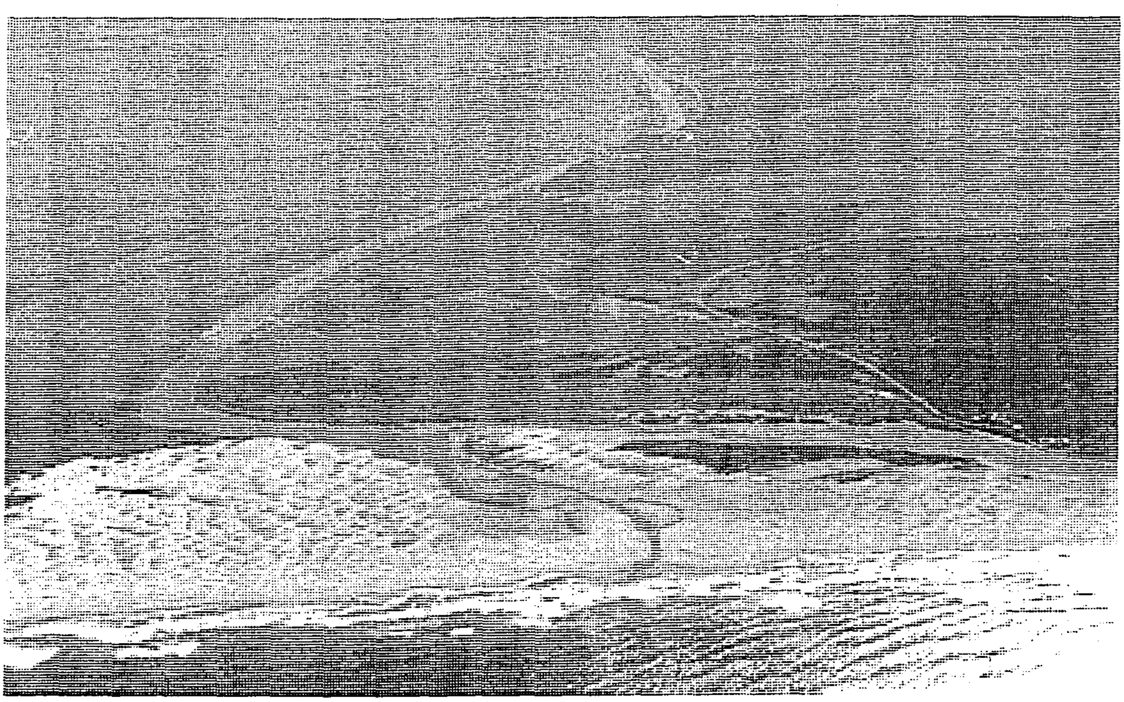

Figure 2.3 Aerial photograph of the mouth of the Gamtoos estuary at approximately spring low tide, taken in August 1992.

foredune, changing the position of the mouth and again allowing a much better exchange with the sea. The tidal inlet was last blocked off from contact with the sea in 1960, and the last flood to scour out the estuary mouth occurred in 1983 (Reddering and Scarr, 1990a). Zhang *et al.*, (1995) report on a storm surge coincident with a spring high tide in May 1992 which resulted in an estimated 27 000 m^3 of bank material and wind-blown dune sand entering the estuary.

The estuary is shallow, generally between 2.5 and 3.5 m in the lower reaches and averaging between 0.5 and 1.5 m in the upper estuary. The difference in depth between the lower and upper reaches of the estuary is illustrated in Figure 2.4. The aggraded areas in the upper estuary influence the extent of tidal flushing during neap low tide. There are localized area of scour throughout the estuary (Reddering and Scarr, 1990b).

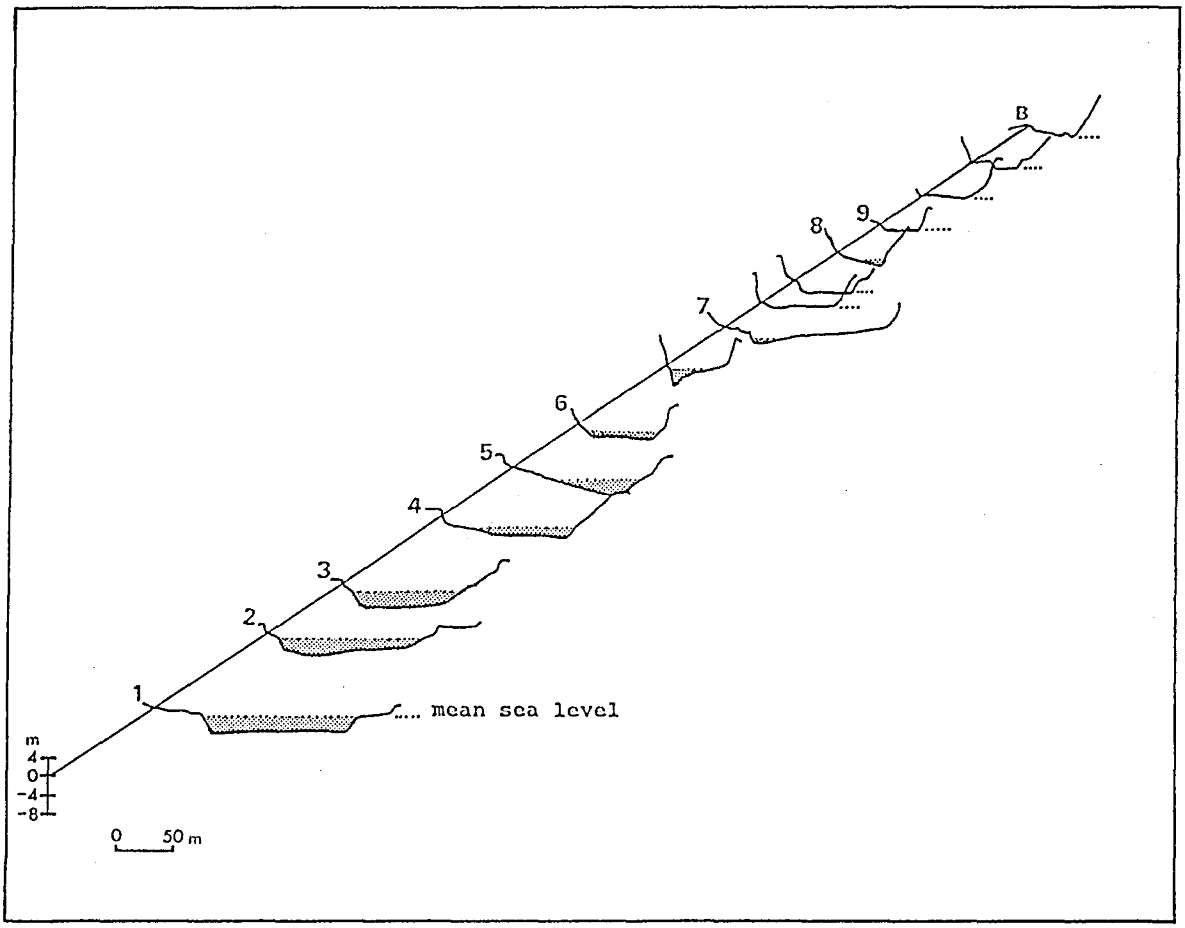

Figure 2.4 Successive profiles of the estuary from station (1) near the estuary mouth up-estuary to Boschhoek (B). The figure shows the diminishing depth and channel width up-estuary (after Reddering and Scarr, 1990b).

2.2.4 Geology and Soils

Figure 2.5 shows the general geology of the study area. The half-graben Gamtoos basin is bounded in the north-east by the Elandsberg fault and in the south-west by the Gamtoos fault (Shone *et al.*, 1990), with the Gamtoos River flowing roughly parallel to the Gamtoos fault. Deposits of the Uitenhage Group occur in the Gamtoos Basin, and in the west, rest unconformably on the rocks of the Bokkeveld Group. The Enon Conglomerate Formation comprises coarse conglomerates which are generally poorly sorted with interbedded

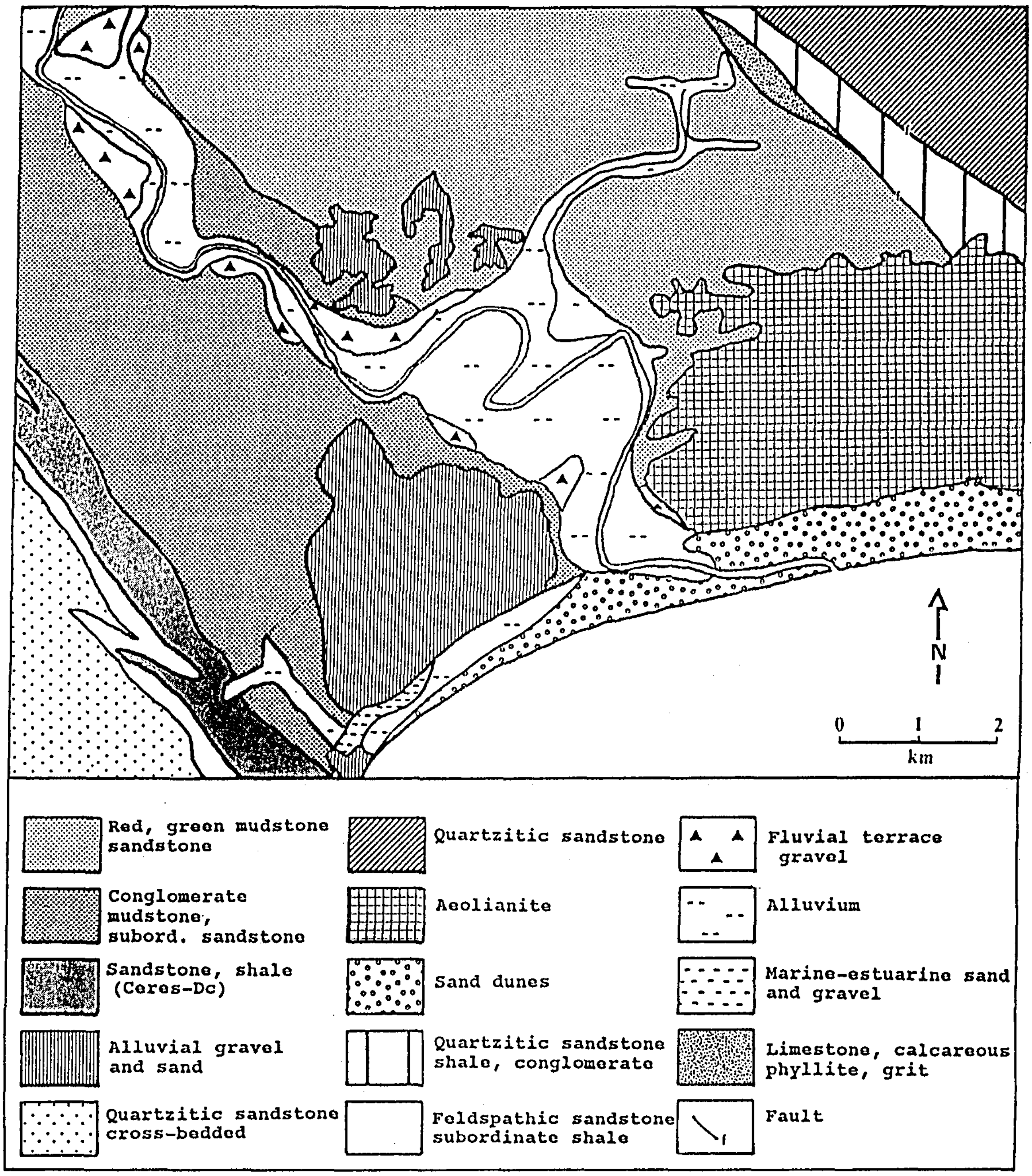

Figure 2.5 Geology of the Gamtoos Region (after Shone *et al.*, 1990; Geological map).

sandstones and mudstones. A series of modern terraces have been cut in these strata, down to river level: the highest terrace comprises poorly sorted gravels, sand and clays, while the lowest terrace, approximately 20 to 30 m above sea level, has thick red soils (Haughton, 1937; Toerien and Hill, 1989).

Extensive shifting sand dunes occur at the Gamtoos estuary mouth. These recent aeolian sands are derived from sandstones of the Cape Supergroup (Toerien and Hill, 1989).

The study area is situated in the alluvial floodplain bordering the estuary and therefore soils will have been influenced by fluvial and marine activity. Furthermore ploughing and reworking of the land has destroyed any previously established soil horizons.

2.2.5 Land Use

Land use in the estuarine floodplain is agricultural, with mostly vegetable cultivation and to a lesser extent the grazing of sheep and cattle. In the lower floodplain grazing predominates and there is also a caravan park and holiday resort adjoining the estuary (Figure 2.2).

In the Loerie Flats (Figure 2.2) there is intensive crop rotation: the 650 ha area is subdivided into numerous small (0.1 ha) fields with differing vegetable cover. At any one time the fields are being prepared for a new planting, lying fallow, being harvested or have crop covers at various stages of growth. As fertilizers are expensive certain farmers adapt their planting practices accordingly. Cabbages for example require a greater nutrient input than carrots, thus at the time of planting cabbages fertilizers are applied, approximately one hundred and twenty days later when the mature crop is harvested carrots are then planted with no further fertilizer application. Different vegetables are grown throughout the Loerie Flats (gem squash, lettuce, sweet potatoes, butternuts) each with differing crop requirements, growing periods and tolerance to saline soil conditions. Certain farmers use organic, phosphate-enriched bird guano rather than the more expensive chemical fertilizers. Approximately two tons of phosphate-enriched bird guano is applied per hectare to various fields throughout the Loerie Flats.

2.2.6 Irrigation

Irrigation water in the Gamtoos region is distributed via a system of canals, siphons and pipelines. The main system comprises a 72 km canal from the Kouga dam, with an adjoining 17 kms of siphons and 8 kms of tunnels. There are an additional 30 kms in five tributary canals. The canals are concrete lined which reduces losses via seepage, with total losses estimated at less than 5%. Water is abstracted from the canal through 150 mm diameter pipes with a control valve. The total length of the pipelines is 91 km. This allows direct access to water at any time and metering of water used, with the meters read at fortnightly intervals. The accuracy of metering is estimated at less than 2% error (Chapman, 1986; Department of Water Affairs and Forestry, 1992).

The irrigation system serves a total scheduled area of 9 880 ha producing mainly vegetables, citrus, tobacco, wheat and lucerne. In the study area irrigation is mostly by way of mobile, metal-arm sprinkler irrigation. The full water quota is 8 000 $m^3.ha^{-1}.a^{-1}$ during normal water years, and any water savings incurred by the farmer as a result of efficient methods of irrigation can be used to irrigate non-scheduled land. Because of below average rainfall during 1992, the farmers were initially on 15% of their normal irrigation quota, and this was increased to 25% following the 1 September review. Full quotas were reinstated on 21 October 1992 following good rains during which the Kouga dam rose to 58% of its capacity (Bentley, 1992).

Excess canal water flows to a balancing dam situated at the end of the canal system; there are two such dams, namely the Loerie dam (north of the estuary - Figure 2.1) and a dam located at Green Acres (west of the estuary - Figure 2.2). A purification works is situated at the Loerie dam from which water is abstracted and purified for domestic use in Port Elizabeth.

2.2.7 The Surface Agricultural Drainage System (Loerie Flats)

Oosterbaan (1994, p. 635) defines agricultural drainage systems as '*systems which make it easier for water to flow from land, so that agriculture can benefit from the subsequently reduced water levels. The systems can be made to ease the flow of water over the soil surface or through the underground, which leads to a distinction between surface drainage systems* *and subsurface drainage systems. Both types of systems need an internal or field drainage system which lowers the water level in the field, and an external or main drainage system, which transports the water to an outlet. A surface drainage system is applied when the waterlogging occurs on the soil surface, whereas a subsurface drainage system is applied when waterlogging occurs in the soil.'*

In the Loerie Flats natural surface runoff occurs away from the river to a low-lying region adjacent to the Melon railway line (Figure 2.6). Runoff from the fields accumulated in this low-lying region and led to the build-up of stagnant water in this area following periods of heavy rainfall and flooding with no natural drainage towards the river. The standing water then resulted in an elevated water table as shown in Figure 2.7 and lands became infertile due to the build-up of salts within the soil.

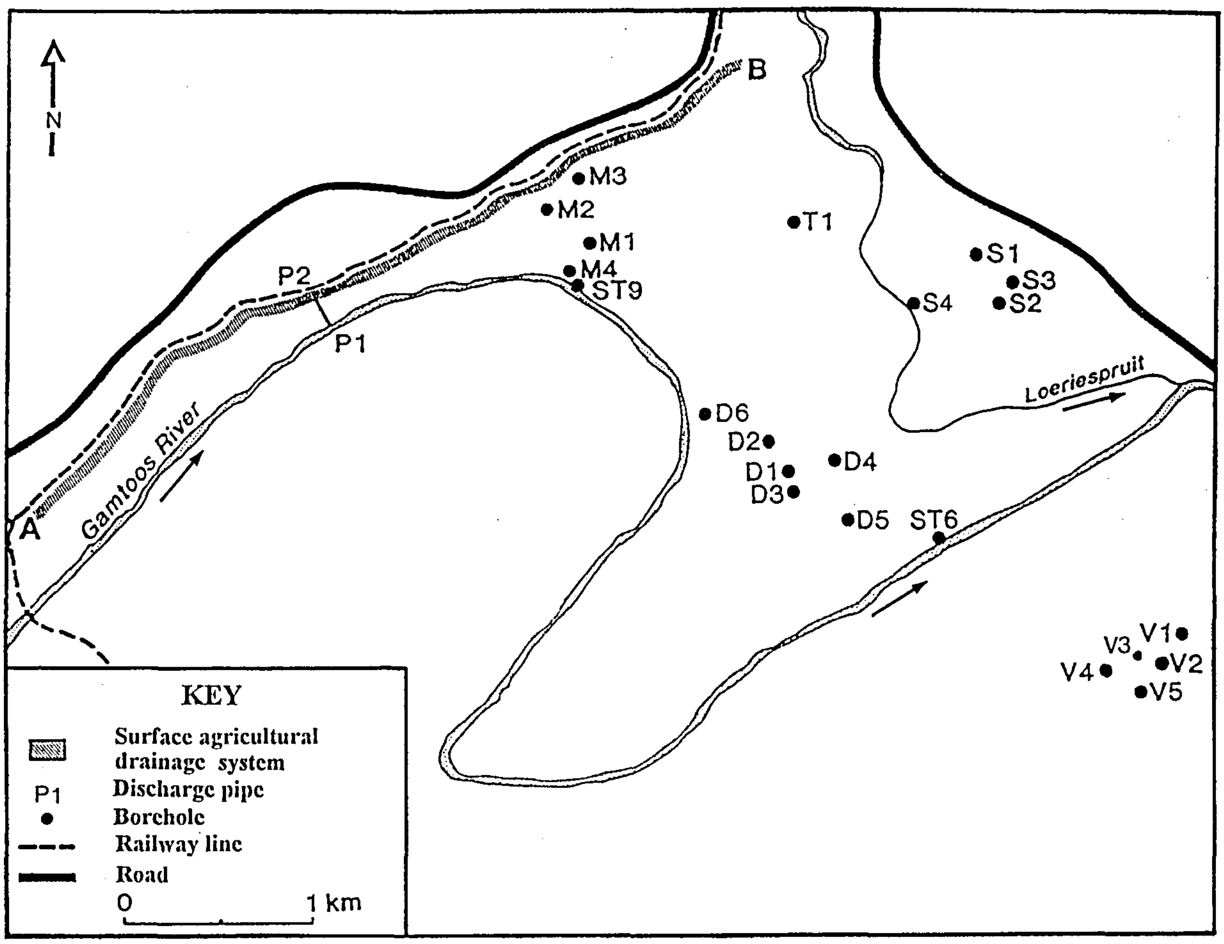

Figure 2.6 The agricultural drainage system installed in the Loerie Flats is indicated from A to B, while P1-P2 shows the drainage pipe. The position of the boreholes is also shown.

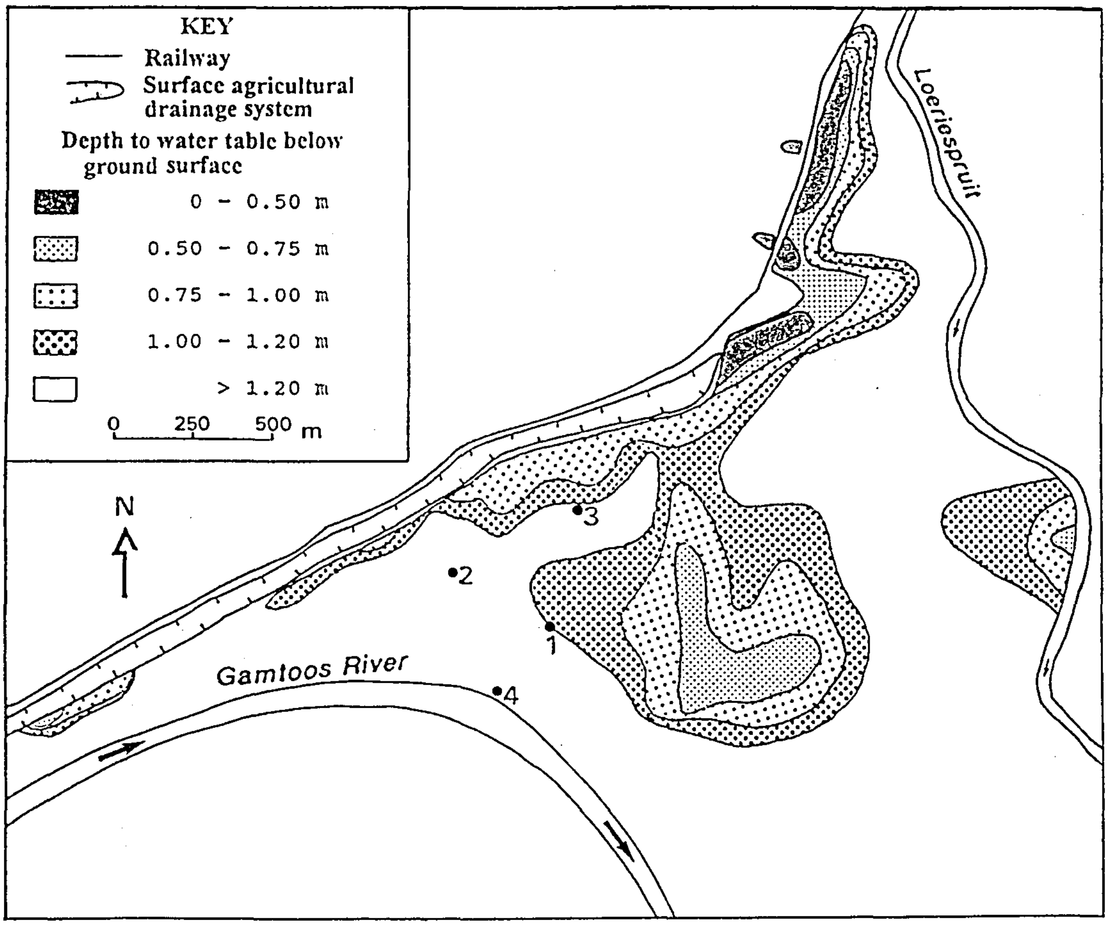

Figure 2.7 The elevated water table in the Loerie Flats prior to the installation of the surface agricultural drainage system (Gamtoos Irrigation Board).

This problem led to the subsequent design and installation of a surface drainage system by the Departments of Agriculture and of Water Affairs in 1985, shown schematically in Figure 2.6. The original drainage system designed by the Departments of Agriculture and of Water Affairs was far more extensive than the drainage system shown in Figure 2.6, and the original system included drainage of the fields bordering the Loeriespruit to the point of confluence with the Gamtoos estuary. The farmers in the area were reluctant to extend the drainage system because they were required to contribute financially to the scheme.

The main drainage system comprises a line of trenches and pipes, extending from A to B in Figure 2.6, lying adjacent to the railway line. The system marked A - B on the figure drains an area of approximately 50 ha during non-flood conditions. Agricultural runoff carried by the drainage system converges at point P2 and then discharges into the Gamtoos River at the drainage pipe at P1. The pipes are sealed to prevent entry and blockage by roots. Manholes and bends to facilitate rodding or pumping, should silt or other debris need to be cleared from the system, occur at 200 m intervals (Joubert, 1992).

2.2.8 The Subsurface Drainage System (Loerie Flats)

A subsurface drainage system, a typical one of which is schematically illustrated in Figure 2.8, usually comprises a series of shallow (less than 6 m below the ground surface), closely spaced drainage pipes. Originally subsurface ceramic or concrete tiles were used, hence these systems are still commonly referred to as tile drain systems. A cheaper, modern substitute is PVC piping.

The regular, subsurface field drainage system comprises buried slotted pipe drains at a depth of less than 6 m leading to an outlet pipe that drains under gravity and usually discharges into a neighbouring river or estuary. The subsurface pipes are surrounded by a gravel-pack which similar to that of a borehole gravel-pack, provide added support to the piping, promote permeability around the drainage system and prevent clogging of openings with fine material. The subsurface layout of the drainage pipes is either in an angled Herringbone system or in a parallel configuration as shown in Figure 2.8. Depending on the size of the system, a number of regularly spaced manholes intercept the system. These allow access to clear blocked pipes (Ritzema, 1994b; Cavelaars *et al.*, 1994).

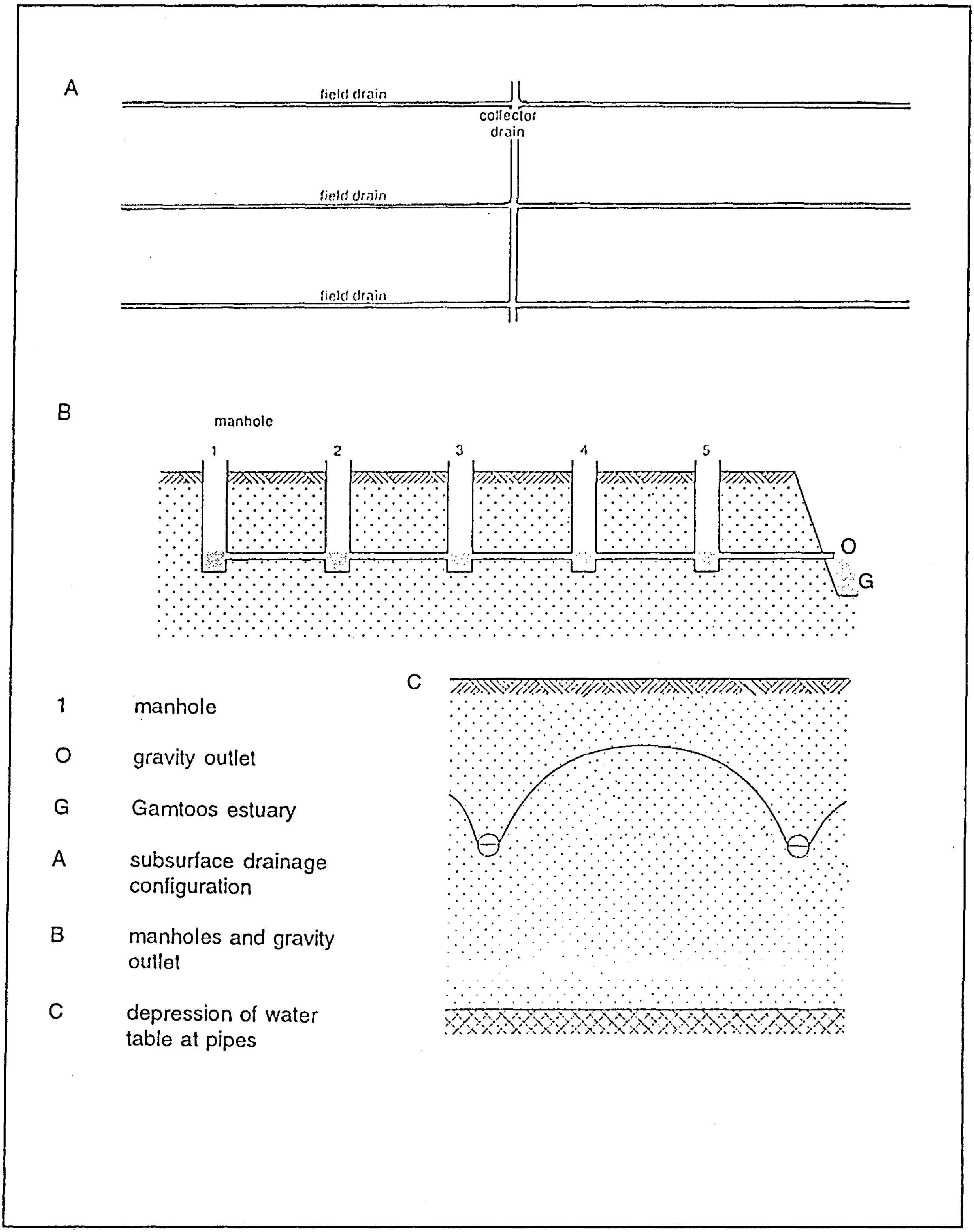

Figure 2.8 Schematic subsurface pipe drain system (adapted from Cavelaars et al., 1994).

3. MATERIALS AND METHODS

The study objective was to investigate the influence of agriculture on the estuary, and therefore the experimental programme had to be designed to monitor conditions both on land and in the estuary. The details given below describe the measurement programme both in the estuary and in utilising boreholes in the agricultural area in order to meet these objectives. Detailed selection of the area where the project was to be undertaken was made after an aircraft survey on 12 August 1992, and a later on-site inspection.

At the start of the investigation a decision was made to conduct a regular measurement programme, interspersed with more intensive measurement periods; manpower and instrumentation constraints meant that continuous intensive monitoring could not take place over the whole experimental period. Some continuous monitoring was initiated using self-recording instruments, but equipment problems meant that unbroken records were not obtained. The mode of investigation, methods and equipment used during both intensive and routine monitoring was similar in order to provide seasonal comparative data. Nonetheless, slight variations occurred, and at times improvements were made over the course of the project. The investigation commenced at the end of a dry period, and there were several occasions when heavy rains were recorded in the river catchment.

3.1 Measurement Periods

The measurements took place over the period August 1992, to February 1994, though some monitoring continued to the end of 1994; details of the measurements are given in subsequent sections. Over the whole period, monthly measurements were made at the boreholes, and water samples taken at selected points.

The estuary and boreholes were monitored over four intensive periods in the eighteen month field study, details of the four intensive measurement periods follow.

November/December 1992

Estuarine and borehole readings were taken over eighteen consecutive days from 23 November (spring tide - 24 November) to spring tide on 10 December. This was the most intensively monitored estuarine investigation, and involved a number of moored, self-recording instruments as well as regular measurements from a small inflatable boat.

March/April 1992

Financial restrictions limited readings to an eight day period over a neap (31 March) to spring (6 April) cycle.

June 1993

A full seventeen day monitoring period from 4 June (spring tide) to 20 June was planned. This would have enabled a comparison with results taken over a full tidal cycle in November 1992 and thereby indicate possible seasonal differences (winter - summer). Due to unforeseen equipment problems the programme ceased monitoring after twelve days (16 June).

In addition to the normal measurements taken (as described in section 3.1) water samples were taken in the centre of channel flow every two hours throughout the time series (both on 5 and 13 June). Analysis of such samples would enable detection of changes in the nutrient characteristics over a tidal cycle - which cannot be detected with the electrical conductivity/ temperature/ depth equipment (henceforth referred to as CTD measurements).

November 1993

For purposes of comparison a final period of intensive measurements was taken over the period 7 November (neap tide) to 14 November (spring tide). As a result of financial constraints readings were taken on three consecutive days over neap tide and three days over spring tide. Measurements differed from the preceding periods in that only stations 4 to 10 were monitored (because of limited assistance available). Water samples were also taken at two hourly intervals during time series measurements to investigate any seasonal differences to those occurring in June.

3.2 Estuarine Measurements

The measurement methodology enabled both the monitoring of selected parameters at specific sites, as well as the determination of estuarine structures; for these purposes ten stations were selected, spread over an 18 km distance from inside the flood-tidal delta, to near the tidal head (Figure 2.2). Water column characteristics were measured at these station positions using a small boat, while self-recording instruments were also deployed at a few selected sites.

Cross-sections had been accurately measured by Reddering and Scarr (1990b) at forty sites spaced by approximately 500 m along the whole length of the estuary (see Figure 2.4). Some changes in cross-section could have taken place as a result of subsequent sedimentation or erosion, but the cross-sections are used here to give an approximate longitudinal profile.

3.2.1 Self-Recording Instrumentation

At the start, a weather station was erected at the Gamtoos Ferry hotel site, station 4, to measure wind speed, gust and direction; and air temperature and humidity using an automated MCS data logger system. Rainfall was recorded using a tipping bucket gauge, recording at 5 minute intervals on an event basis only. However, considerable difficulties were experienced with the recording system, and few reliable data were obtained.

MCS tide gauge recorders were used to obtain records of water level variations in the estuary. These consisted of a float and pulley system mounted in a closed pipe with an MCS encoder and logger recording at 15 minute intervals. The bottom of the pipe was sunk into the sandy soil of the estuary, acting therefore as a filter to remove high frequency water level fluctuations. The accuracy is estimated at 1 mm, though some settling of the support structure and pipe may have affected this.

The main tide recording station was established at the hotel site in November 1992, attached to the fixed jetty. This gave good data, with a few gaps, until heavy rain and flood waters damaged the jetty in June 1993, and the gauge had to be removed. At the start, a second suitable site to monitor water level variations was identified at station 6, and prior to the intensive measurement periods a tide gauge was erected. However, the whole tide gauge structure was found toppled over in the estuary in June 1993; it was never ascertained how this happened, but the instrumentation was beyond repair, and the site was not used again.

Pressure sensors constructed by the CSIR were also used in the first experiment in 1992 to measure water level variations. Two such meters were used, one deployed at station 1 and the other at station 9 and set to record at 15 minute intervals. To represent conditions in the adjacent ocean, hourly water level data were obtained from a tide gauge maintained in the Port Elizabeth harbour by the South African Naval Hydrographer. Propagation of the tidal signal is from west to east along the coast with very little variation over the whole region.

Two Vector Averaging Current Meters (VACMs) constructed by the CSIR were used in the first experiment, anchored at stations 4 and 9, in an attempt to measure the flow of water past these positions. However, an analysis of the results showed that the meter at station 9 was not situated in the main flow region, while the meter at station 4 was affected by the density stratification in the water body, at times being in the surface flow and at other times in the bottom flow. Not enough meters were available to adequately resolve these problems, and current meters were not deployed on subsequent experiments.

Two temperature sensors were maintained for part of the project at the hotel site. The top sensor was attached to the floating jetty, while the second was established on the estuary bed. Unfortunately these could not be regularly serviced, and while the top sensor was recovered, the bottom sensor was swept away or buried in the flood of June 1993, and was not recovered; the flood also removed the floating jetty.

3.2.2 Boat Measurements

During the intensive measurement periods, a small inflatable boat was used to make the measurements at the ten selected stations in the estuary (Figure 2.2). At each station the boat was anchored approximately in the centre of the estuary, and maximum water depth. Secchi disc depth, and vertical profiles of temperature, electrical conductivity (EC), salinity (determined from EC) and dissolved oxygen (DO) were measured; at a few selected stations a profile of current speed and direction was also obtained. To establish the vertical profile readings were taken at 0.5 m intervals from the bottom to the surface; where a sudden change in a variable was detected, readings were then taken at every 0.1 or 0.2 m to determine the precise structure of the change.

Readings were taken twice a day at low and high tides. South African tide tables (1992; 1993) were used to determine the estimated time of high and low tides in the ocean in the vicinity of the Gamtoos River mouth. Because of the delay in the tidal influence within the estuary, measurements commenced at station 1 half an hour after the respective tidal peak or trough, as determined for the Gamtoos River mouth. In some instances very low waters during neap low tide prevented the passage of the boat over an aggraded stretch upstream of station 7.

Electrical conductivity (in $mS.m^{-1}$), salinity (in parts per thousand, or ‰, but in keeping with modern usage, since it is measured as a ratio, will be given here as a non-dimensionalised value), temperature (in ^0C) and depth readings were made using a Valeport series 600 MkII CTD with accuracy's of 0.3 and 0.1^0C, respectively. Faulty apparatus created problems in the measurement of DO, and this resulted in many occasions where data could not be obtained. Two instruments were used to measure DO and these often showed differences in registered values at the same site. The one instrument used was a portable, microprocessor-based, autocalibration dissolved oxygen meter (Hanna HI 914) with a built-in thermistor for temperature measurements and compensation. The instrument has a 1% accuracy over the entire scale of 0 to 19.99 ppm ($mg.l^{-1}$). The second instrument used was a YSI Model 57 Oxygen Meter which required manual calibration for salinity.

Current readings from the anchored boat were taken using a Valeport BFM 108 Direct Reading Current Meter, with a given accuracy of 1 $cm.s^{-1}$, although considerable variability was experienced and it is doubtful that the current measurements were this accurate. The inclination and orientation of the instrument were also difficult to control at times.

The estimate of freshwater input is important to this investigation. It rained in November 1992, prior to the first experiment taking place, and a considerable volume of freshwater entered the estuary as runoff. During the course of the experiment this freshwater input tailed off, and estimates were made of this flow at station 10: this was done by obtaining a current speed profile using the Valeport current meter, and also making an accurate measurement of the estuary cross-section depth at that site. Unfortunately, regular measurements were only started on 1 December, after the main influx of water had already abated. Although an estimate of the flow velocity over a short distance near station 9, was made on 29 November. Quite clearly these measurements can only give a rough estimate of the volume flow. Rough estimates of flow were determined in a similar manner during the April 1993 and June 1993 measurement periods.

In order to determine the chemical character of the estuarine water, samples were taken just below the water surface, at each station at low tide on both spring and neap tides. The low tide situation was chosen since it would be at that time that flow from upstream would penetrate into the estuary. Where salinity readings showed a marked difference between bottom and surface waters, (i.e. an intense halocline was evident), then water samples were taken at the bottom and surface. Bottom water samples were taken just above the estuarine sediment, using a weighted, plugged bottle with pull-string release for sample capture. All samples were kept cool, frozen and later analysed as discussed in section 3.5.

3.3 Borehole Measurements

At the start of the project suitable sites were selected, and a total of twenty boreholes were established. Monitoring of rest water levels indicates trends in water table changes from a seasonal perspective and possible short-term responses to rainfall and irrigation inputs. Groundwater samples were analysed to determine the input of nutrients to groundwater via the soil profile.

The measurements in these boreholes were made in conjunction with the estuarine measurements during the intensive periods indicated in section 3.1; i.e. twice daily monitoring of rest water levels, with groundwater samples obtained twice during each period. Outside of the intensive periods, groundwater samples and rest water levels were taken once per month, and this was done over the whole period of the experimental programme.

3.3.1 Borehole Installation

Boreholes sites were identified in accordance with the geological structures within the study area according to Zhang (1995). Boreholes were positioned in close proximity to vegetable fields but clear of ploughing and cropping practices. Boreholes were grouped together at five sites, the close positioning of the holes relative to each other necessitated by the highly variable underlying alluvial material. As far as possible boreholes were placed in a triangular configuration so that direction of groundwater flow could be ascertained.

Once suitable sites had been located, the boreholes were installed over the period August to October 1992. The number of boreholes installed was limited to twenty for the following reasons:

- the installation process was time consuming
- to allow sufficient time (the remainder of the allocated study period) for monitoring of rest water levels and groundwater quality
- time constraints involved with the routine monthly sampling when nine surface and twenty groundwater samples were collected in a day.

An auger and vibracorer were used to drill the holes, the vibracorer was the instrument of choice (as opposed to a drill rig) for the following reasons:

- easy access to borehole sites without damaging crop fields
- limited expertise required to operate the system
- low cost
- changes in the soil profile were easily identified and depths of soil change could be determined and samples obtained.

The main disadvantage in using a vibracorer, was that the depth to which the holes could be bored was limited to a maximum of 6 m. However, in accordance with the aims of the project, namely to investigate whether the groundwater would be influenced by agriculture and enter the estuary, it was acceptable to limit the groundwater investigation to the upper 6 m. In sandier soils the vibracorer did not retain the sample, and even when a non-return catch claw was installed at the end of the borer pipe, persistent attempts could not retain the sample and the hole caved in. The underlying soil characteristics thus determined the final depth of the boreholes which ranged from 4.30 to 5.73 m.

During borehole installation each colour change and textural change in soil was sampled and the depth of change noted. Soils textures were differentiated according to particle size composition (section 3.5.3).

Method of Borehole Installation

Holes were augered manually as far as possible and thereafter the vibracorer, fitted with a 6 m length of 63 mm diameter aluminium piping, was used to bore to the required depth. Once the borer pipe was removed the borehole casing was immediately placed down the hole to prevent further caving. Casing comprised 63 mm diameter PVC piping with a 1 m length of continuous slotted screen in the lower, capped end. The 0.2 mm slot openings of the screen provide adequate open area and strength to the borehole. The area around the screen was sand-packed using washed quartz sand (Reynders, 1992). The removed soil was used to back-fill the bore space to the surface, and clay material was then packed around the borehole at the ground surface to prevent contamination of the deeper water by surface runoff.

The depth of the borehole below the ground surface and the height to which the casing protruded above the ground were noted. The boreholes were accurately surveyed relative to mean sea level at a later date (section 3.4.2). Boring processes disturb the soil adjacent to the hole, and thus boreholes were left to stabilise for a month after installation before routine monitoring commenced.

3.3.2 Borehole Monitoring

Rest water levels (hereafter referred to as RWLs) were measured using a Gemlab water contact sensor lowered down the borehole. An electrical circuit closes and a buzzer sounds when groundwater is reached and the length of conduit could then be measured to obtain the depth. This device has an accuracy equal to other RWL recorders (i.e. to the nearest cm) and is easy to use, reliable and has low initial and maintenance costs.

Two continuous recording MCS data loggers, together with a float, counter-weight and pulley system were established on boreholes D1 and D5 on 13 August 1993. Unfortunately problems were experienced with the measurement system, and only limited records are available for analysis. These holes were chosen for the following reasons:

- the two boreholes are relatively close to each other (380 m apart)
- from routine monitoring results the boreholes appeared to be functioning effectively and responded to rainfall and irrigation inputs
- the depth of the water table relative to mean sea level differed in the two holes (on average by about 1 m).

The loggers were initially set to log every 10 minutes but this was later changed to log on an hourly basis. The memory capacity of the MCS data loggers was sufficient to enable data to be collected continuously for many months. However the system was powered using batteries, which limited the logging period to about two months.

Groundwater Sampling

The boreholes were also used for groundwater sampling, and initially a submersible pump was used to obtain water samples. Despite the presence of a sand-pack and the use of the smallest screen slot openings available, the high percentage of very fine material (<0.1 mm) in the vicinity of the screen resulted in a large amount of fines being drawn into the hole and pump. Continuous use of this method in all twenty boreholes would have damaged the pump and shortened the monitoring life of the boreholes; consequently a grab sampler was used in the later stages.

3.3.3 Aquifer Recovery Tests

Aquifer recovery tests were performed on all boreholes. Conditions were not conducive to aquifer testing, since the small-diameter, shallow holes were tapping mostly clay material. However, holes were pumped dry and their recovery rates monitored.

The tests were conducted using a battery-operated submersible pump. The length of time it took to empty the borehole and the quantity of water removed were noted. The recovery test commenced as soon as the borehole ran dry and the pump was removed. Residual drawdowns were monitored as far as possible to full recovery (restoration of initial RWL). Hydraulic conductivity was determined (section 3.6.4) by applying the Ernst equation (Oosterbaan and Nijland, 1994) to the data obtained from the aquifer recovery tests.

3.4 Additional Sampling and Surveying

3.4.1 Routine Water Sampling

To supplement the measurements in the estuary and the boreholes, additional sampling points were identified. The volume of discharge at the agricultural drainage pipe (section 2.2.7 and Figure 2.6) was measured at the same time as routine surface and groundwater sampling. The length of time it took to fill a bucket of known capacity was measured, repeated and an average value obtained. Water samples were taken for later analysis.

On the regular trips where no boat was available for comprehensive estuarine sampling, some samples were nonetheless taken at sites 4, 6, 9 and at Boschhoek (Figure 2.2) in order to obtain an idea of the inherent longer-term variability of the system. This routine sampling was set to be conducted at low tide. However, because of the number of samples that had to be obtained this was not always possible. Staff gauges were installed on the estuary bank at stations 6 and 9 to enable the exact height of the estuary water level to be determined (relative to mean sea level) at the time of sampling. By reading off the depth of water at the staff gauge, the water table gradient between the boreholes inland of each staff gauge and the estuary could be calculated. Apart from the method of sampling discussed in sections 3.2 these water samples were obtained on a monthly basis at neap low tide .

The irrigation water in the canal originating from the Kouga dam was sampled at Green Acres (Figure 2.2) on 16 July, 24 November, 27 November in 1992, on 1 April, 29 July and 12 October in 1993, and on 2 February in 1994.

3.4.2 Surveying

The boreholes, tide gauge at site 4, and staff gauges were surveyed relative to mean sea level and relative to each other. This enabled their positions to be accurately plotted and the groundwater level relative to mean sea level throughout the area determined. This in turn provided the basis for determining the hydraulic gradient and direction of groundwater flow.

The forced centric traversing method (Durban Corporation, 1991) was followed in the survey, using a Wiid T 1 600 theodolite and D 1 600 electronic distomat. The maximum error of this method is approximately 1 cm with original levels taken from bench marks to give values relative to mean sea level.

3.5 Water and Soil Analyses

All water samples were kept in a cooler box with ice-packs until the laboratory was reached, where they were filtered to render silt free water, since silt interferes with certain analytical methods. Samples were kept frozen until the chemical analyses could be done.

3.5.1 Chemical Analyses

Chemical analyses were done by the Port Elizabeth Municipality using standard methods and equipment (APHA, 1989). Electrical conductivity ($mS.m^{-1}$) at 25^0C was determined using a Knick 702 conductivity meter. The cations: calcium (Ca^{++}), magnesium (Mg^{++}), sodium (Na^+), potassium (K^+) and anions: chloride (Cl^-), and sulphate (SO_4) were determined using an ion chromatograph. Salinity was calculated from the Cl^- concentration. Nitrate as N was determined using the sodium salicylate method, nitrite as N and total phosphorus (as P) by calorimetric development and total alkalinity by automatic titration with sulphuric acid in combination with pH by electrode.

3.5.2 Pesticide Screening

The number of samples obtained (six) for pesticide screening was limited by the expense of such analyses. A broad screening method - the cholinesterase inhibition test (Boehringer Mannheim) was used. It is a calorimetric test for the determination of organophosphate and carbamate pesticides in water by means of inhibition of acetylcholinesterase. Acetylcholinesterase is an enzyme found in pests; organophosphate and carbamate pesticides operate by inhibiting the functioning of this enzyme. The method determines this inhibition thus making it a semi-quantitative method only. It has a sensitivity of approximately 0.05 $mg.l^{-1}$ and results are given in paraoxone equivalents (POE). Pesticide determination was done by the Port Elizabeth Technikon PETCRU unit.

3.5.3 Soil Particle Size Distribution

Soil particle size distribution (Black, 1965) was determined for the soil samples obtained during borehole installation. A settling column designed by Reddering using the method based on Stoke's Law gave percentage composition of the soil as follows: % mud (<0.05 mm diameter), % very fine sand (0.05 mm - 0.1 mm), % fine sand (0.1 mm - 0.25 mm), % sand (0.25 mm - 0.5 mm), % coarse sand (0.5 mm - 1.0 mm) and % gravel (>1.0 mm).

3.6 Data Analysis and Presentation Methodology

A considerable volume of data was collected over the course of the project, and this section gives details of some of the data editing and processing used in analysing the results and presentation techniques.

3.6.1 Time Series Data

All the self-recording instruments (discussed in section 3.2.1) and the continuous borehole loggers, as well as the tide gauge data from Port Elizabeth harbour provided time series data at intervals ranging from 5 minutes to an hour. These data had to be milked from the appropriate memory modules, and were then edited and processed using standard procedures developed in the Department of Oceanography (Schumann and Martin, 1994); they are available in a binary format for any further analyses.

Depending on the analyses required, various filters can be used to investigate different frequency bands. In particular, a low frequency Cosine Lanczos filter is used to eliminate the shorter period fluctuations and obtain hourly values, while a further filter is used on these hourly values to eliminate tidal fluctuations.

3.6.2 Boat Data

The boat data were recorded by hand on prepared schedules, and then had to be transcribed into computer files for use with specially written computer programs to plot out estuary sections of the different variables; the time series measurements taken at station 4 were treated in a similar fashion.

3.6.3 Trilinear Diagrams

The hydrochemical indices obtained from water sample analysis were be used to classify and identify water types by means of trilinear diagrams; these also facilitate comparison between numerous samples (Matthess, 1982). The trilinear diagram proposed by Piper (Figure 3.1) is not based on absolute quantities but considers the ratio of specific anions and cations to the total anion-cation composition (Back, 1966). Once plotted on the trilinear diagram, water types were identified, according to Figure 3.1, with the relative composition a fingerprint of specific influences on water quality.

The Piper diagram comprises a cation triangle, an anion triangle and the resultant plot on the diamond. Total cations comprise (i) sodium plus potassium, (ii) calcium and (iii) magnesium. The total anions comprise (i) sulphate, (ii) chloride and (iii) carbonate and bicarbonate, with all ions expressed in meq.l^{-1}. Trilinear diagrams are referred to in section 5 to illustrate differing water types.

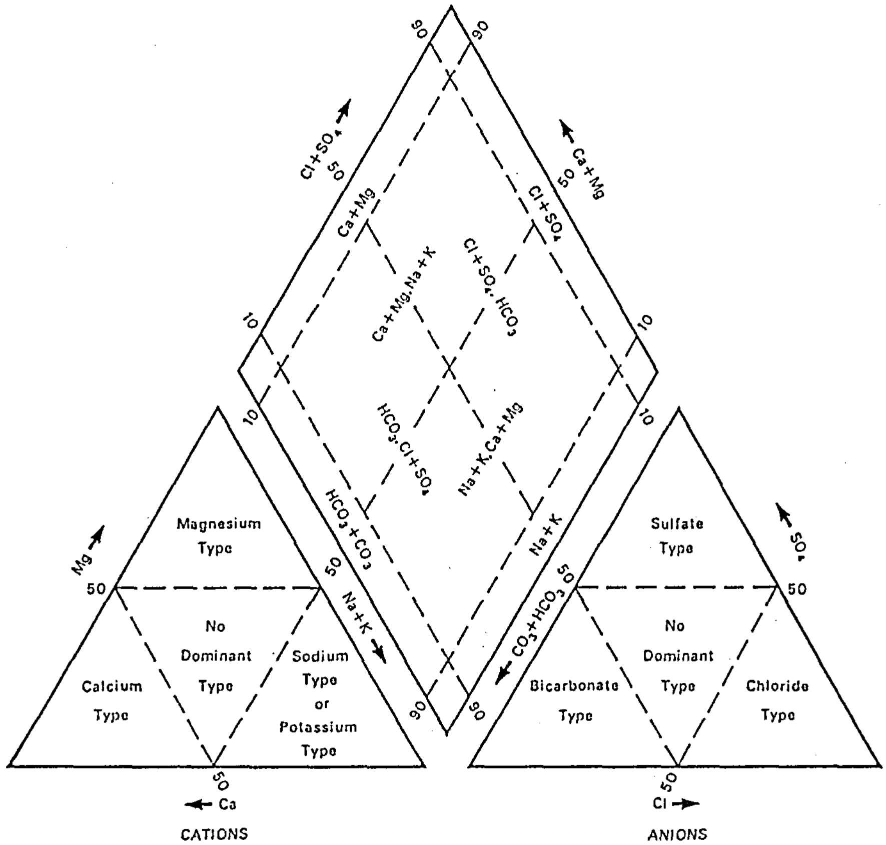

Figure 3.1 The Piper diagram (after Back, 1966).

3.6.4 Hydraulic Conductivity

In 1856 Darcy derived an equation describing the flow of groundwater. Darcy's Law states that the volume of water flowing through a soil column per unit of time is proportional to the column's cross-sectional area and its difference in head and inversely proportional to the length of the soil column. Darcy's Law is given as:

$$Q = KA\frac{\Delta h}{L} \qquad (1)$$

where Q = rate of flow through the column ($m^3.s^{-1}$)

h = head loss (m)

L = length of the column (m)

A = cross-sectional area of the column (m^2)

K = a proportionality coefficient, called hydraulic conductivity ($m.s^{-1}$)

Hydraulic conductivity, K, is the constant of proportionality in Darcy's Law, defined as the volume of water that will move through a porous medium in unit time, under a unit hydraulic gradient, through a unit area, measured at right angles to the direction of flow. The K-value of a soil profile can vary within a soil layer and at different depths. K-values at a particular point can also change over time, although this occurs mostly in clays that are subject to alternating periods of wetting and drying (Bos, 1994).

Oosterbaan and Nijland (1994) describe the auger-hole method (a small-scale in-situ method) of determining hydraulic conductivity. The auger-hole method is similar to the aquifer recovery test described in section 3.3.3, and the Ernst equation, which is normally applied to the findings of the auger-hole method, has been applied to the results obtained from the aquifer recovery tests. As it is an adaptation of the method not all data met the requirements of the Ernst equation and therefore hydraulic conductivity was only determined for certain boreholes that met the Ernst equation requirements.

The theory of the Ernst equation as cited by Oosterbaan and Nijland (1994) determines hydraulic conductivity (K) based on the average rate of rise in groundwater level following the emptying of the hole. The Ernst equation (refer to Figure 3.2) is given as:

$$K = C\frac{H_0 - H_t}{t} \qquad (2)$$

where C = a factor as defined in equation (3) below

t = time elapsed since the first measurement of the rising water level in the hole (s)

H_t = depth of the water level in the hole below a reference level at time t (cm)

$H_0 = H_t$ when t = 0

The C factor depends on the depth (D) of a hypothetical impermeable layer below the bottom of the hole, and the average depth of the water level in the hole below the water table (h') as given in equation (3).

$$C = \frac{4000\frac{r}{h'}}{(20+\frac{D_2}{r})(2-\frac{h'}{D_2})} \tag{3}$$

where D_2 = depth of the bottom of the hole below the water table (cm), with the condition that $20 < D_2 < 200$

r = radius of the hole (cm): $3 < r < 7$

h' = average depth of the water level in the hole below the water table (cm), with the condition: $h' > D_2 /5$

Note: In the application of this equation to Gamtoos data, D was taken to be greater than D_2 (if D = 0 then a different equation for determining C is used).

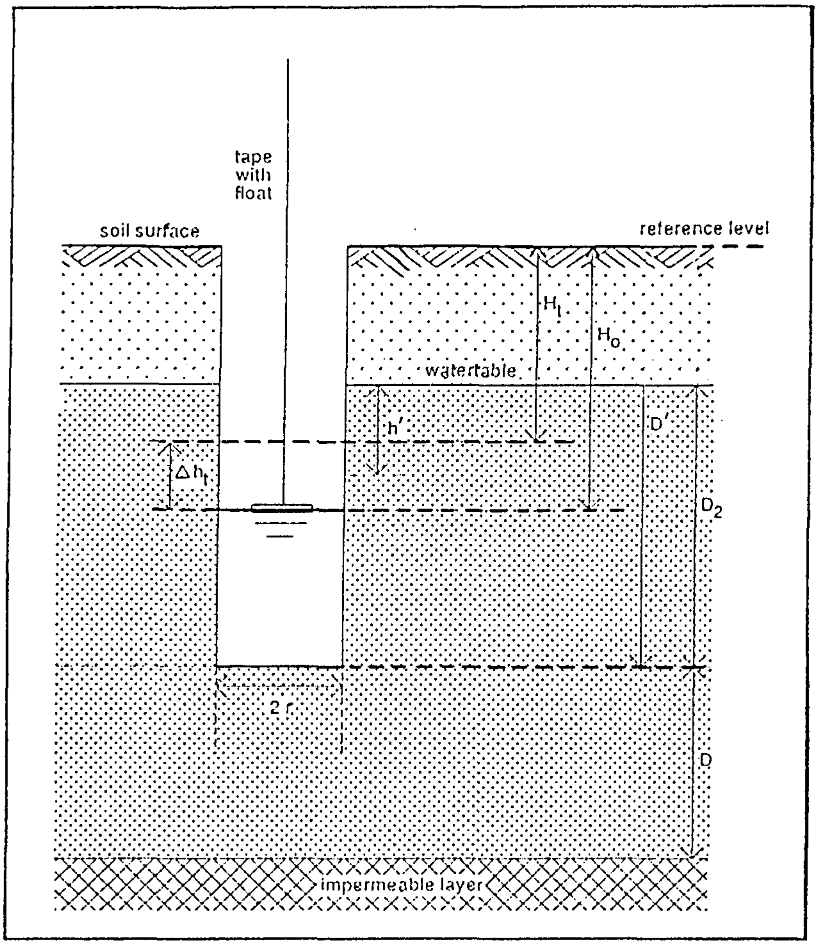

Figure 3.2 Factors applicable to the Ernst equation (Oosterbaan and Nijland, 1994).

4. SURFACE WATER

This section will deal with the results obtained from the sampling and analysis of surface water over the experimental period. In this context, surface water is taken to mean the estuary, irrigation water supplied from the Kouga dam, the water emanating from a surface agricultural drain, rainfall and dam releases into the estuary.

4.1 Irrigation Water

The irrigation water abstracted from the canal (discussed in section 3.4.1) is good quality water. The irrigation water was sampled on seven occasions and the average chemical constituents are given in Table 4.1.

In the appraisal of the suitability of irrigation water, not only must the salinity (EC) of the water be examined, but more important is the relative amount of sodium in the water. In the study area, although the irrigation water has a low EC (mean 32 $mS.m^{-1}$) under full quota irrigation (8 000 $m^3.ha^{-1}.a^{-1}$) 1.664 tons of salts will be added to each hectare of soil under irrigation on an annual basis. Thus from irrigation alone, 5 056 tons of salts will be added to the soil annually in the study area of 3 040 ha. Of this amount, 802.6 tons comprises sodium ions.

The quality of irrigation water is important from a soil perspective. Soil particles are negatively charged, with the magnitude of the charge varying with the amount of organic matter and clay present and the type of clay mineral. Certain cations (Ca, Mg , Na, K and H) are adsorbed onto clay particles and are exchangeable (Hoorn and Alphen, 1994). Gypsum ($CaSO_4$) is sometimes added to saline soils, or where sodic water is the main or only source of irrigation. The calcium ions from the gypsum are absorbed by the soil which then releases sodium ions. If crops are irrigated using water with a high sodium content, the sodium ions will be adsorbed onto clay particles, and thus large amounts of exchangeable sodium may be present in the soil (Hoorn and Alphen, 1994), even under increasing levels of applied gypsum (Bajwa and Josan, 1989). Gypsum can only alleviate problems where soil drainage is sufficiently good to allow leaching of the sodium ions from the soil.

Constituent	Value
Electrical Conductivity $mS.m^{-1}$	32.2
Total Alkalinity $mg.l^{-1}$	20.1
Calcium $mg.l^{-1}$	25.4
Magnesium $mg.l^{-1}$	29.9
Sodium $mg.l^{-1}$	33.2
Potassium $mg.l^{-1}$	1.7
Chloride $mg.l^{-1}$	128.8
Sulphate $mg.l^{-1}$	26.6
Nitrate (as N) $mg.l^{-1}$	0.3
Nitrite (as N) $\mu g.l^{-1}$	7.6
Total phosphorus (as P) $mg.l^{-1}$	0.3

Table 4.1 Mean chemical composition of irrigation water as abstracted from the irrigation canal at Green Acres.

Sodium has a dispersant effect on soil aggregates thereby affecting soil structure and permeability and lowering the infiltration capacity. The toxic side effects which sodium has on plants may however be experienced prior to soil structure breakdown. In the Loerie Flats area, all fields are irrigated by mobile, metal-arm sprinkler systems using water from the Kouga dam. Similar types of fertilizers are applied throughout the area and similar crops are grown on a rotation basis. Despite these similarities, certain fields, in the vicinity of borehole T and borehole D6, were abandoned because of previous poor crop yields and stunted crop growth. Such low crop yields can be attributed to soil salinization in these areas.

The percentage of sodium relative to magnesium and calcium can be used as an index to assess the suitability of irrigation water: ideally the sodium concentration should be lower than the calcium and magnesium concentration (Driscoll, 1986). The potential sodium hazard of irrigation water can be calculated using the sodium absorption ratio (SAR) formula:

$$SAR\ (mmol.l^{-1})^{0.5} \quad = \frac{Na}{(Ca + Mg)^{0.5}} \tag{4}$$

where Na, Mg and Ca are the sodium, magnesium and calcium concentrations respectively of the irrigation water in $mmol.l^{-1}$ (Bonn *et al.*, 1985; Department of Water Affairs and Forestry, 1993b).

Soils under irrigation are susceptible to the formation of surface seals, this reduces the infiltration rate of the soil. The SAR target guideline range for such soils, to ensure an adequate infiltration rate, is 0 to 1.5 $mmol.l^{-1}$ (Department of Water Affairs and Forestry, 1993b). The SAR of water used for irrigation in the Gamtoos agricultural area was calculated to be 1.05 $mmol.l^{-1}$. The EC value and SAR calculated for irrigation water has been used by the Food and Agriculture Organisation of the United Nations (FAO, 1985; cited in Hoorn and Van Alphen, 1994) to provide a table of guidelines for infiltration problems. By applying this to irrigation water abstracted from the canal at Green Acres, the study area falls within the category of 'slight-to-moderate' infiltration problems.

4.2 The Agricultural Drainage System

The agricultural drainage system has a twofold influence, namely (i) it influences groundwater quality in the vicinity of the drainage system - this aspect is discussed in section 5.3.1, and (ii) the agricultural effluent discharging into the estuary will impact on estuarine water quality. The drainage system was designed to receive agricultural runoff from adjacent fields and throughflow from the upper 1.2 m of soil. It was the responsibility of the farmer (assisted by the Department of Agriculture) to adequately drain his fields to the collection point of the drainage system.

A survey of the elevation of the land surface at site M showed a gradient of 0.85 m from borehole M4 to M3 (Figure 2.6). Soil tracer studies (Van Zyl, 1994) adjacent to the drainage system, at site M, showed a preferential drainage of subsurface water in the upper 1 m of the vadose zone towards the drainage system.

The movement and proportion of salts (chemical residues from fertilizers) and pesticides entering the drainage system is governed by the irrigation input and hydraulic conductivity of the upper 1.2 m of soil. Water will also percolate downwards forming an input to the underlying alluvial aquifer. These factors will thereby determine the nature of the runoff and leachate entering both the surface drainage system and groundwater system.

4.2.1 Water Quality at the Drainage Pipe

The drainage system is completed by the drainage pipe (P1 - Figure 2.6) which enters the Gamtoos estuary 18.5 km upstream of the river mouth. Discharge from the drainage pipe was variable in response to irrigation runoff and rainfall events. The average discharge, based on thirteen readings and omitting the maximum value from the calculation, was 3.4 $l.s^{-1}$. During dry conditions the lowest discharge recorded from the drainage pipe was 0.07 $l.s^{-1}$ on 3 April 1993. The maximum observed discharge was 11 $l.s^{-1}$ after a rainfall of 41.4 mm over the three days preceding measurement on the 13 November 1993 (Pearce and Schumann, 1994).

The range and mean chemical composition of the agricultural effluent measured at the drainage pipe is given in Table 4.2. Based on the average flow, and excluding the higher discharges which occurred during rainfall, the loading of nutrients to the estuary can be estimated. Taking the mean total phosphorus (as P) concentration of 0.7 $mg.l^{-1}$ (the maximum recorded at the drainage pipe was 3.35 $mg.l^{-1}$), a total of 75 kg of phosphorus would have entered the estuary on an annual basis. Similarly taking the mean nitrate (as N) concentration of 1.2 $mg.l^{-1}$ and nitrite (as N) concentration of 0.15 $mg.l^{-1}$ would yield an annual input of 129 kg of nitrate and 16 kg of nitrite to the estuary. These are conservative estimates and do not take into consideration the substantial inputs from surface runoff to the drainage system which could be expected during rainfall.

4.3 The Appearance of *Azolla filiculoides*

A dense mat of *Azolla filiculoides*, an invasive, aquatic macrophyte, was noted for the first time at Boschhoek on 16 March 1993. It is possible that *Azolla* was present upstream of Boschhoek prior to the commencement of sampling. The presence of *Azolla* is not necessarily indicative of elevated nutrients in the river. Because *Azolla* can assimilate atmospheric nitrogen via its symbiotic relationship with the alga *Anaboena azollae* (unlike other invasive macrophytes) it can proliferate in waters with low nitrogen values; on the other hand, plants also remove nitrate and therefore could yield low values in the water sampled. The nitrate (as N) content of water at Boschhoek was consistently below 0.01 $mg.l^{-1}$ and nitrite (as N) ranged from 0.2 to 67.0 $ug.l^{-1}$.

Chemical	Range	Mean
EC $mS.m^{-1}$	1027 - 5390	2211
Tot. Alkalinity $mg.l^{-1}$	196 - 753	316
Calcium $mg.l^{-1}$	174 - 2562	1052
Magnesium $mg.l^{-1}$	796 - 9105	4032
Sodium $mg.l^{-1}$	618 - 11785	4925
Potassium $mg.l^{-1}$	24 - 360	146
Chloride $mg.l^{-1}$	2891 - 23303	8391
Sulphate $mg.l^{-1}$	942 - 4457	2724
Nitrate (as N) $mg.l^{-1}$	<0.1 - 12.1	1.2
Nitrite (as N) $mg.l^{-1}$	0.004 - 0.7	0.15
Total phosphorus $mg.l^{-1}$	0.14 - 3.35	0.70

Table 4.2 Mean chemical composition of the agricultural discharge measured at the drainage pipe.

Azolla however has a high phosphate requirement and where *Azolla* is cultivated commercially, it often requires the supplemental input of phosphate fertilizers for adequate growth (Bieleski and Lauchli, 1992). The phosphate content of water at Boschhoek ranged from 0.04 to 1.02 $mg.l^{-1}$. According to Subudhi and Watanabe (1981) whilst many species of *Azolla* are able to grow normally at a minimum phosphorus concentration of 0.06 $mg.l^{-1}$, *Azolla filiculoides* and *Azolla* *caroliniana* require a higher phosphorus concentration to maintain normal metabolic activity. Rains and Talley (1979; cited in Subudhi and Watanabe, 1981) found that 0.34 $mg.l^{-1}$ of phosphorus was required for the maximal growth of *Azolla filiculoides* and *Azolla mexicana*. Watanabe *et al.*, (1980) state that a phosphorus concentration of between 0.13 and 0.43 $mg.l^{-1}$ is necessary for the growth of *Azolla pinnata* in paddy fields. The growth and production of *Azolla* will increase with a rise in the phosphorus content of the aquatic environment, up to a maximum concentration of 5 $mg.l^{-1}$. Beyond this level, *Azolla* will continue its uptake of phosphorus, but will not exhibit any increase in productivity (Subudhi and Watanabe, 1981).

Besides the basic nutrient requirements (phosphorus, nitrogen, potassium, calcium and magnesium) for growth, *Azolla* can adapt to suit environmental conditions and can maintain high vegetative reproductive rates at low nutrient concentrations. The optimal growth temperature for *Azolla* is 27.5^{0}C but it can grow at temperatures between 5 and 45^{0}C (Ashton and Walmsley, 1976). Conditions at Boschhoek were favourable for the growth of *Azolla* in that *Azolla* likes a certain amount of shading during growth, and this was adequately provided by the large railway bridge above the river; the sluggish water movement at Boschhoek enabled dense mats to cover the entire surface of the river. Between stations 10 and 7 *Azolla* mats were found close to those river banks which provide shade. As *Azolla* is a free-floating species it is little influenced by estuarine stratification or changes in water levels of the main water body. However, winds and waves break up *Azolla* mats. Downstream of station 7 the estuary widens and is more exposed to the action of south-westerly and north-easterly winds. The absence of *Azolla* downstream of station 7 is attributed to the influence of the wind and wave action, and the higher salinity of the estuary.

The presence of *Azolla* can lead to several negative side effects, namely:

- dense mats at the water surface prevent light penetration to underlying layers
- the growth physically hinders the passage of oxygen into the water, furthermore the death of *Azolla* plants results in their sinking to the bottom of the water column and the subsequent decomposition in the bottom waters will exacerbate any low oxygen condition (Ashton and Walmsley, 1976; Ashton *et al.*, 1986)
- a low dissolved oxygen content can result in the die-off of aquatic populations (see section 4.10).

4.4 Freshwater Input into the Estuary

4.4.1 Rainfall

Table 4.3 gives details of the monthly rainfall recorded at Hankey and Jeffreys Bay (Figure 2.1) over the three years 1992 to 1994 and the long-term mean for these areas (Weather Bureau, 1994). The bimodal nature of the rainfall pattern is evident, with most rain falling in autumn and spring (Heydorn and Tinley, 1980).

Month	Hankey				Jeffreys Bay			
	*Mean	1992	1993	1994	**Mean	1992	1993	1994
JAN	27.4	17.5	77.0	16.4	30.4	9.5	62.2	21.6
FEB	36.4	116.0	25.5	39.1	35.2	149.0	14.6	42.8
MAR	48.1	12.5	6.0	45.0	43.6	15.3	12.3	30.6
APR	37.1	25.0	59.4	47.8	42.6	37.2	86.2	26.0
MAY	35.2	16.5	27.7	30.5	50.2	37.5	26.6	35.9
JUNE	25.1	16.0	115.6	18.5	45.0	70.1	167.0	40.7
JULY	24.9	35.0	0	45.7	44.0	34.6	8.2	82.1
AUG	33.6	53.5	43.0	88.3	49.5	98.6	104.8	99.8
SEPT	37.8	0.3	143.7	12.5	45.3	36.2	189.1	27.5
OCT	46.9	109.5	18.5	40.0	52.4	105.9	10.9	48.5
NOV	42.7	100.0	53.4	5.0	45.8	117.5	31.0	15.8
DEC	27.4	0	72.3	107.5	25.8	7.0	61.1	105.5
TOTAL	422.5	314.3	642.1	496.3	509.6	469.9	774.4	576.8

Table 4.3 Rainfall (in mm) recorded at Hankey from *1892 to 1994, and at Jeffreys Bay from **1933 to 1994. The monthly totals for the years 1992, 1993 and 1994 show the variable rainfall during the study period (Weather Bureau, 1994).

Low rainfall regions tend to have a greater temporal variability than high rainfall regions. Considerable variability is evident in the Hankey - Jeffreys Bay region as shown in Table 4.3. In 1992 the mean annual rainfall was 108.2 mm below the twenty year running mean for Hankey, the main deviation from the average rainfall occurred over the seven month period from March to September when in 1992 a total of 158.8 mm of rain was recorded, some 83 mm below the twenty year running mean for the same months. The rainfall of 1993 was 219.6 mm above the long-term mean, with 153.6 mm above the average rainfall occurring in the previously mentioned seven month period.

4.4.2 Dam Releases

The presence of the Kouga dam (Figure 2.1) has reduced the average freshwater flow into the Gamtoos estuary to an estimated 1 $m^3.s^{-1}$ or less. Water is not released from the Kouga dam into the Kouga River, it is released into the irrigation canal, however, natural overtopping of the dam may occur when the dam is at capacity. There was no overtopping of the Kouga dam in 1992, in 1993 the dam overflowed from 25 September to 3 November. During 1994 overtopping occurred from 17 August to 2 November. There are no gauging stations on the Gamtoos River and the quantity of water overtopping the dam is not monitored. The Gamtoos River also receives the saline and turbid Karoo-Formation water from the Groot River. Karoo shales form clays and silts rendering the water more turbid than that associated with sandstone. The Groot River in turn receives less saline discharges from sandstone sources of the Sand and Wit Rivers and releases from the Beervlei dam. River flow is measured at a weir on the Groot River where water quality is variable depending on the main source of flow. The monitoring of the quality and quantity of freshwater inputs to the Gamtoos River and estuary would be complex, to cope with a complex situation. As it happens, the monitoring done is rather simple and inadequate.

Water is released frequently from the Loerie dam to make freshwater available for the farmers downstream. There was also natural overtopping of the dam in 1992, 1993 and 1994.

4.4.3 Freshwater Inflow

Shifts in the hydrology of an estuary are often imperceptible, yet directly impact on the physical structure and functioning of the aquatic environment. According to Newson (1994) reduced freshwater flow to an estuary can adversely affect its ecology as there is an interdependency of water quality and quantity, adjacent land-based activities and downstream effects.

Because the viability and productivity of estuaries is influenced by freshwater flow and, in view of the growing need to utilise all available water resources in South Africa without disturbing ecologically sensitive areas, Jezewski and Roberts (1986) examined the freshwater requirements of a number of estuaries in South Africa. The freshwater requirements they established were based on overcoming the loss as a result of evaporation and flooding requirements. Flooding requirements were defined by Jezewski and Roberts as the quantity of water needed to keep the tidal inlet open, flush out accumulated sediment and flood the wetlands bordering the estuary. Their calculations were based on a mean annual rainfall of 600 mm. According to Jezewski and Roberts the Gamtoos estuary requires a freshwater input of 36.800 x 10^6 $m^3.a^{-1}$. Of this amount 3.504 x 10^6 $m^3.a^{-1}$ is the flooding requirement and the remainder the evaporative requirement of the estuary. These calculations do not take ecological requirements into account. Comment is made on these figures given by Jezewski and Roberts in the context of the findings of this report, in section 6.4.

The uppermost part of the tidal influence is at Boschhoek (shown on Figure 2.2, where the railway bridge crosses the Gamtoos River). It is at this point that the estimated inflow of freshwater to the estuary is approximately 1 $m^3.s^{-1}$. A consistent flow of 1.16 $m^3.s^{-1}$ throughout the year would produce the estimated required freshwater flow of 36.800 x 10^6 $m^3.a^{-1}$ given by Jezewski and Roberts (1986).

The results to follow refer to specific measurements of flow during the four monitoring periods. Daily rainfall totals pertaining to the four measurement periods are detailed in Table 4.4. Figure 4.1 shows the varying estimated freshwater input into the estuary in response to the rain over the measurement period during November and December 1992, as well as June 1993. There are no reliable values for the maximum flow reached during the rain in the catchment prior to the November measurements taking place, but it is likely to have been more than 10 $m^3.s^{-1}$; the figure then shows the gradual decrease over the almost three week period to an estimated ambient flow level of 1 $m^3.s^{-1}$.

An estimate of the extent of the volume of freshwater movement in the upper reaches of the estuary over a period of a day is given. Based on the cross-sectional measurements of Reddering and Scarr (1990b), taking a mean width of 30 m and mean depth of 1 m, a volume flow of 1 $m^3.s^{-1}$ will penetrate a distance downstream of approximately 2.9 km per day, with a total inflow of 86.4 x 10^6 m^3. A volume inflow of 10 $m^3.s^{-1}$ can reach the mouth. The Gamtoos estuary broadens and deepens markedly downstream particularly beyond station 7, and again beyond station 4 (see Figure 2.4). The mean volume of the estuary is about 3.6 x 10^6 m^3, which means that even at a volume inflow of 10 $m^3.s^{-1}$, the flushing time for the estuary would be in excess of four days (Schumann and Pearce, 1997). The implication of these rough estimates is that a volume inflow of 10 $m^3.s^{-1}$ will flush out the upper reaches of the estuary within a tidal period. In the lower reaches, because of the large volume of resident water, a much greater freshwater input will be required for the same effect in this region.

Sampling date (a)	**No. of dry days preceding (a)**	**Quantity of rain (mm) and date thereof**	
26 November 1992	0	15; 6	24, 25 November
4 December 1992	9	-	
10 December 1992	15	-	
1 April 1993	14	-	
6 April 1993	3	8; 27.4	2, 3 April
7 June 1993	4	28.4	3 June
14 June 1993	0	87.2	10 - 13 June
7 November 1993	18	-	
13 November 1993	2	0.5; 38.1; 2.8	9 - 11 November

Table 4.4 Number of dry days preceding sampling, quantity of rain and the days on which rain fell, prior to- and during measurement periods.

An estimate of flow based on the measurements made at station 10 during low tide on 1 April 1993, following fourteen rain-free days was approximately 0.5 $m^3.s^{-1}$. On 6 April, three days after rainfall the freshwater inflow to the estuary was just above 1 $m^3.s^{-1}$.

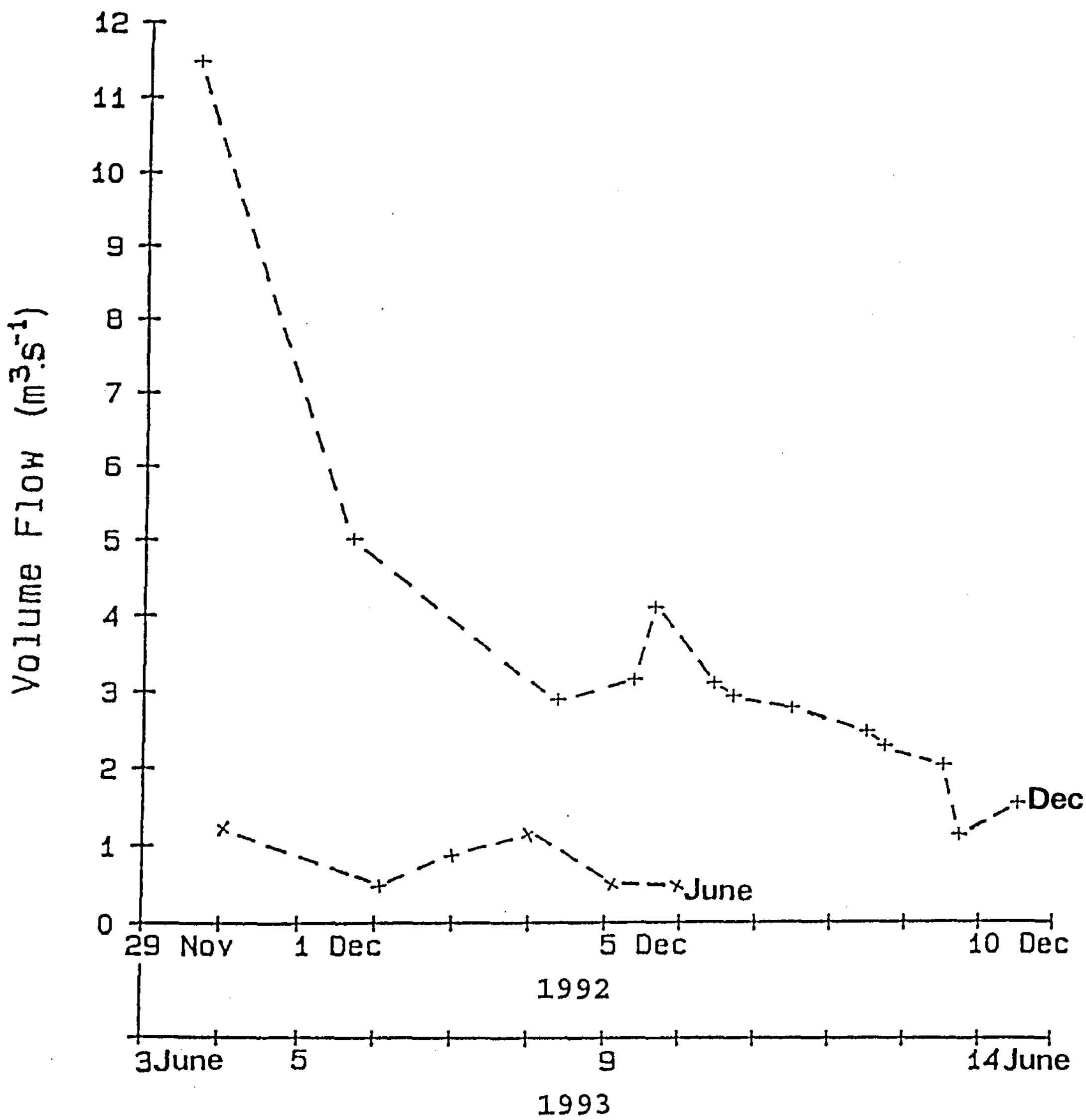

Figure 4.1 Freshwater input into the estuary in November/December 1992 and June 1993.

Figure 4.1 also shows the estimated freshwater input to the estuary from 4 June 1993, where flow was estimated at just over 1 $m^3.s^{-1}$ following 28.4 mm of rain. The input to the estuary rose above 1.3 $m^3.s^{-1}$ on 8 June which is probably related to a freshwater release or rainfall higher up in the Groot River catchment area. By 10 June the flow had dropped to an estimated

0.5 $m^3.s^{-1}$. Instrumentation was of insufficient capacity to measure the flow which followed the 87.2 mm of rain between 10 and 13 June. A rough estimate is that flow exceeded 100 $m^3.s^{-1}$. At a discharge of 100 $m^3.s^{-1}$, it would take less than one day to replace the total volume of the estuary.

The freshwater input to the estuary on 7 November 1993 was an estimated 0.8 $m^3.s^{-1}$, this followed eighteen rain-free days. The freshwater discharge rose to above 2 $m^3.s^{-1}$ (13 November) following the rainfall between 9 and 11 November as detailed in Table 4.4.

4.5 Ocean-Estuary Exchanges

Ocean-estuary exchanges are essentially dependent on the configuration of the estuary mouth. If the estuary is to be flushed of pollutants, then it is clear that the ocean-estuary exchanges should be operative. Constriction in the tidal inlet caused by the flood-tidal deltas restricts the ocean-estuary exchanges. To give an indication of this influence the water level fluctuations in the sea (South African Tide Tables, 1992) and in the estuary at spring and neap tides in November and December 1992 are plotted in Figure 4.2. The dominant M_2 tidal variation is apparent, but the effect of the mouth constriction was considerable, with the amplitude of the tidal fluctuations immediately within the estuary only about 30% of those in the adjacent sea. An investigation in 1989 (Institute for Coastal Research, unpublished data) found that the ocean tides were attenuated by 50 to 60% immediately inside the estuary, which means that the exchanges through the tidal inlet had decreased considerably over the intervening three to four years.

Time delays in tidal variations occur in the mouth region, with the peak of the tide exhibiting a delay of about one hour, and the trough a delay of almost three hours compared to that in the adjacent ocean. This asymmetric tidal variation is typical of flood-tidal estuaries (Aubrey and Speer, 1985; MacKay and Schumann, 1991), and can be ascribed to the generation of an M_4 overtide. Further upstream the time delays relative to the ocean increased further, while the tidal asymmetry was also accentuated, and the amplitudes increased slightly as the estuary channel decreased in width.

Coherency and phase spectra were determined from the time series data from combinations of the measurement sites. As expected the coherency was very high at the M_4 tidal period (12.42 hours), leading to correspondingly accurate values of the phase angles (Jenkins and Watts, 1968). A delay of 2 hours and 11 minutes was found between the ocean and the first measurement point at station 1, and then 11.5 minutes from station 1 to 4, and 5.5 minutes between stations 4 and 9; this also agreed with the 16 minute delay between stations 1 and 9. The M_4 tidal propagation remained locked to the M_2 tide between stations 1 and 4, however the lag time for the M_4 tide increased by 6 minutes between stations 4 and 9; this may have been caused by the shallow sand banks between stations 7 and 8.

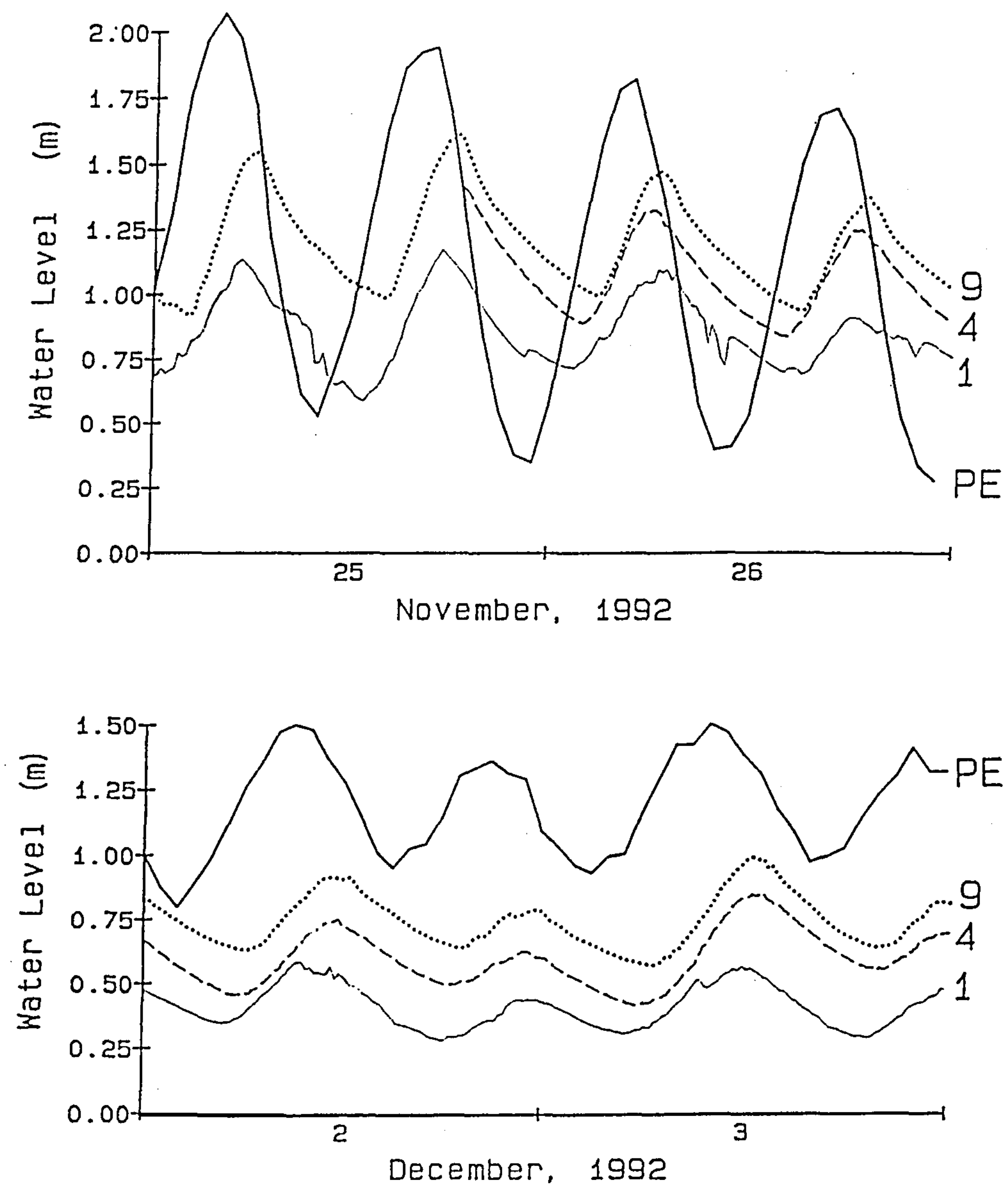

Figure 4.2 Sea water level fluctuations as measured at Port Elizabeth harbour (PE) during November and December 1992. Corresponding tidal fluctuations within the estuary, at stations 1, 4 and 9 are also shown.

An estimate of the tidal prism: at spring and neap tides using the tidal amplitudes in Figure 4.2 and the estuary widths and depths measured during 1989, an amplitude of 55 cm at spring tide yields a tidal prism of about 9.6 x 10^6 m^3. At neap tide an amplitude of 25 cm gives a value of around 4.4 x 10^6 m^3.

4.6 Estuarine Structures and Variability

The sub-sections to follow deal with the salinity, temperature, turbidity and dissolved oxygen profiles within the estuary in response to freshwater inputs. Section 4.7 examines the nutrient characteristics throughout the estuary, whilst the presence of pesticides is discussed in section 4.8.

4.6.1 Salinity Structures

The salinity structures within the Gamtoos estuary are presented in the context of freshwater inputs to the estuary. For this reason Table 4.4 provides a summary of the rainfall characteristics associated with each measurement period. Releases from the Loerie dam and overtopping of the Kouga and Loerie dams are discussed in section 4.4.2. Selected salinity profiles of the estuary during each of the four periods are presented.

Between 1 and 12 November 1992, 79 mm of rain was recorded at Hankey, this was followed by dry conditions until measurements commenced on 24 November. 15 mm of rain was recorded at Hankey on 24 November and 6 mm on the 25th. The effect of the freshwater input to the estuary following the rainfall event can be seen in the salinity profile of the estuary shown in Figure 4.3. At low tide, water with a salinity of less than 13 reached the sea in the surface flow. At high tide (Figure 4.3) the inflow from the sea caused substantial mixing at the freshwater/saltwater interface and prevented further seaward movement of the interface. The main halocline extended just downstream of station 7 during both high and low tide.

Dry conditions prevailed for the remainder of November and December 1992, during which time the halocline moved upstream as shown in Figure 4.4. By 4 December (neap low tide - Figure 4.4) the surface layer was relatively thin and salinity at the inlet had risen to 14. Discharge from the estuary continued to decrease and a comparison of low and high tides on 4 December (Figure 4.4) showed that a higher salinity water moved progressively upstream. On 10 December (Figure 4.5) at spring low tide the freshwater flow had decreased further and the surface water entering the sea had a salinity greater than 25, though the upstream position of the main halocline had not changed much. At high tide station 1 exhibited a salinity of greater than 34 and almost vertically homogeneous conditions occurred right up to station 4, a salinity gradient occurred seawards and saltwater penetration almost reached station 8.

A calculation using the tidal prism and estuary dimensions indicates that at spring tide the volume of sea water entering the estuary from low to high tide would have extended to the vicinity of station 2. The salinity structures in Figure 4.3 tend to confirm this estimate, with most of the variation and backing up of freshwater occurring in the lower estuary. The sea water input at the following neap tide on 4 December was much less, and the structures at the next spring tide on 10 December confirm the more important effect of the sea water input relative to the freshwater input, at spring tides.

The measurements of 1 April 1993 (neap tide - Figure 4.6) were preceded by fourteen dry days, and only 6 mm of rain fell during the first two weeks of March. According to Figure 4.6 the estuary was stratified at low tide. The 10 isohaline extended from the bottom waters upstream of station 8 to the surface layers downstream of station 7. Station 9 had salinity values ranging from 1.7 (surface) to 8.0 (bottom). Station 10 could not be reached because the river was too shallow. Stratified conditions are still clearly visible during high tide in Figure 4.6 although less so for the upper estuary upstream of station 7. The freshwater nature of the upper estuary can be attributed to the aggraded stretch upstream of station 7. The highly sinuous thalweg of deeper water through the aggraded portion permits a limited amount of penetration by saline waters. On 2 April after 8 mm of rain fell followed by 27.4 mm the next day, the CTD and DO equipment gave problems during wet conditions and repairs delayed further measurements until 6 April. Despite some indication of a freshwater input in the upper reaches, the marked effect of spring high tide on the 6 April is evident in Figure 4.7, with vertically homogeneous conditions predominating and extending as far as station 10.

Estuarine measurements recommenced on 7 June (Figure 4.8), three days after spring tide. Rainfall occurred four days prior to these measurements when 28.4 mm was recorded at Hankey. The 10 isohaline is in the vicinity of stations 8 and 9 at low tide. Stratified conditions are apparent with the saltwater wedge extending to station 10, with the 25 isohaline extending from the surface waters downstream of station 4 to station 7. At high tide the 10 isohaline was pushed back, with not much change in the upper estuary. Bottom waters at station 9 and 10 had salinities of 22 and 18 respectively.

Between 10 and 13 June 87.2 mm of rain was recorded. On 12 June a record maximum quantity of rain fell within a twenty-four hour period, for the one hundred and two years and sixty-one years of data for Hankey and Jeffreys Bay respectively. Estuarine conditions on 14 June, are given in Figure 4.9 which shows the freshwater input to the estuary following the extensive rains. Initially (14 June, which was two days after neap tide) the freshwater appeared to flow mainly as a layer, to a depth of almost a metre, all the way to the sea, without mixing at the bottom of the freshwater layer. A strong freshwater input from the Loeriespruit tributary at station 5 occurred and is particularly noticeable in the salinity profile in Figure 4.9 at high tide. During this time the saltwater wedge, identified by the 25 isohaline, was confined to the lower layers of stations 1 to 7. Unfortunately, with no gauging weir, it is not known when the main freshwater runoff reached the estuary, but conditions on 16 June (Figure 4.10) are markedly different from those of the 14 June with freshwater having flushed out the upper reaches. It is not known how long it took for the 'normal' (non-rainfall) estuarine stratification, similar to that of November and April, to be restored because the wet weather again hampered instrument functioning and readings ceased on 17 June.

During early to mid-October, 18.5 mm of rain was recorded at Hankey but no rain fell between 21 October and 9 November. However, overtopping of the Kouga dam following rainfall in the catchment area, occurred between 25 September and 3 November. This freshwater input was evident in the salinity profile of 7 November 1993, as shown in Figure 4.11, where the salinity at stations 7 to 10 was below 1. The salinity profile on 7 November, shows that the main saltwater wedge, as well as the 10 and 18 isohalines extended to stations 4 and 5. The continued freshwater input was still evident during low tide on 13 November (Figure 4.12). At spring high tide however, the saltwater pushed upstream to station 7 and caused the salinity to rise above 6. The profiles of November 1993 can be interpreted in a limited way only because the measurements were made on two consecutive days over spring and neap tide. Measurements differed from the preceding periods in that only stations 4 to 10 were monitored. The measurements taken were to provide some degree of comparison with those made the previous November/December.

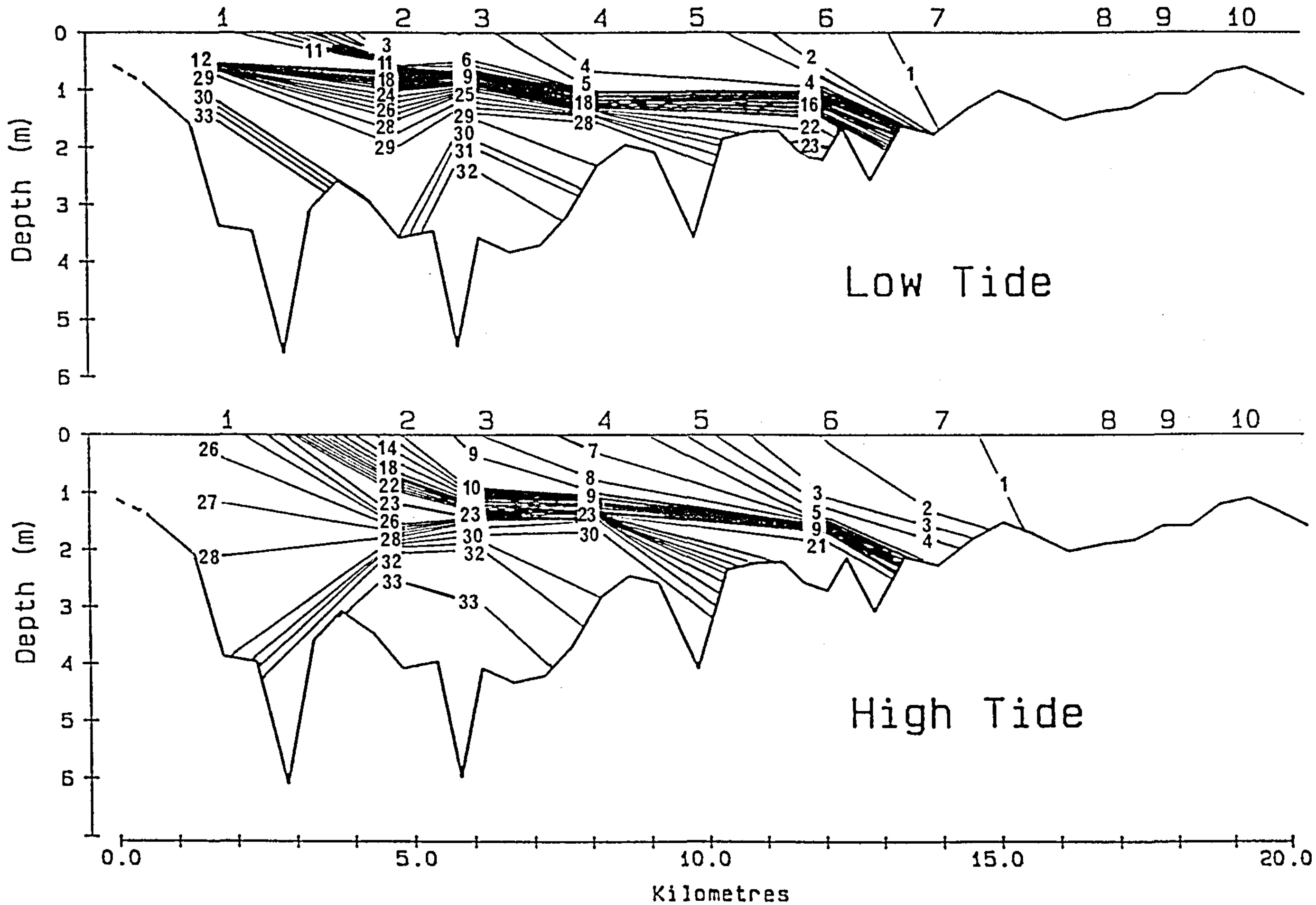

Figure 4.3 Salinity profiles at low and high tide on 26 November 1992.

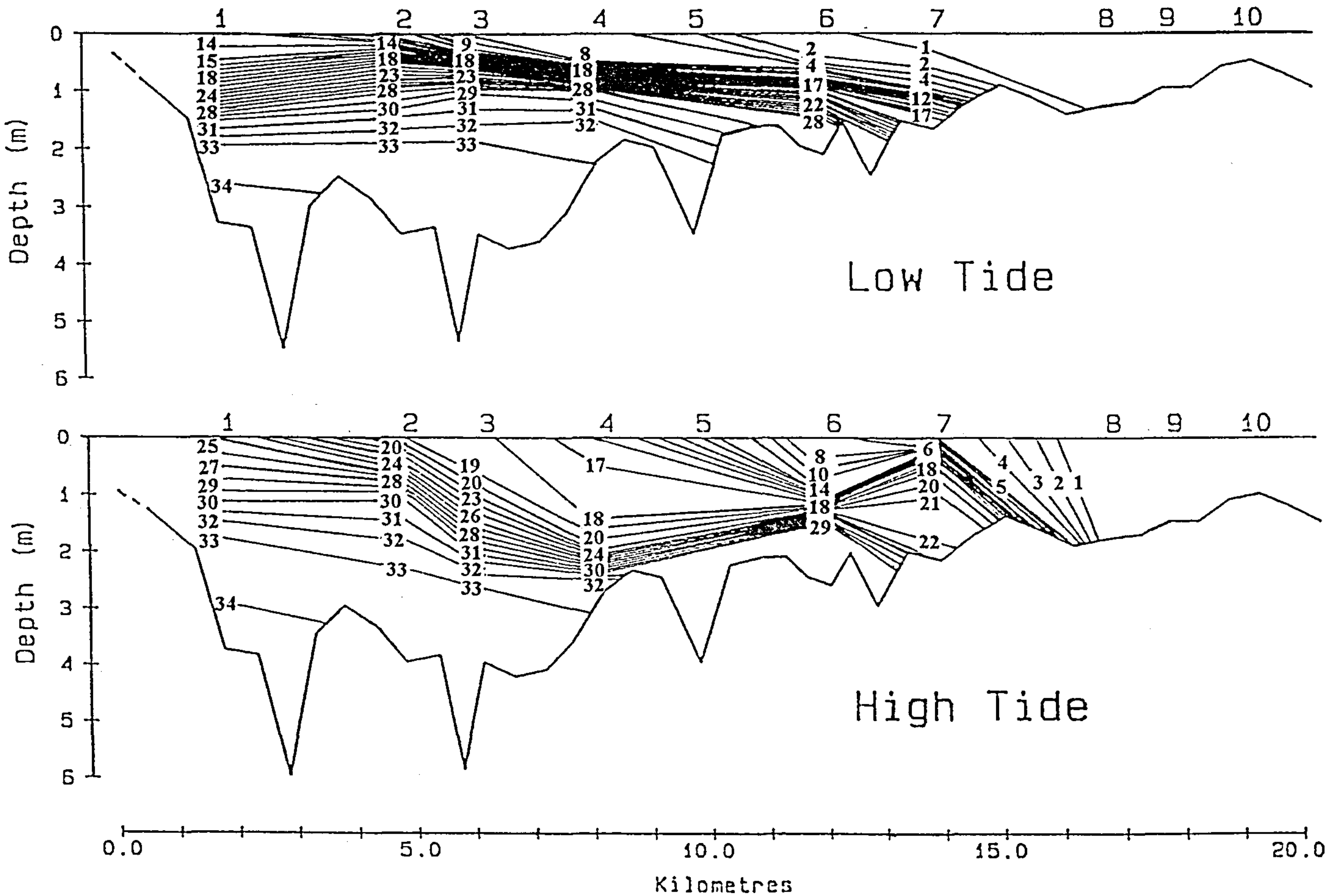

Figure 4.4 Salinity profiles at low and high tide on 4 December 1992.

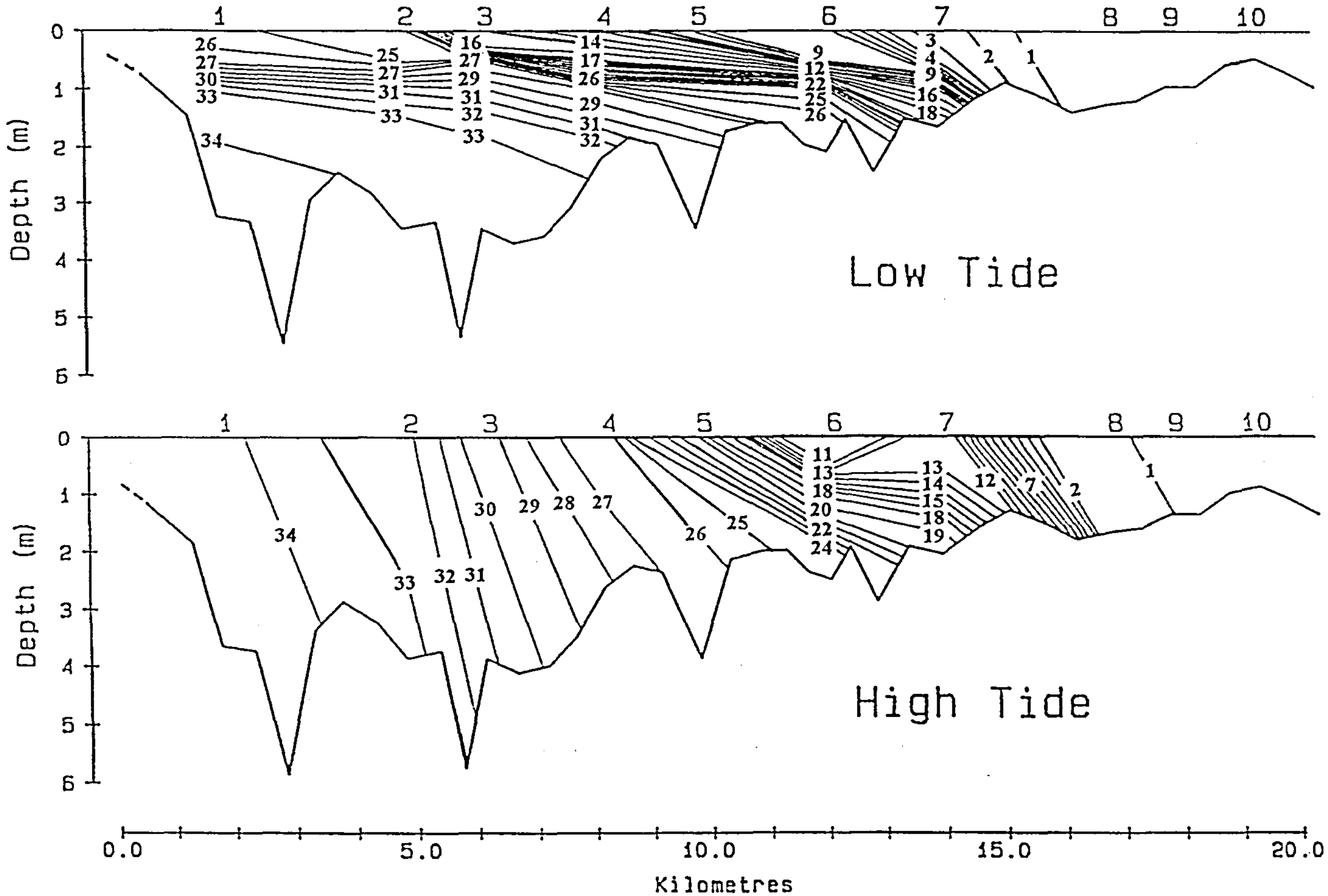

Figure 4.5 Salinity profiles at low and high tide on 10 December 1992.

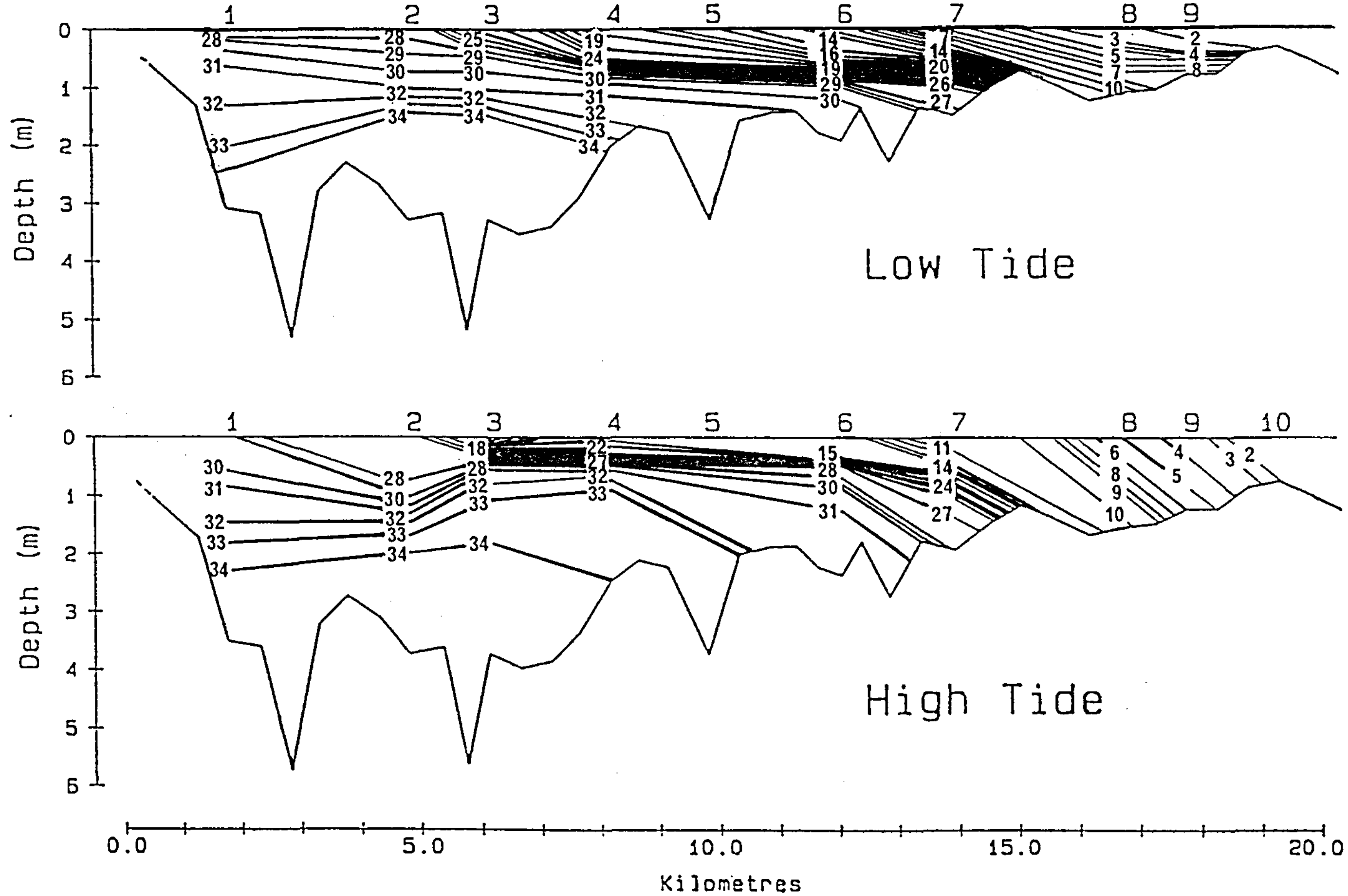

Figure 4.6 Salinity profiles at low and high tide on 1 April 1993.

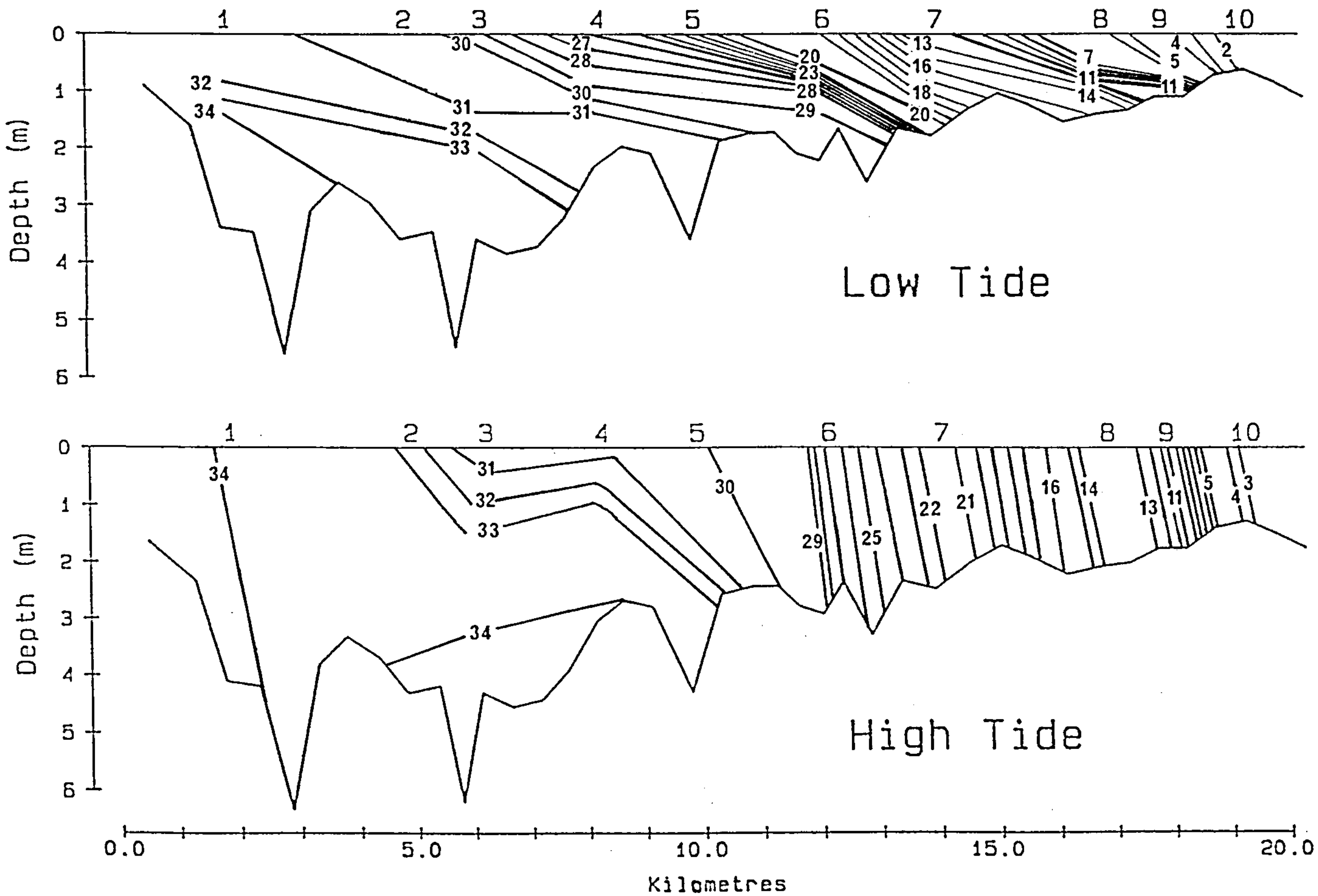

Figure 4.7 Salinity profiles at low and high tide on 6 April 1993.

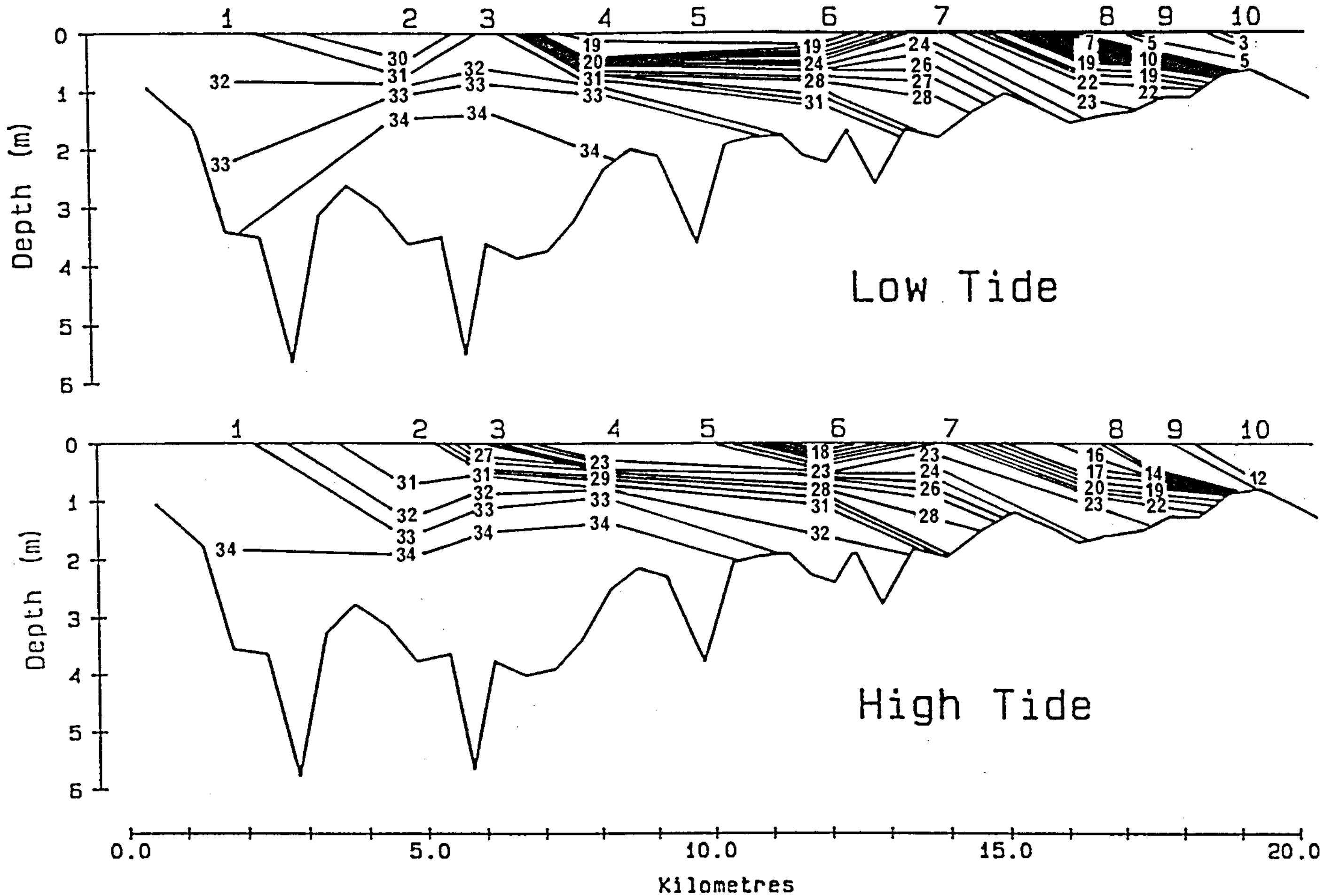

Figure 4.8 Salinity profiles at low and high tide on 7 June 1993.

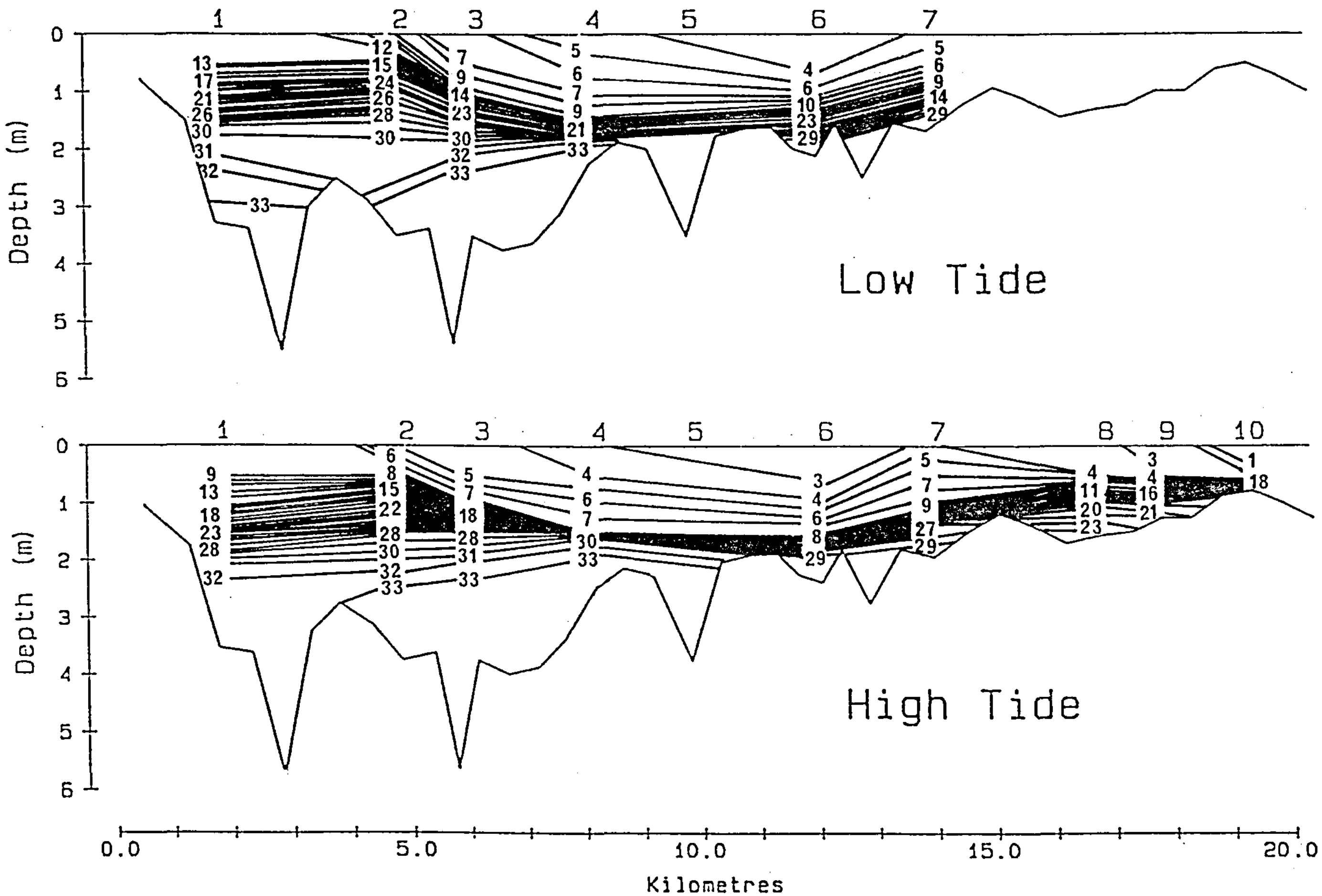

Figure 4.9 Salinity profiles at low and high tide on 14 June 1993.

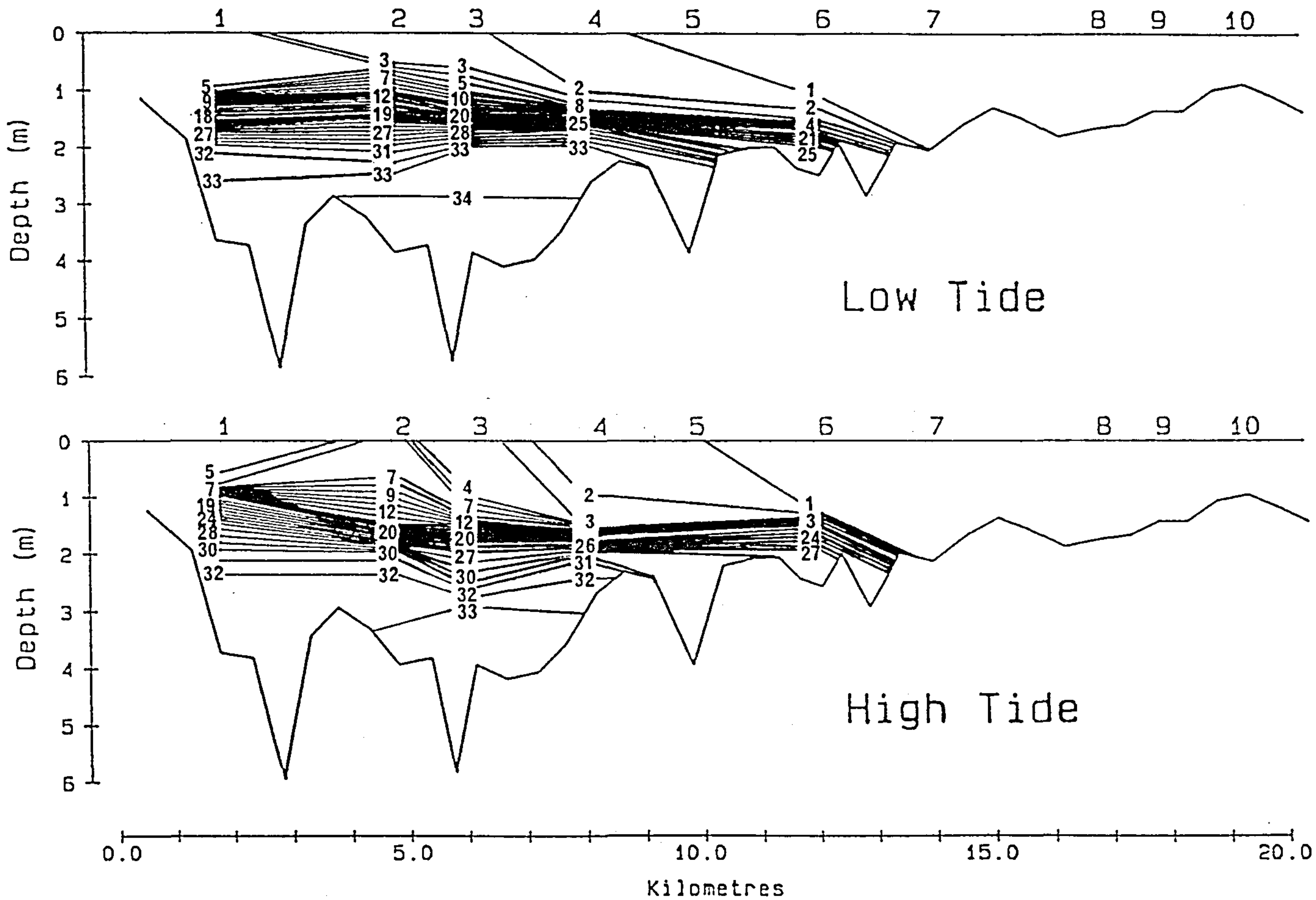

Figure 4.10 Salinity profiles at low and high tide on 16 June 1993.

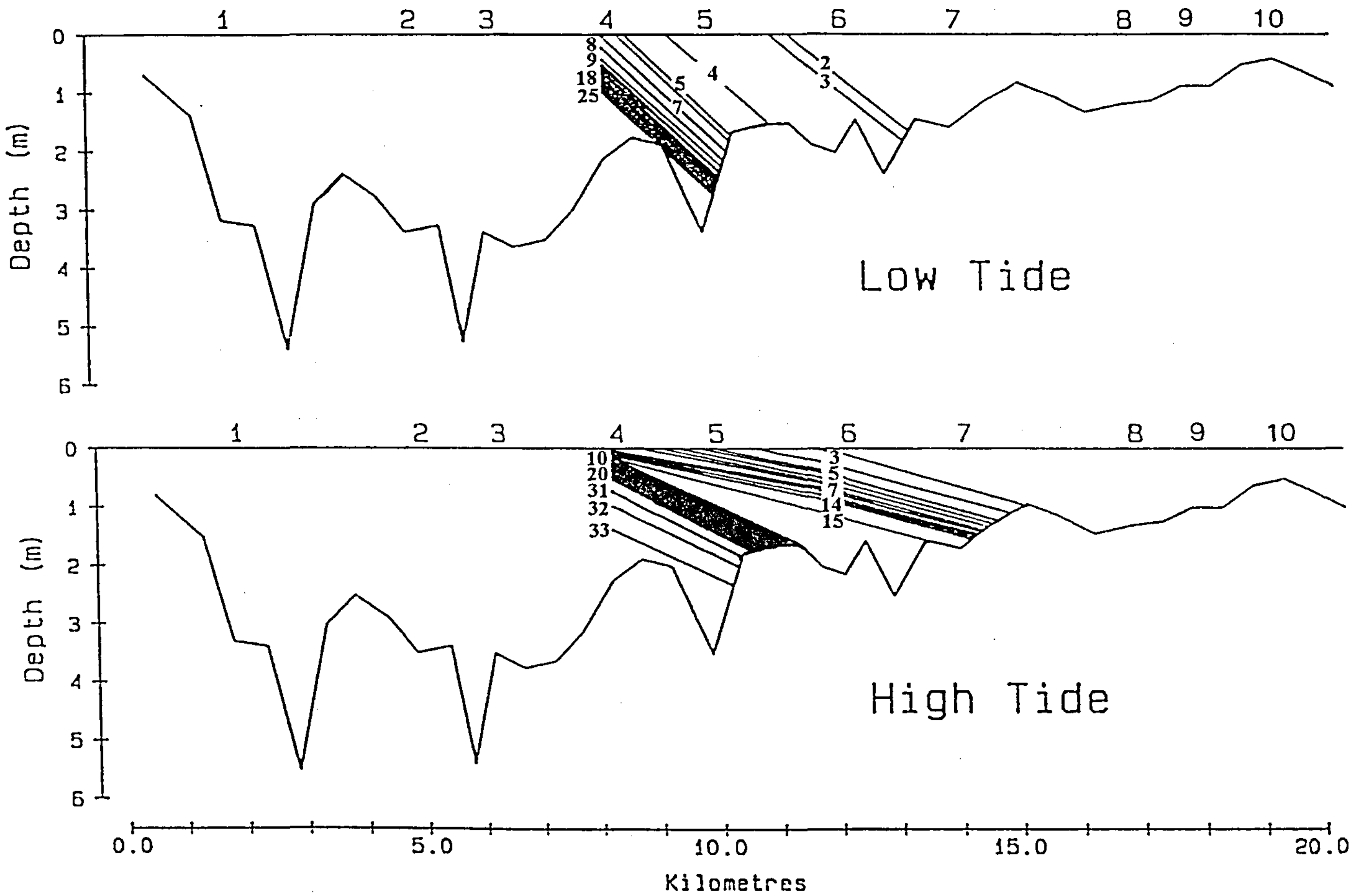

Figure 4.11 Salinity profiles at low and high tide on 7 November 1993.

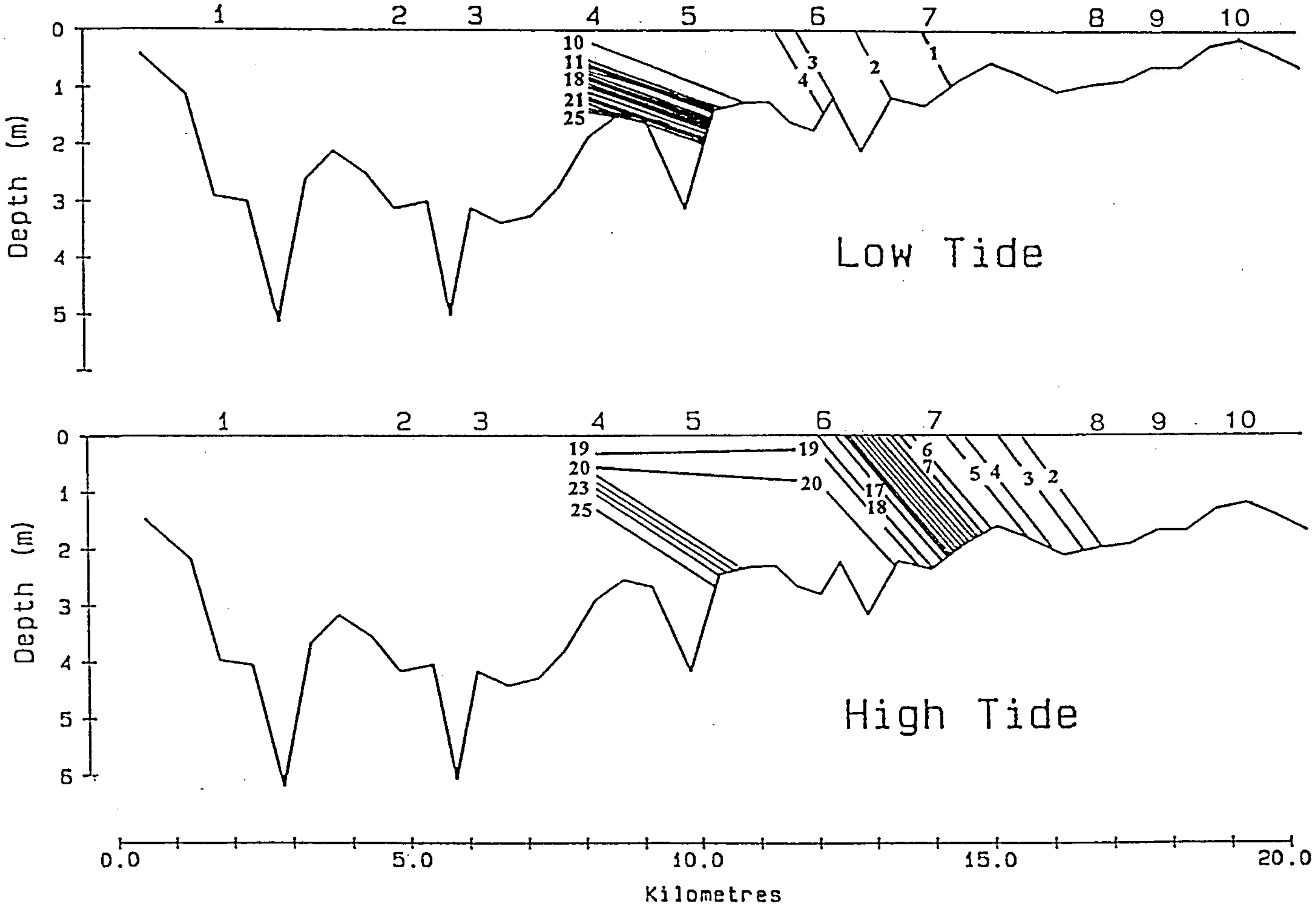

Figure 4.12 Salinity profiles at low and high tide on 13 November 1993.

4.6.2 Temperature Structures

Temperature structures within an estuary are governed by insolation, conduction, evaporation, rainfall and the ratio of tidal inflow to river discharge. The temperature of the aquatic environment will in turn influence other parameters, for example, as temperatures increase the solubility of oxygen in water will decrease. Also in the presence of certain pollutants, an increase in temperature will increase the toxic effect of the pollutant (Department of Water Affairs and Forestry, 1993a).

The temperature of the sea water entering the estuary can change substantially from day to day, particularly in summer when upwelling initiated by easterly winds along the coast can bring in water 8 to 10^{0}C colder than ambient summer values (Schumann *et al.,* 1988; Schumann and De Meillon, 1993).

On 26 November 1992 the temperature of the freshwater entering the head of the estuary was 21.7^{0}C. In contrast, it is apparent that the sea water entering the estuary had a temperature of around 17^{0}C. Figures 4.3 and 4.13 show that at high tide near vertically homogeneous salinity and temperature (17.0 - 17.5^{0}C) respectively prevailed at station 1. It is not clear whether this cold water can be related directly to upwelling, since westerly winds prevailed over the two days preceding the measurement. Goschen and Schumann (1995) found that the western sections of the large south coast bays react differently to those along straight portions of the coast, and that westerly winds can produce cold water along these coasts. The following day (27 November) temperatures at station 1 at high tide had risen, ranging from 20.8^{0}C at the surface and decreasing to 19.2^{0}C at a depth of 3 m. The impact of sudden changes in temperature on estuarine biota is discussed in section 4.10.

Water temperature is a non-conservative parameter, altered by solar radiation during the day. Results show that from 4 to 10 December 1992 (Figure 4.14) the temperature of the aquatic environment in the upper estuary was at least 1^{0}C warmer during the afternoon (shown as high tide in Figure 4.14) than in the morning. At shallower, more exposed stations (i.e. less shading vegetation) the upper layers of the water column were almost 2^{0}C higher by late afternoon in the upper reaches of the estuary. This can be attributed to the warm air temperatures in the Gamtoos valley, cloud-free skies and restricted tidal influence in the upper reaches of the estuary.

On 1 April 1993 (Figure 4.15) an input of marine water with a temperature in the vicinity of 21^{0}C was evident in the lower estuary. During low tide on this date stratified conditions were apparent. Surface waters upstream of station 4 were below 22^{0}C, whilst the bottom waters upstream of station 6 had temperatures greater than 25^{0}C. The salinity profile of 1 April 1993 at low tide (Figure 4.6), shows the input of freshwater in the surface layers of the upper estuary. The temperature of these surface layers is thus determined by the temperature of the freshwater input, which was below 22^{0}C. The warmer bottom waters could be a remnant of warmer water which persisted as a result of the limited tidal penetration and mixing of the freshwater input. The input of marine water was particularly evident during high tide and resulted in almost vertically homogeneous temperatures which increased upstream. Although the temperature profile exhibited a fairly mixed situation, the salinity profile during this time showed some degree of stratification. Solar radiation during the day led to the warming of the surface layers in the upper estuary by almost 2^{0}C (Figure 4.15 - high tide), although the warmer bottom water remained at station 7. The increased mixing during spring high tide on 6 April (Figure 4.16) led to more vertically homogeneous conditions ranging from less than 19^{0}C at station 1 and increasing upstream to a temperature of 22^{0}C at station 10.

On comparison of the temperature profiles of Novembers 1992 and 1993 with that of June 1993 (Figures 4.17 to 4.19), as expected, distinct seasonal differences were apparent. Temperatures at the head of the estuary were about 15^{0}C in June and 24^{0}C in November and at the mouth where temperatures ranged from 14^{0}C (June) to 20^{0}C (November). Figure 4.17 shows that the temperature of surface water in the upper estuary was less than 13^{0}C (high tide), with bottom water greater than 17^{0}C. The large freshwater input following the June 1993 heavy rainfall appeared to have a temperature of around 14^{0}C (as seen in Figure 4.18) whereas that of the marine water was around 16^{0}C. Figures 4.17 to 4.19 show that generally in the lower estuary, there was little thermal structure apparent, with a change of less than 2^{0}C over the entire water column.

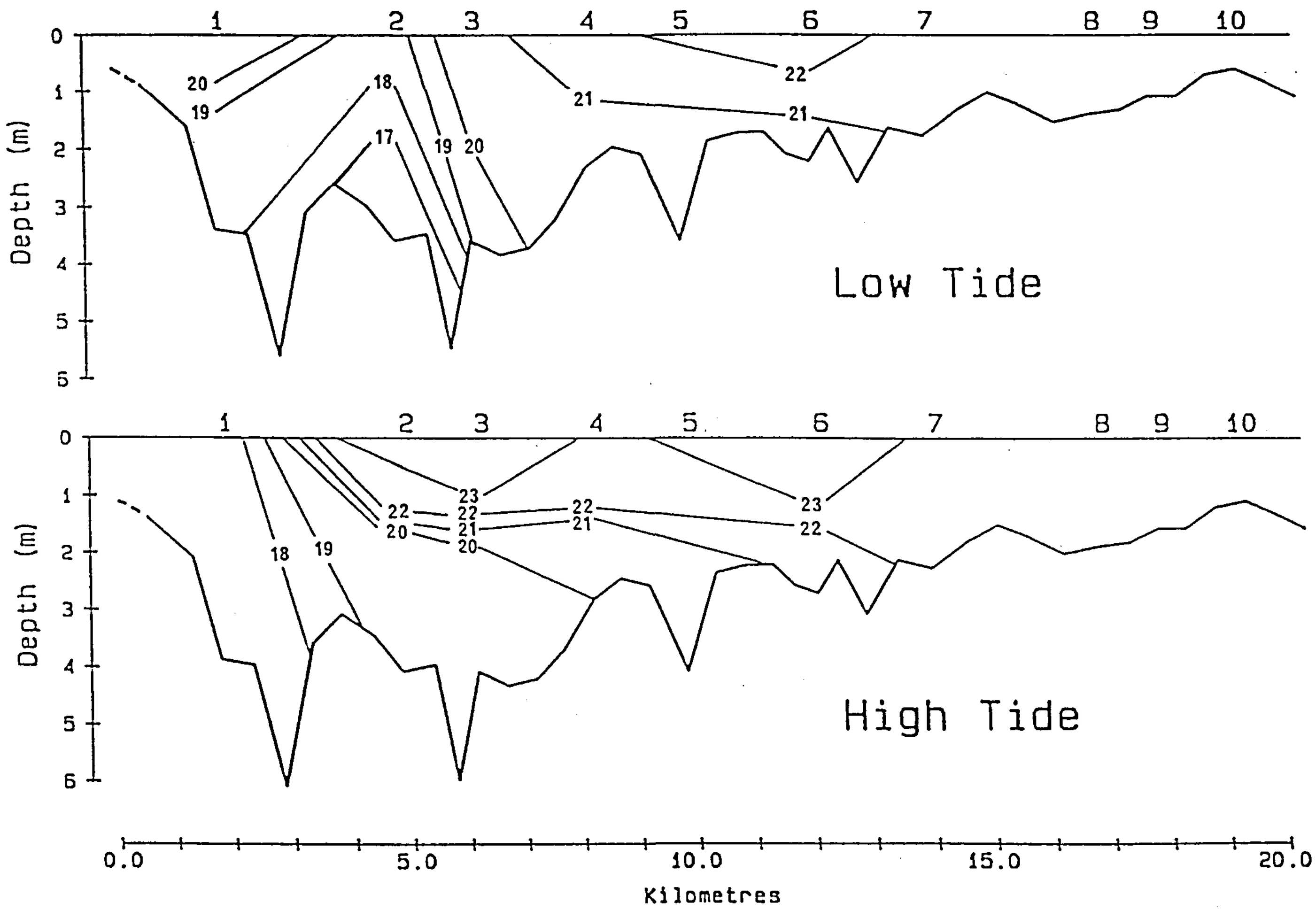

Figure 4.13 Temperature (°C) profiles at low and high tide on 26 November 1992.

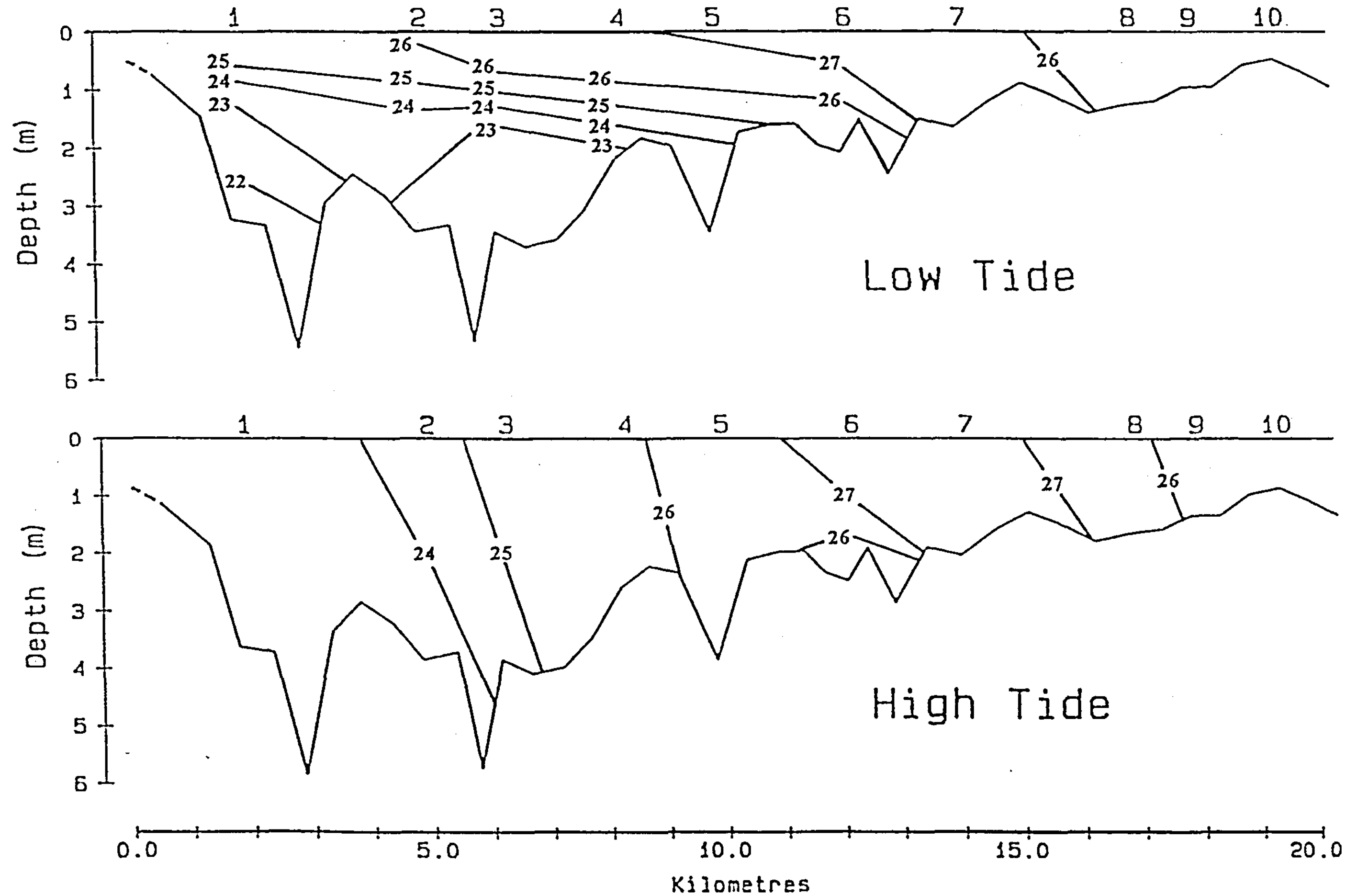

Figure 4.14 Temperature (°C) profiles at low and high tide on 10 December 1992.

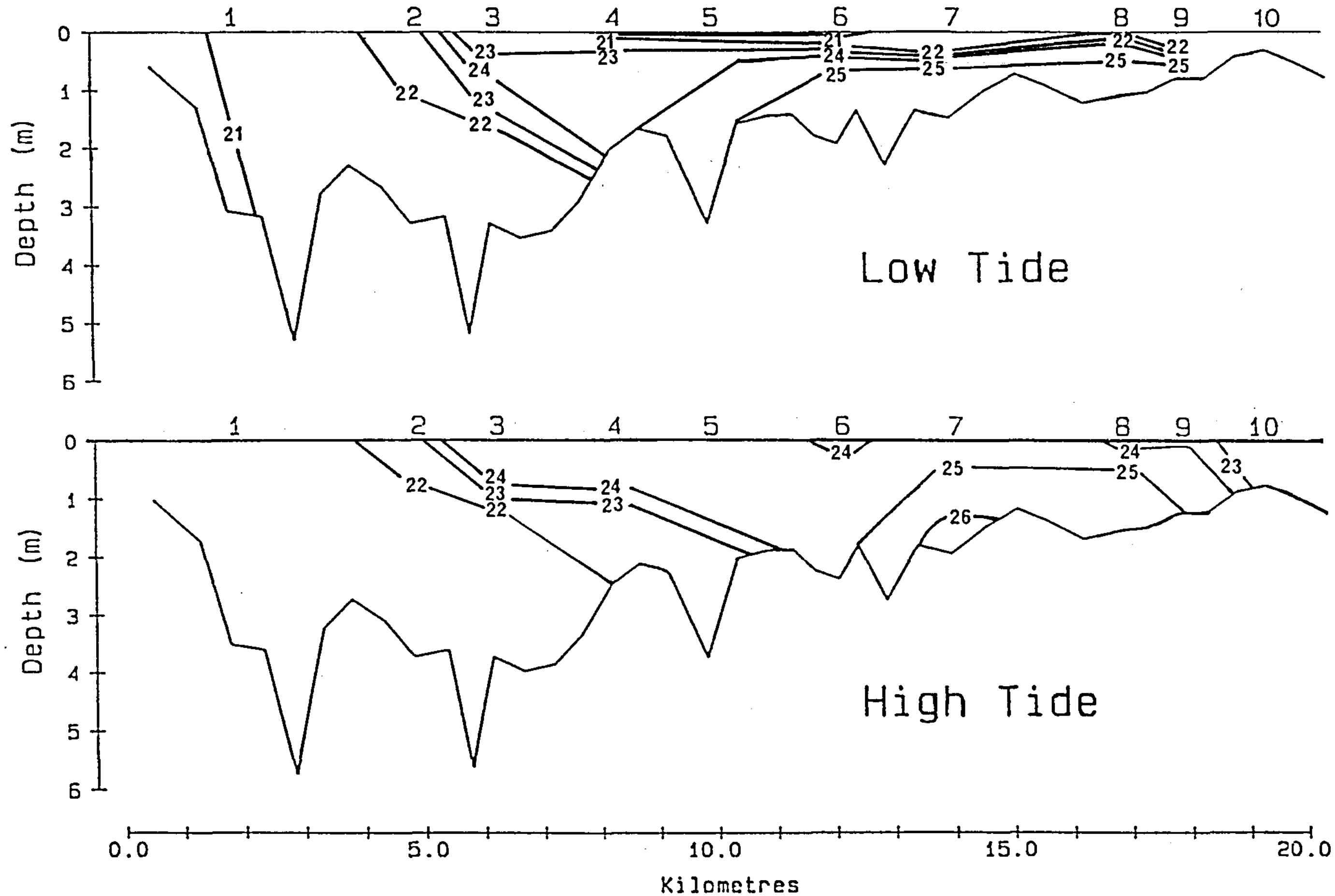

Figure 4.15 Temperature (^{0}C) profiles at low and high tide on 1 April 1993.

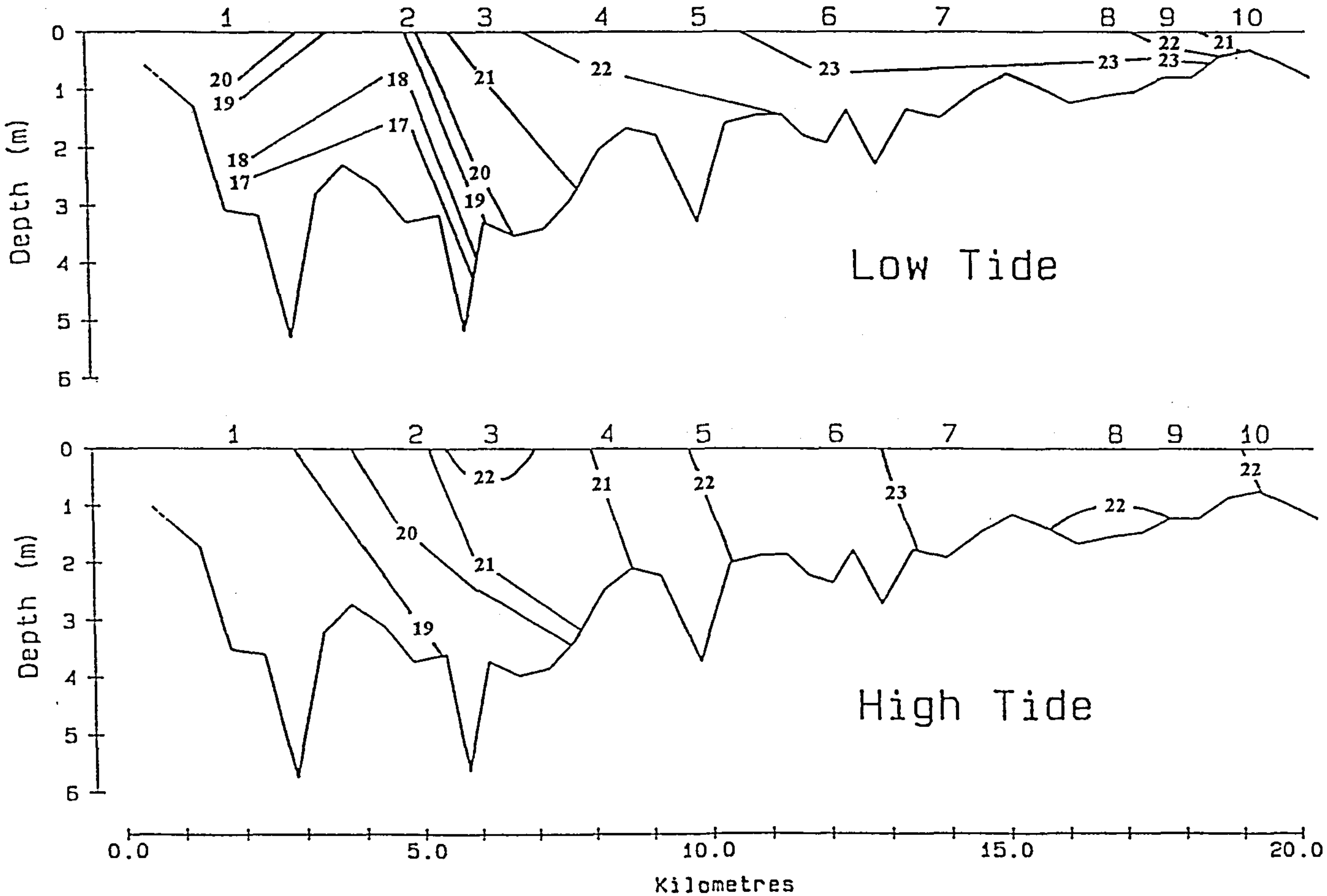

Figure 4.16 Temperature (°C) profiles at low and high tide on 6 April 1993.

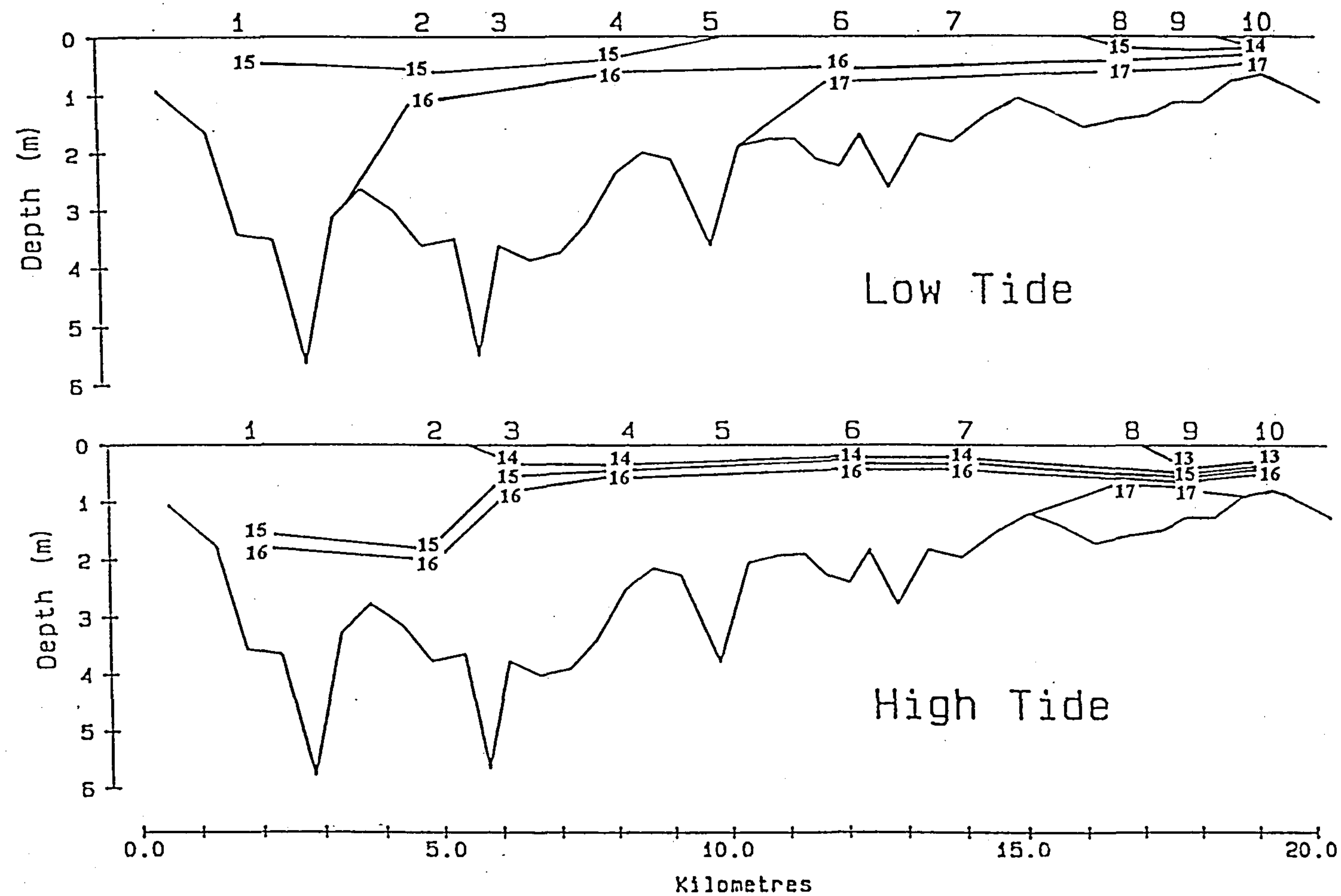

Figure 4.17 Temperature (^{0}C) profiles at low and high tide on 7 June 1993.

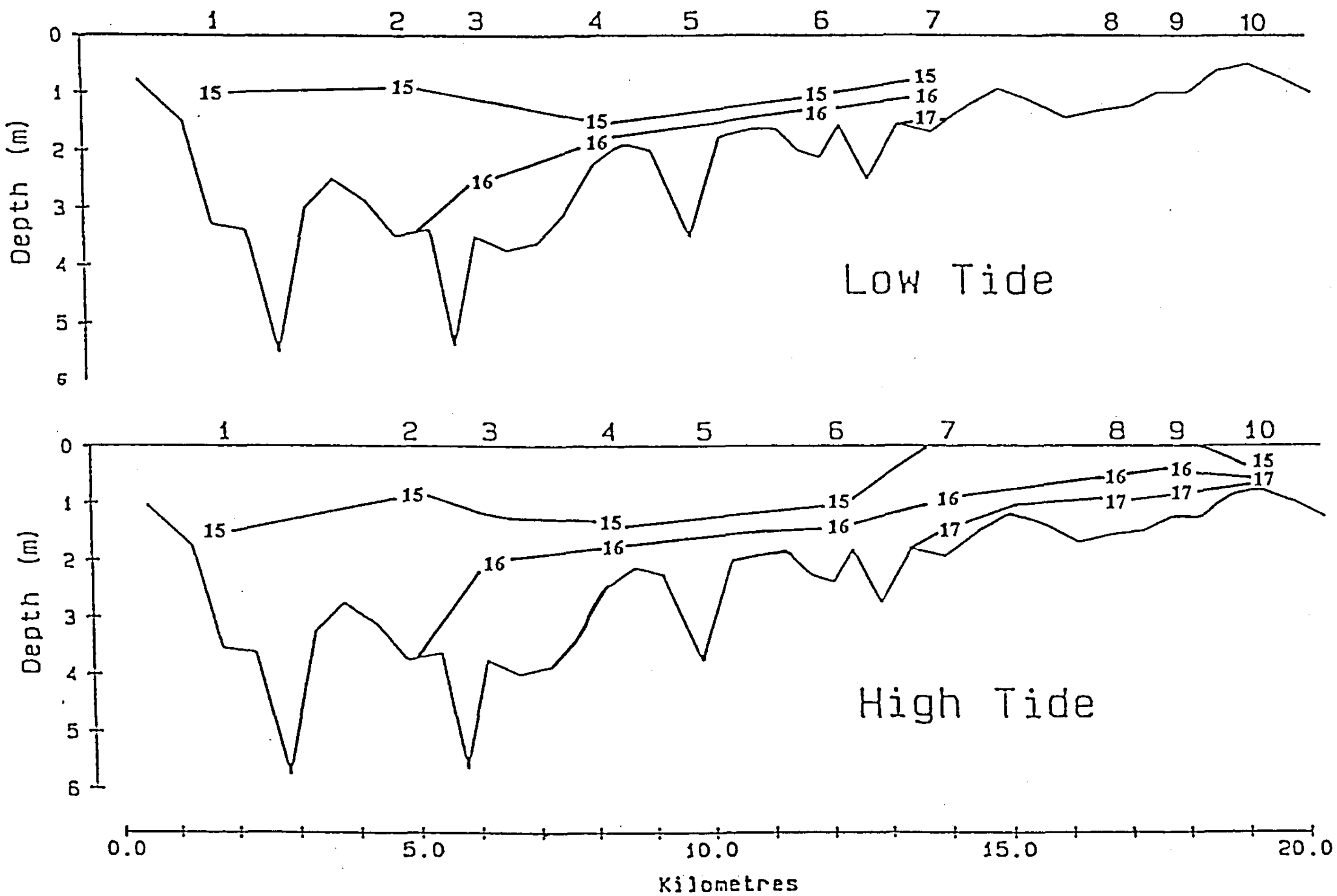

Figure 4.18 Temperature (^{0}C) profiles at low and high tide on 14 June 1993.

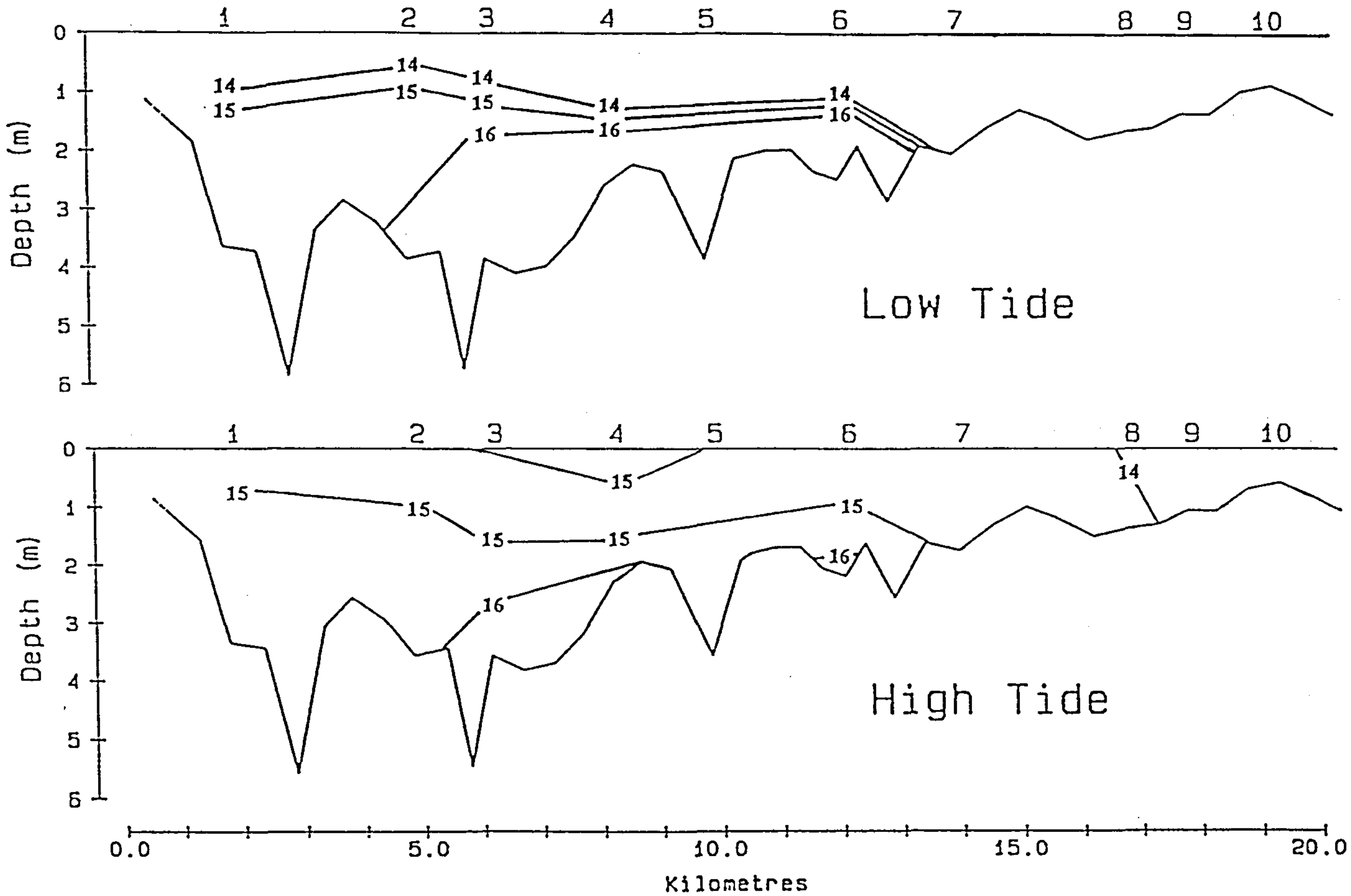

Figure 4.19 Temperature (°C) profiles at low and high tide on 16 June 1993.

4.6.3 Turbidity

Turbidity refers to the lack of water clarity and provides an indication of the suspended organic and inorganic matter in the water (Kirk and Akhurst, 1984). A basic means of measuring turbidity or the approximate depth (in metres) to which 5% of sunlight will penetrate, is using a Secchi disc.

According to Bruton (1985) the occurrence of large quantities of suspended matter and the related reduction in light penetration can have the following negative effects on fish:

- reduced egg and larval survival, altered breeding patterns and reduced growth rates
- reduced feeding efficiency
- reduced population size
- impaired gill function.

Negative effects on other aquatic organisms (Dallas and Day, 1993) include:

- a reduction in photosynthesis by aquatic plants which in turn are a source of food to other organisms within the food chain
- smothering of benthic organisms.

Dallas and Day (1993) draw attention to a study by Ractliffe (1991) who investigated the adverse effect of increased suspended solids in the Lourens River (south-western Cape) which was related to agricultural practices. However, in South Africa there is as yet no recommended turbidity limit for the protection of aquatic life.

The range and mean depths to which 5% light penetration occurred in the Gamtoos estuary during November, December 1992 and June 1993 are given in Table 4.5. Table 4.5 generally shows a trend of increasing clarity upstream. Differences between spring and neap tides were not apparent as water clarity was largely determined by surface runoff and disturbance of river bed sediments during rainfall, regardless of tide. Differences between the lower and upper estuary however can be attributed to the stronger, tide-induced mixing currents causing a greater suspension of sediment in the lower reaches of the estuary. Table 4.5 shows that the mean light penetration during the measurement period of November and December 1992 was deeper than 0.79 m at all stations with the exception of station 5, where readings were between 0.2 m and 0.65 m. Station 5 is situated at the point of discharge of the Loeriespruit tributary into the Gamtoos estuary. Cows often waded into the narrow tributary at the place where readings were obtained, disturbing sediments and causing an increase in turbidity. Generally during November and December 1992 water clarity was greater in the lower reaches during high tide and better in the middle reaches at low tide. Differences in water clarity from high to low tide cannot be determined for the upper estuary as the entire water column was often visible during low tide (except following rainfall) but not always at high tide, thus the difference may be a function of water depth rather than any differences in turbidity caused by tidal processes.

Station	Nov/Dec 1992		4-10 June 1993	
	Range	Mean	Range	Mean
1	0.6-1.90	1.07	0.5-1.25	0.96
2	0.5-1.85	0.96	0.9-1.25	1.09
3	0.6-1.15	0.79	0.75-1.25	1.01
4	0.6-1.15	0.79	1.1-1.5	1.29
5	0.2-0.65	0.44	1.0-1.5	1.16
6	0.75-2.0	1.10	1.0-2.0	1.33
7	0.75-2.25	1.35	1.0-2.4	1.48
8	1.0-1.8	1.38	1.0-1.5	1.28
9	1.0-2.15	1.45	1.2-1.5	1.31
10	1.1-2.3	1.83	1.25-1.8	1.36

Table 4.5 The range and mean values of the depth (in metres) to which 5% of light will penetrate, according to Secchi disc readings taken at the ten estuarine measurement stations. Results are given for November and December 1992 and June 1993.

No light penetration data are available for April 1993. Table 4.5 shows the range and mean Secchi disc depths at the 10 stations from 4 to 10 June (prior to rainfall). On 7 June 1993, prior to the heavy rain, there was a high degree of clarity throughout the estuary at both high and low tide when stations 1 to 10 had a 5% light penetration greater than 1 m. The 87.2 mm of rain that was recorded between 10 and 13 June, resulted in surface runoff with a large sediment load entering the estuary as well as disturbance of estuarine sediments caused by turbulent flow. Because of the murky nature of the water no Secchi disc depths were obtained during this time. By 14 June all stations had a 5% light penetration of less than 0.3 m, on this date the Secchi disc reading at station 5 was < 0.1 m. By 16 June the sediment carried into the estuary by the rain had settled slightly allowing 5% light penetration to a depth of approximately 0.5 m throughout the estuary. Analysis of data does not reveal any trends attributable to differences in tide, apart from those caused by changes in water depth.

In November 1993, a limited number of readings were taken at stations 4 to 10. Despite sampling having been preceded by eighteen rain-free days, water clarity was low, during high tide on 7 November, 5% light penetration occurred to depths ranging from 0.4 m (station 9) to 0.5 m (stations 6, 7 and 10); during low tide depths ranged from 0.2 m (stations 6, 7 and 8) to 0.4 m (station 4). On 13 November, at low tide Secchi disc depths varied from 0.3 m at station 5 to 0.6 m at station 10. During high tide depths varied from 0.2 m (stations 5 and 9) to 0.5 m (station 4). Figure 4.11 shows an inflow of relatively freshwater to the estuary during the November 1993 measurements, thus indicating that dam releases and rainfall higher in the catchment area may have been responsible for the low water clarity experienced at this time.

4.6.4 Dissolved Oxygen

The solubility of oxygen in water is salinity and temperature dependent. The equipment used to measure DO in the Gamtoos estuary gave readings which corrected for the effect of salinity on DO; the influence of temperature on DO is discussed in section 4.9. Factors which tend to increase the concentration of DO in water include reaeration of surface waters as a result of turbulent flow, an increase in atmospheric pressure and a drop in temperature. Photosynthesis of aquatic plants increases aquatic DO during the day, although this is dependent on the degree of light penetration, followed by night-time respiration (decrease in DO) results in a natural diurnal fluctuation of DO. Factors which tend to reduce the DO concentration in water include an increase in salinity and temperature, consumption by benthic organisms, decomposition of organic matter and the breakdown of chemical pollutants (Zagorc-Koncan *et al.*, 1991; Dallas and Day, 1993).

In South Africa the recommended DO concentration, for the protection of aquatic life should not fall below 4.0 $mg.l^{-1}$ (Dallas and Day, 1993). The results to follow are presented with little comment, whilst a more detailed discussion pertaining to dissolved oxygen is given in section 4.9 and centres around the 4.0 $mg.l^{-1}$ concentration level.

DO on 26 November 1992 at both high and low tide was greater than 7.0 $mg.l^{-1}$ at all stations. The following day the lower 0.5 m of the water column at station 6 was below 4.0 $mg.l^{-1}$. Three days later (1 December 1992) the bottom 0.5 m at stations 5 and 6 was less than 4.0 $mg.l^{-1}$ at low and high tide. DO values of less than 4 $mg.l^{-1}$ occurred in the bottom water of stations 5, 6 and 7 at both high and low tide on 4 December 1992. DO values dropped below the recommended limit at station 7 on 5 and 6 December, thereafter DO rose to above 4.0 $mg.l^{-1}$ at all measurement points within the estuary.

The measurements on 1 April 1993, were preceded by fourteen rain-free days, a DO of 1.6 $mg.l^{-1}$ DO was recorded in the bottom water at station 8 (low tide - Figure 4.20) at 8:30 am. Station 9 could not be reached because of the presence of sand banks. By 12:40, during the high tide measurements, the DO content at station 8 had increased to 2.9 $mg.l^{-1}$ and was greater than this value at the other stations. The changes in DO between 8:30 and 12:40 could be partly attributed to natural diurnal fluctuations in DO as thick aquatic plant growth occurs in the upper Gamtoos estuary. Apart from the readings on 1 April 1993, the DO changes recorded during other measurement periods do not correspond to natural diurnal fluctuations and thus factors other than diurnal fluctuations alone are responsible for the changes in DO content during consecutive measurements. During consecutive daily measurements up to 3 April, at no time and at no station did DO fall below the recommended limit. After 3 April readings ceased for equipment repairs until 6 April. During spring low (6 April - Figure 4.21), the bottom waters of stations 7 to 9 exhibited hypoxic conditions, the lowest DO value being recorded at station 9 (2.88 $mg.l^{-1}$) whilst surface waters ranged from 4.80 $mg.l^{-1}$ at station 7 to 5.51 $mg.l^{-1}$ at station 9. During high tide corresponding with the influx of marine water a higher DO was manifested in the lower estuary. This influx moved upstream replenishing the hypoxic sinks. Thus during spring high the DO at all stations downstream of station 8 exceeded 4 $mg.l^{-1}$, with the bottom waters of stations 8 and 9 less than 4 $mg.l^{-1}$.

Figure 4.22 shows the DO status of the estuary (on 7 June) prior to rainfall. DO levels during low tide on 7 June ranged from supersaturated conditions in the surface water of stations 1, 2, 3 and 4 to minimums of slightly less than 6.0 $mg.l^{-1}$ in the bottom water of stations 6 and 7. At high tide on the same day only the bottom layer of station 8 had a DO below 4 $mg.l^{-1}$. By 14 June, a drop in DO following the 87.2 mm of rain is visible in Figure 4.23 where a DO of below 1 $mg.l^{-1}$ occurred in the bottom waters of stations 6 and 7. No data are available for stations 8, 9 and 10 at low tide. The influx of marine water during high tide allowed some replenishment of oxygen to the estuary downstream of the aggraded section between stations 7 and 8. The more exposed stations downstream of station 7 show elevated DO levels at the water-atmosphere interface. Hypoxic conditions however occurred in the bottom waters of stations 2, 3 and 6 to 10. At station 9, hypoxic conditions occurred through the entire water column, with the surface water exhibiting a maximum of 1.8 $mg.l^{-1}$, this could be attributed to organic-rich discharges from the agricultural discharge pipe upstream of station 9. As the greater freshwater input continued as shown in the salinity profiles of 16 June, the DO content decreased accordingly. Figure 4.24 (low tide) shows that the bottom waters of stations 2 to 6 had a DO of less than 2 $mg.l^{-1}$ with the bottom waters of the entire estuary falling below the recommended limit. The freshwater outflow dominated during high tide (Figure 4.10) and although the oxygen content increased slightly (Figure 4.24 - high tide), the DO content at most of the estuarine stations remained below the recommended aquatic limit.

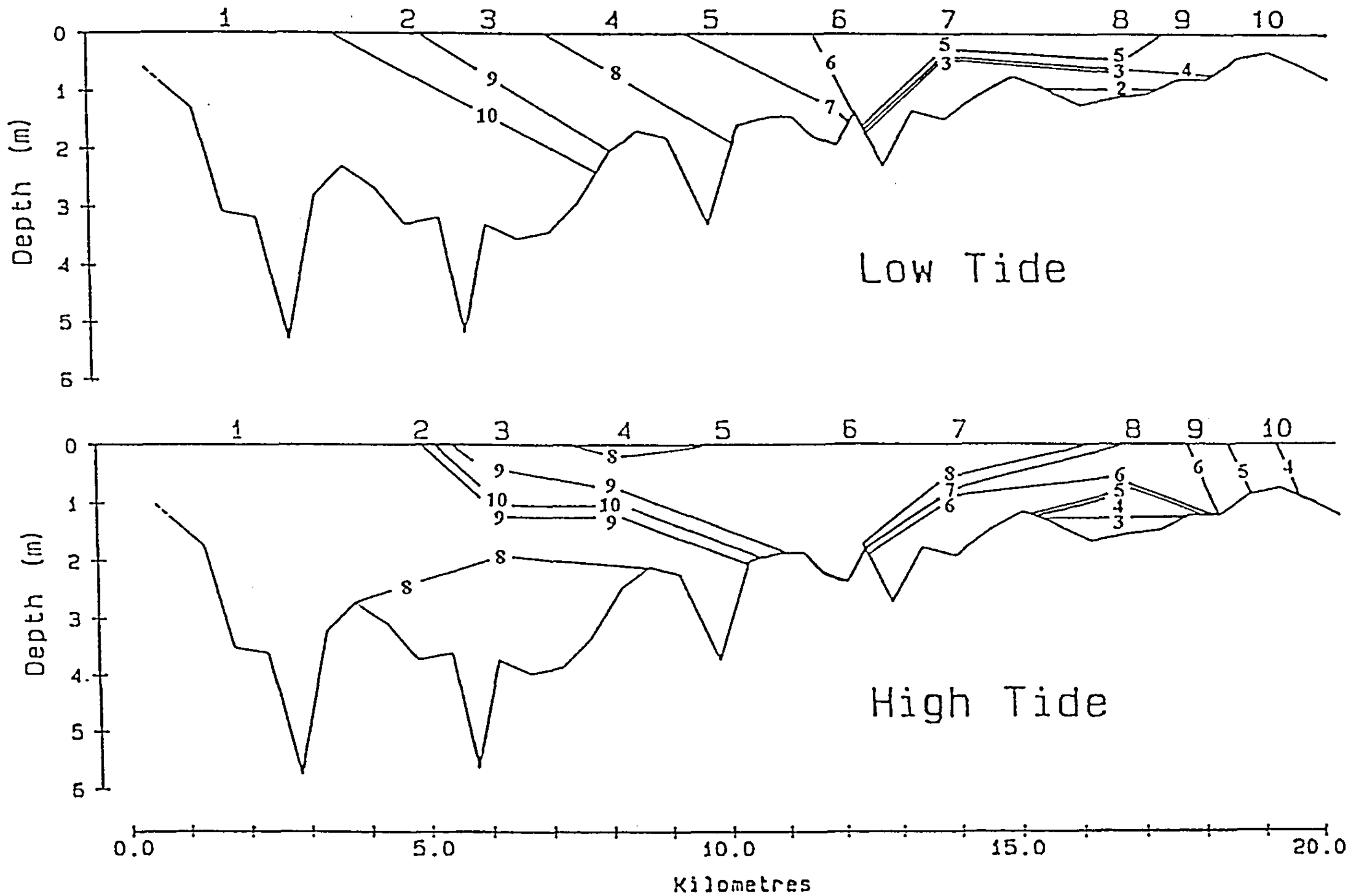

Figure 4.20 Dissolved oxygen ($mg.l^{-1}$) profiles at low and high tide on 1 April 1993.

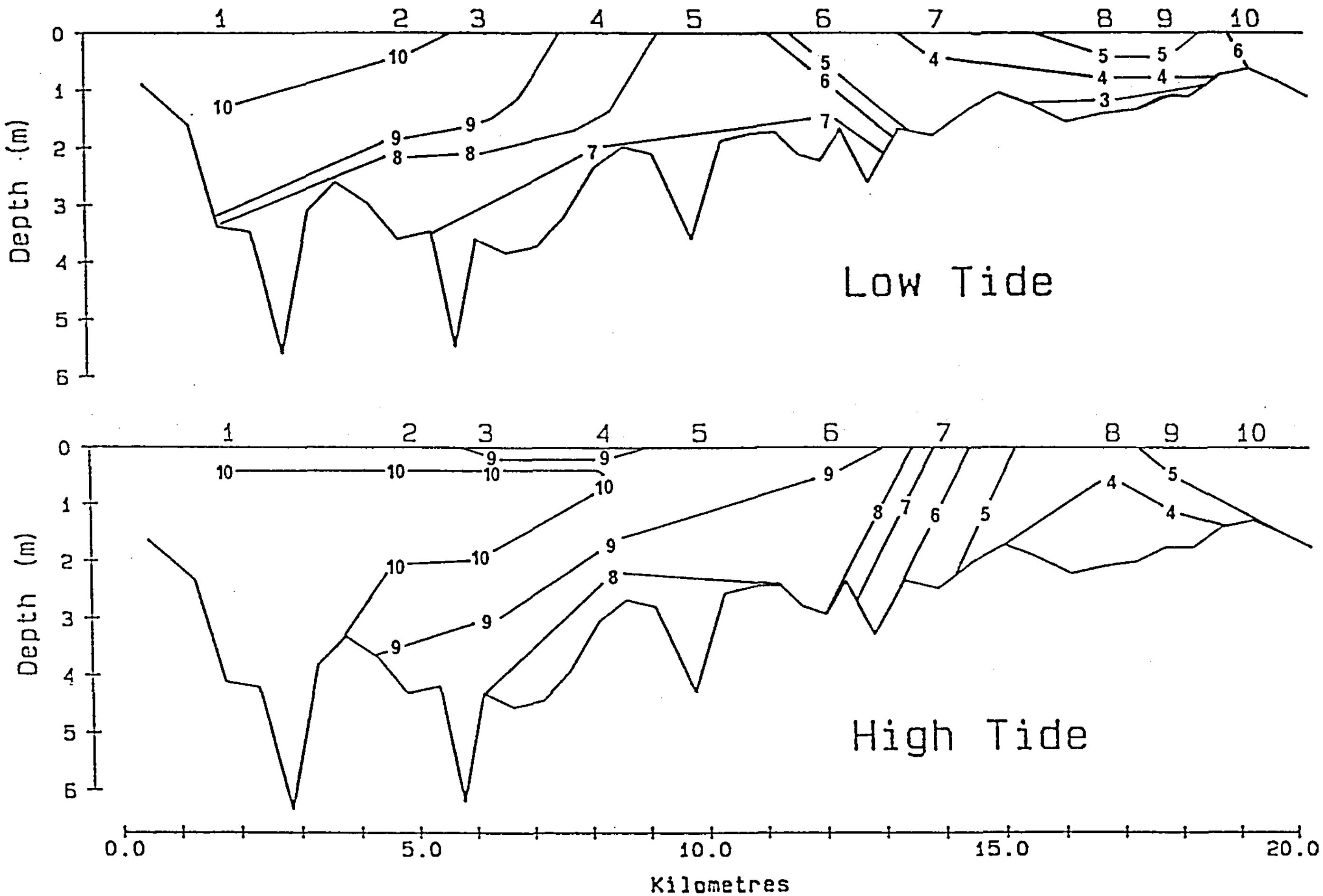

Figure 4.21 Dissolved oxygen ($mg.l^{-1}$) profiles at low and high tides on 6 April 1993.

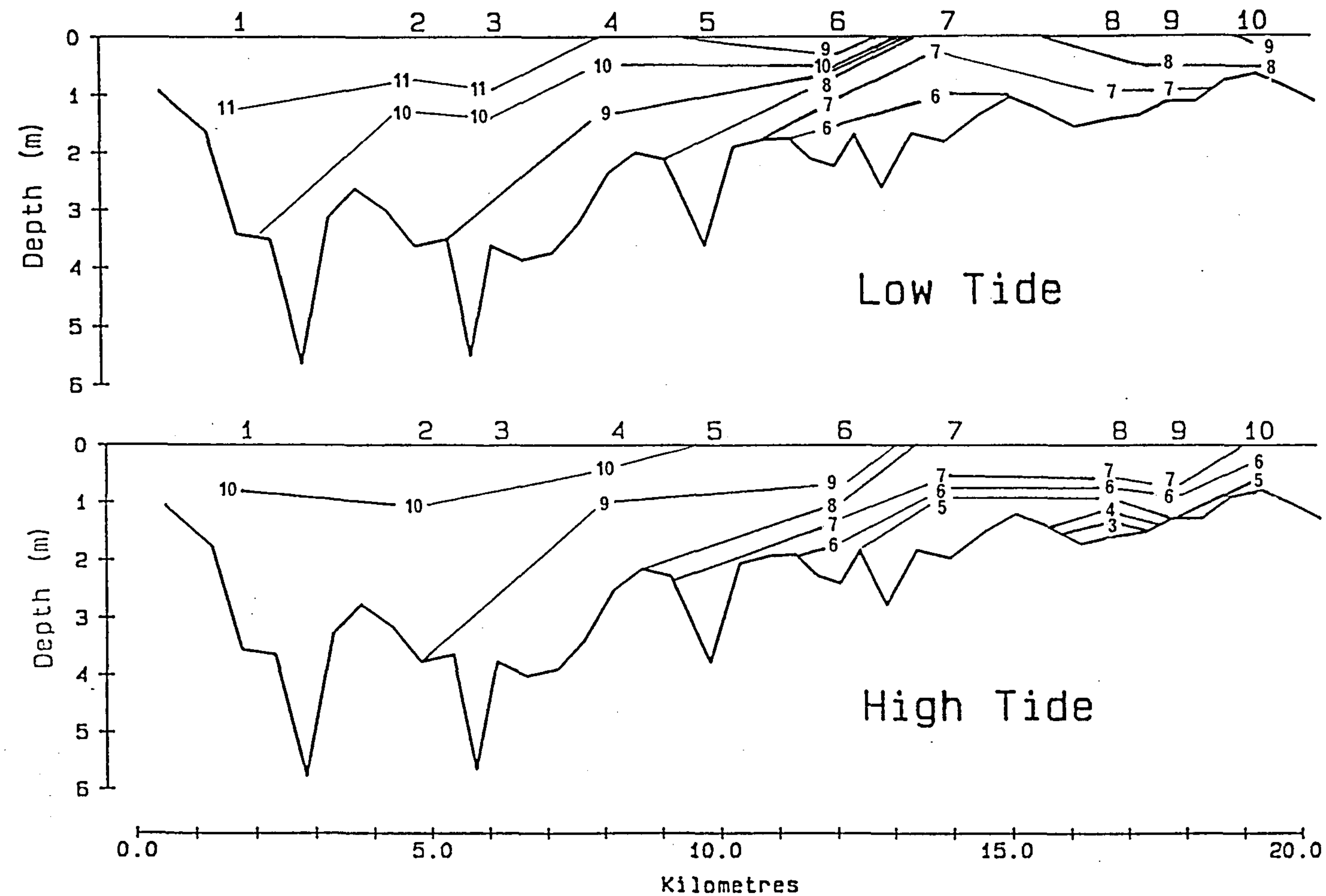

Figure 4.22 Dissolved oxygen (mg.l^{-1}) profiles at low and high tide on 7 June 1993.

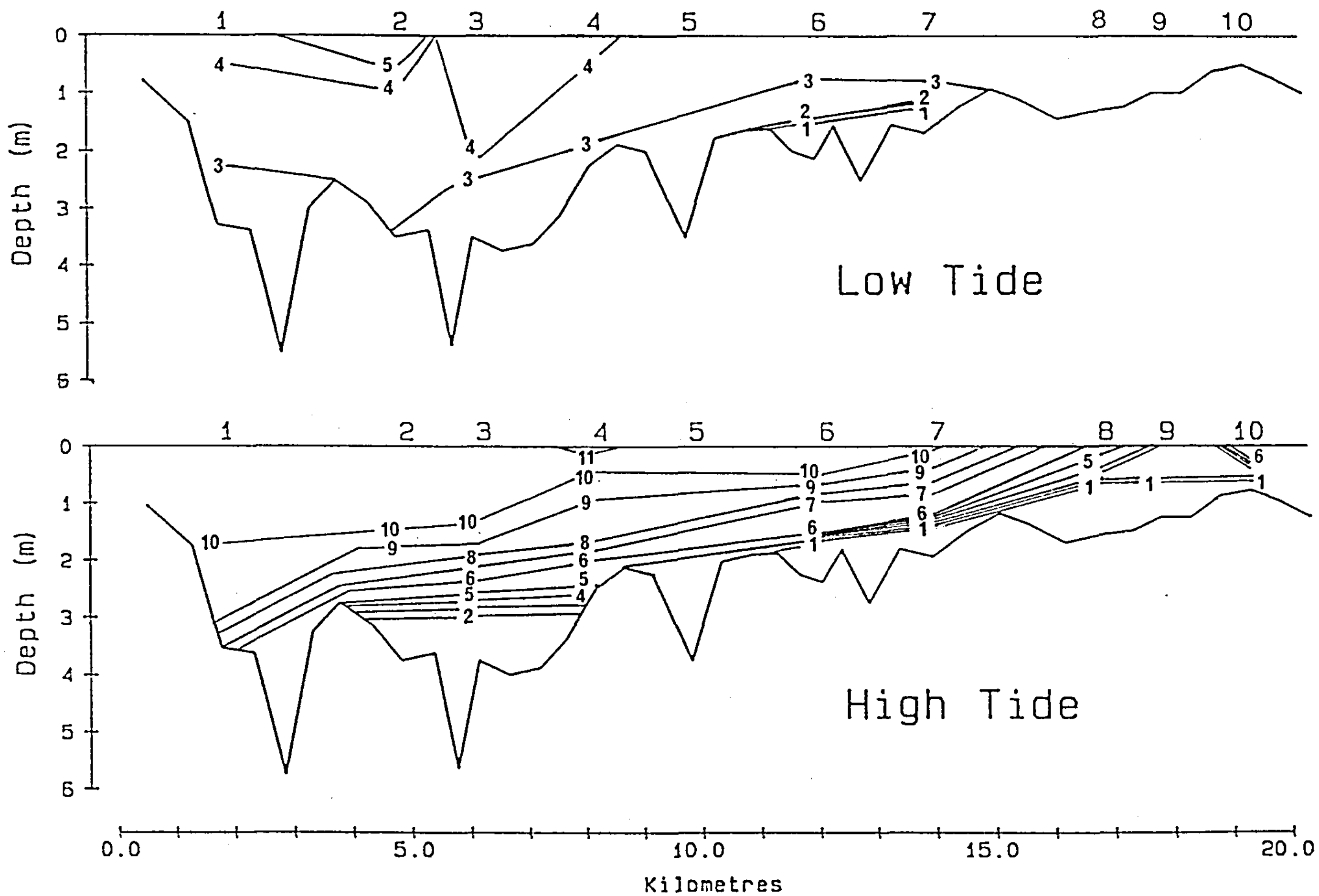

Figure 4.23 Dissolved oxygen (mg.l^{-1}) profiles at low and high tide on 14 June 1993.

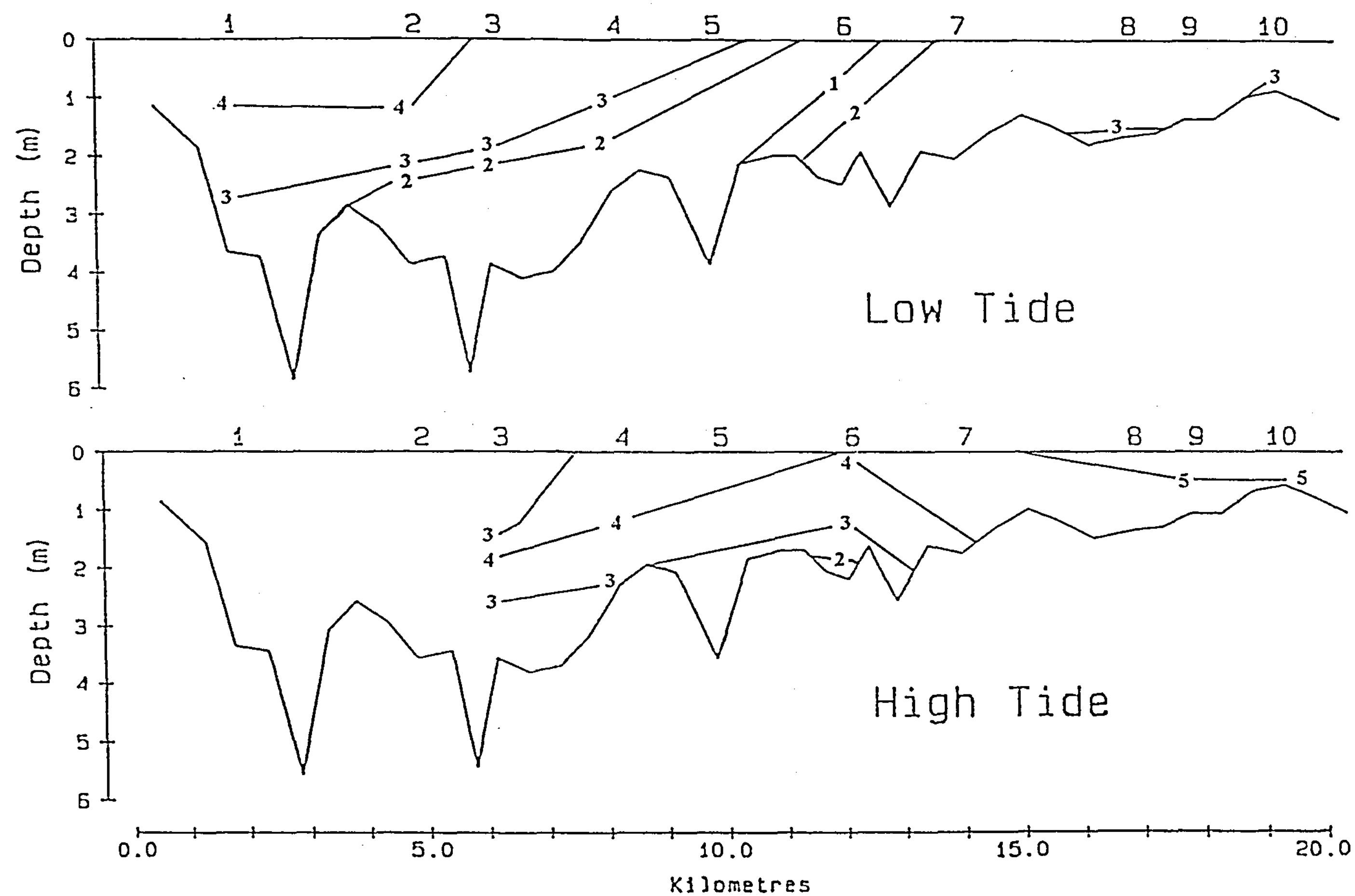

Figure 4.24 Dissolved oxygen ($mg.l^{-1}$) profiles at low and high tide on 16 June 1993.

4.7 Nutrients

When examining nitrate values in the aquatic environment it is important to note whether results are given with nitrate as N (as in this study) or nitrate as NO_3^-. Standards and recommended criteria differ accordingly and misinterpretation of results can easily occur. Fertilizers are one of the main sources of nitrates which are introduced to the aquatic environment via irrigation return flow (which includes surface runoff). Because nitrates are used during photosynthesis they are normally found in a fairly low concentration in surface waters (<0.1 $mg.l^{-1}$) and a recommended limit not exceeding 90 $mg.l^{-1}$ is permissible for the protection of aquatic life. Nitrite and ammonia (both as N) form part of the nitrogen cycle and are harmful to aquatic life if present in concentrations above 60 $ug.l^{-1}$ and 16 $ug.l^{-1}$ respectively. The toxicity of nitrite is inversely related to the concentration of chloride in the water body (Dallas and Day, 1993).

Nutrient levels throughout the estuary were generally low. During routine monthly sampling the nitrate-N content throughout the estuary was equal to or below that of the detection sensitivity for the method of determination i.e. 0.10 $mg.l^{-1}$. On three occasions the nitrate-N level exceeded this routine low value, these being on 15 June 1993 at station 5 where the nitrate-N content was 0.26 $mg.l^{-1}$; on 26 August 1993 values of 0.26 $mg.l^{-1}$ (station 5), 0.84 $mg.l^{-1}$ at station 6 and 0.22 $mg.l^{-1}$ at station 9 were recorded. On 12 October 1993 a nitrate-N value of 0.30 $mg.l^{-1}$ was recorded at station 9, this could have been caused by the high nitrate input of 12.14 $mg.l^{-1}$ at the agricultural drainage pipe upstream of station 9. Although no supporting data are available, it could be speculated that the subsurface agricultural drain discharging into the estuary just upstream of station 6, could have accounted for the high nitrate values at station 6. The elevated nitrate levels at station 5 can be attributed to the excrement from cows which were frequently seen drinking at the site. The elevated nitrate levels were localised in nature and it appears as if dilution has occurred downstream.

Kunishi (1988) examined the fate of nitrogen in the Wye River estuary, Chesapeake Bay and concluded that the low nitrate levels in the estuary (despite an elevated nitrate input) were mostly the result of dilution. Furthermore Kunishi states that denitrification of nitrate and transformation to organic nitrogenous forms could also account for the decrease in estuarine concentrations.

During the time series measurements at station 4 (described in section 3.6.1) nitrate-N values at no time exceeded 0.10 $mg.l^{-1}$. This value is low compared to the mean of 0.52 $mg.l^{-1}$ found by

Emmerson (1985) in the Swartkops estuary, as well as that of the Sundays estuary (0.63 $mg.l^{-1}$) as determined by Emmerson (1989). Emmerson and Erasmus (1987) recorded a mean nitrate-N value of 0.06 $mg.l^{-1}$ in the Kromme estuary.

Table 4.6 gives the mean nitrite-N concentration for the estuarine stations, the values and dates on which the recommended aquatic limit of 60 $ug.l^{-1}$ was exceeded are also given. The nitrite-N concentrations measured during the time series at station 4 on 5 and 13 June 1993 exhibited a small range of values, as shown in Figure 4.25b compared to those of 8 and 14 November 1993 (Figure 4.25a). On 5 June the nitrite-N values ranged from 10 to 41 $ug.l^{-1}$ and those on 13 June from 18

Site	Mean Nitrite-N $ug.l^{-1}$	Date on which the limit was exceeded	Elevated Nitrite-N $ug.l^{-1}$
Station 1	10	-	-
Station 2	9	-	-
Station 3	9	-	-
Station 4	15	9/11/93	102
Station 4	-	14/11/93	274
Station 4	-	2/02/94	68
Station 5	9	-	-
Station 6	7	-	-
Station 7	36	4/06/93	162
Station 8	35	4/06/93	180
Station 9	17	6/11/93	151
Boschhoek	13	2/02/93	67

Table 4.6 Mean nitrite-N concentration of estuarine water and dates on which the recommended aquatic limit of 60 $ug.l^{-1}$ was exceeded.

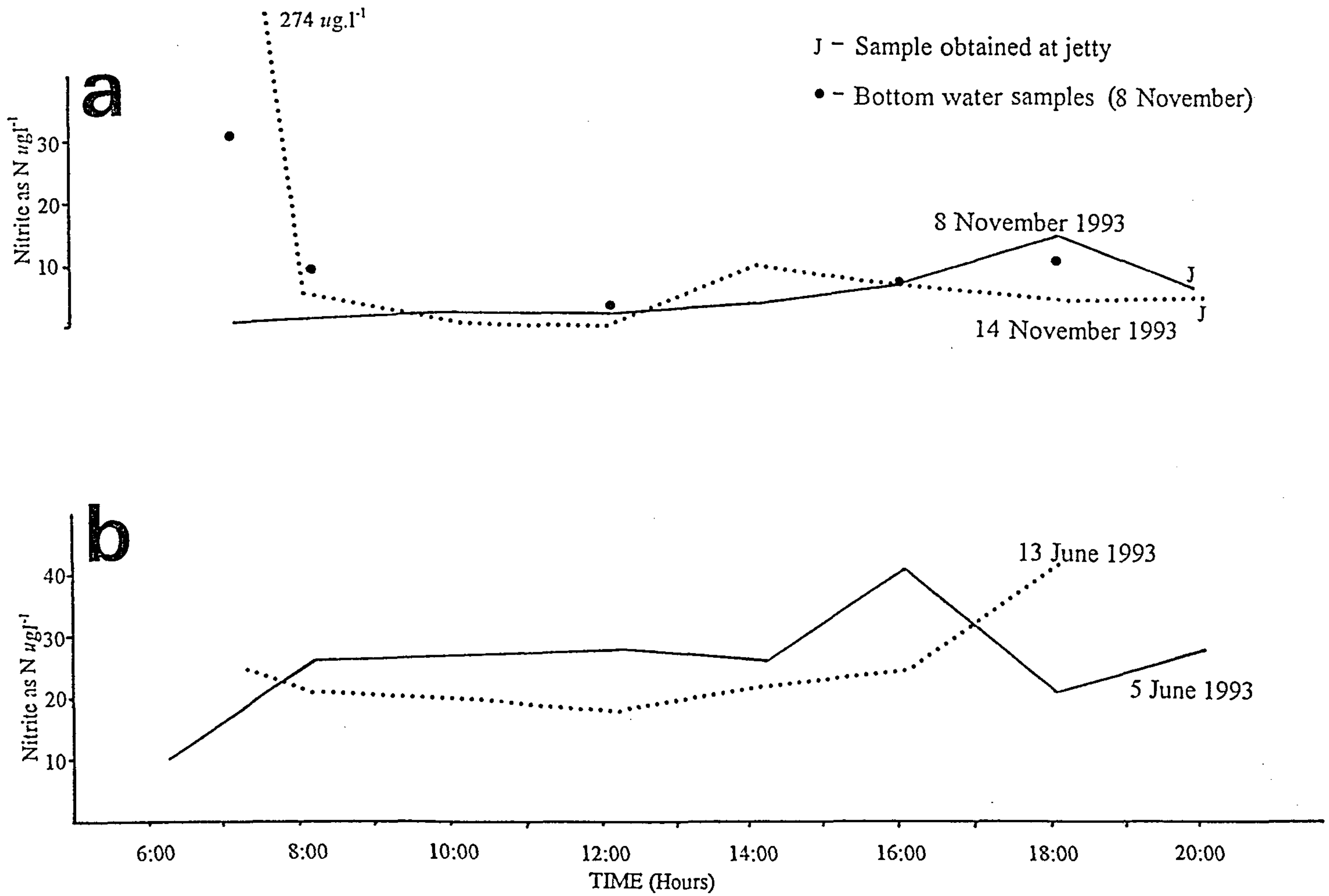

Figure 4.25 Nitrite-N values obtained during the time series measurements at station 4 on **(a)** 8 and 14 November 1993 and **(b)** 5 and 13 June 1993.

to 42 $ug.l^{-1}$.

Whereas on both 8 and 14 November minimum nitrite-N values were less than 0.5 $ug.l^{-1}$ with a maximum recorded value of 31 and 274 $ug.l^{-1}$ respectively. The values given in Table 4.6 can be compared to those of the Swartkops, Kromme and Sundays estuaries where mean values were 13 $ug.l^{-1}$, 5 and 18 $ug.l^{-1}$ according to Emmerson (1985), Emmerson and Erasmus (1987) and Emmerson (1989) respectively.

In South Africa a recommended level of total phosphorus (as P) for the protection of aquatic life is not specified, while the Australian limit is 0.04 to 0.06 $mg.l^{-1}$ (Dallas and Day, 1993). The mean total phosphorus concentration for each station as measured during the study period is given in Table 4.7. On occasions the phosphate concentration was high; these high values are also given in

Site	Mean PO_4-P $mg.l^{-1}$	Unusual values*
Station 1	0.36	-
Station 2	0.15	-
Station 3	0.10	-
Station 4	0.15	-
Station 5	0.28	-
Station 6	0.16	3.81 15/05/95
Station 7	0.10	-
Station 8	0.11	-
Station 9	0.14	-
Boschhoek	0.09	1.02 15/05/94

Table 4.7 Mean total phosphorus concentration and unusual values* (and the date on which they occurred) as measured at estuarine stations 1 to 9 and Boschhoek.

the table, but were excluded from the calculation of the mean at each station.

Phosphorus is readily adsorbed onto soil particles. Thus large quantities of phosphorus may enter an estuary with the sediment load of surface runoff during rainfall (Staver *et al.*, 1996). In the Gamtoos study however, there was no apparent correlation between salinity and the phosphorus content of estuarine water.

Surface water samples obtained during the time series measurements at station 4 ranged from 0.09 to 0.15 $mg.l^{-1}$ on 5 June 1993 (two days after 28.4 mm of rain) and from 0.09 to 0.38 $mg.l^{-1}$ on 13 June 1993 (following 87.2 mm of rain), as shown in Figure 4.26b. Figure 4.26a shows that on 8 November 1993 values ranged from 0.06 to 0.19 $mg.l^{-1}$ in the surface water at station 4 and from 0.05 to 0.14 $mg.l^{-1}$ in the bottom water, although 0.23 $mg.l^{-1}$ was measured later at the jetty (the other measurements were made in the estimated centre of flow). During the November 1993 time series measurements, where a notable difference in salinity occurred between the surface and bottom waters at the estimated centre of flow, samples were obtained. Despite the difference in salinity, analysis showed little or no difference in the total phosphorus content of surface and bottom waters with two exceptions, that of 8 November, three hours after low tide when a phosphorus value of 0.06 $mg.l^{-1}$ occurred in the surface waters, whilst that of the bottom water was 0.14 $mg.l^{-1}$. Similarly on 14 November, two-and-a-half hours after low tide the surface water contained 0.06 $mg.l^{-1}$ phosphorus whilst that of the bottom water was 0.14 $mg.l^{-1}$. The low of 0.06 $mg.l^{-1}$ obtained on 8 November coincided with a layer of markedly lower salinity water (4.36) compared to that measured in all subsequent samples over the thirteen hour monitoring period when total phosphorus values exceeded 0.10 $mg.l^{-1}$, during this time salinity ranged from 14.02 to 31.91. However, on 14 November total phosphorus values were generally lower (mean 0.07 $mg.l^{-1}$) than that of 8 November (mean 0.13 $mg.l^{-1}$ - excluding the jetty measurement on each date) throughout the fourteen hour monitoring period and showed no correlation to salinity which ranged from 6.67 to 28.86.

Phosphorus (phosphates) and nitrogen are the main indicators of the nutrient status of a water body, particularly phosphorus, the results in this study show a higher than recommended level of phosphorus throughout the estuary. Low nutrient levels in an estuary, despite elevated inputs may be partly attributed to sedimentation of phosphorus and nitrogen and rapid denitrification and volitization of nitrogen (Stanley, 1993; Chambers *et al.*, 1995). Utilisation of nutrients within the

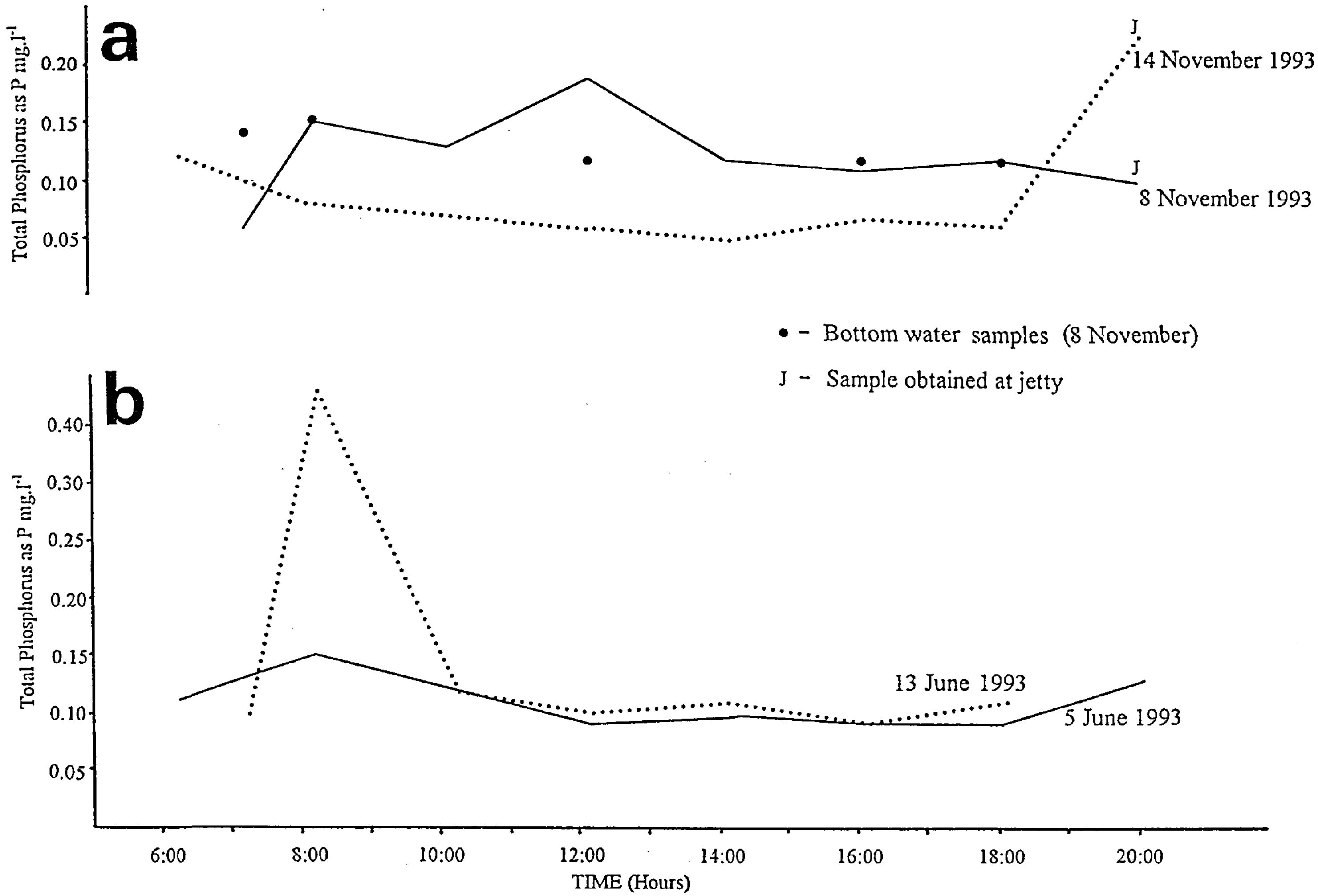

Figure 4.26 Total phosphorus values obtained during the time series measurements at station 4 on **(a)** 8 and 14 November 1993 and **(b)** 5 and 13 June 1993.

water body during primary production may also result in low nutrient levels within the estuary. In retrospect, chlorophyll analyses should have been performed simultaneously with the nutrient analyses in that this would have enabled a comparison of the nutrient levels with primary production in the Gamtoos estuary. There are however, a number of problems associated with the collection and analysis of chlorophyll samples (Tett *et al.*, 1977; Du Preez and Bate, 1991). A large disparity between vertical and horizontal values often necessitates the collection of a large number of samples. Furthermore, variations in algal groups, as well as localised variations in light and nutrient conditions within the water body requires the application of differing analytical techniques to produce reliable results (Ashton *et al.*, 1984).

4.8 Presence of Pesticides

The policy that restrictions on pesticides are more rigorously enforced in Europe, has resulted in the development of a screening test for the detection of pesticides in food and water. Advantages of the cholinesterase screening test are that it is simple, inexpensive, rapid, has a high sensitivity and in certain cases unpurified or minimally purified extracts can be used for analyses (Kumaran and Morita, 1995; Medyantseva *et al.*, 1995). The test detects the presence of organophosphate and carbamate pesticides. Use of organochloride pesticides is generally being replaced with organophosphates, which despite possessing an acute high toxicity, have a low persistence (Hendjii *et al.*, 1993; Skladal, 1992; Dzydevich *et al.*, 1994). Gilliom *et al.*, (1985) define this persistence in soil as ranging from one to twelve weeks after application, furthermore organophosphates are highly soluble in water.

Results of the cholinesterase bio-assay used in the Gamtoos study are given in Table 4.8 as the total cholinesterase inhibition, that is inhibition from pesticides as well as other inhibitive components present in the water sample. Based on this bio-assay it is difficult to extrapolate results directly to the quantity of organophosphate and carbamate pesticides present. Normally one would then further analyse the positive tests for specific pesticides thereby avoiding unnecessary expense on negative samples. Analysis of specific pesticides was beyond the budget of this project.

A similar cholinesterase bio-assay was formulated by Van der Walt (1996) to screen samples to avoid costly analysis of negative samples. Van der Walt's study found that approximately 30% of the cholinesterase inhibition was caused by the presence of pesticides. Furthermore, in certain samples exhibiting cholinesterase inhibition, no pesticides were present or the detection level of the chromatograph technique was below that of the cholinesterase test. The cholinesterase test is a 'safe' test in that it is very sensitive and does not yield any false negatives.

Sample obtained from:	**Date**	**POE ($ug.l^{-1}$)**
Surface discharge pipe	16/03/93	0.12
Surface discharge pipe	6/04/93	0.67
Surface discharge pipe	2/02/94	0.47
Borehole D3	2/02/94	ND
Estuary station 9	2/02/94	ND
Subsurface drain	2/02/94	ND

Table 4.8 Cholinesterase inhibition (POE) as measured in surface-, subsurface and estuarine water samples. ND - no inhibition detected.

Many farmers in the Loerie flats generally only apply pesticides (which are expensive) during summer because the cooler winter temperatures reduce pest problems (Du Preez, 1993). For this reason pesticide screening tests were only done on samples taken during the summer months.

Because the agricultural drainage system drains a large area under intensive cropping, tapping both surface runoff and subsurface flow, three of the six samples screened for pesticides were taken at the agricultural drainage pipe. The results of the analyses, given in Table 4.8, show that cholinesterase inhibition occurred in the three samples obtained from the surface agricultural drainage pipe whereas no inhibition was detected in the single sample of groundwater (obtained from borehole D3), estuarine water downstream of the surface drainage discharge pipe (station 9) and the subsurface drainage discharge. Whilst few conclusions can be drawn from such a small sample group, results indicate that the expense of a larger scale investigation into the use of specific pesticides could be justified.

4.9 Discussion

Estuarine temperature stratification as occurred during high tide on 26 November 1992, low tide on 10 December 1992 and at low tide on 1 April 1993 may influence the fate of nutrients entering the estuary. Where thermal stratification exists there is a corresponding difference in water density, with the less dense water lying above the more dense water. Surface water will flow as a coherent layer exhibiting little mixing with underlying waters. Thus nutrients entering the estuary during this time would be held within specific layers with little mixing. However, apart from the previously mentioned dates there was little thermal structure, particularly in the lower estuary, with temperature differences frequently less than 2^0C over the entire water column.

The solubility of oxygen in water decreases with an increase in temperature. A comparison of Figures 4.15 and 4.20; 4.16 and 4.21 at low tide only; 4.17 and 4.22; and of Figures 4.18 and 4.23 showed a general correlation between the warm water and the area having the lowest DO content. For example, on 1 April 1993 the 25^0C isotherm corresponded with the lowest recorded DO of 3 $mg.l^{-1}$. At both high and low tides on 7 June 1993 the area containing the lowest DO levels corresponded with the relatively warm water (17^0C isotherm). Similarly, on the 14 June 1993 the 17^0C isotherm corresponded to the areas of lowest DO content. However, temperatures of 25^0C can hold in excess of 6 $mg.l^{-1}$ at normal atmospheric pressure (APHA, 1989) and thus whilst the depressed DO levels may be partly attributed to the slightly elevated water temperatures, the low DO cannot be attributed to temperature differences alone (other factors are discussed later in this section). On 1 April 1993 the water having a temperature greater than 25^0C had a DO content of 3 $mg.l^{-1}$ during low tide and ranging from 3 to 8 $mg.l^{-1}$ during high tide. On 7 June 1993 estuarine water with temperatures between 16 and 17^0C had DO contents ranging from 5 to 10 $mg.l^{-1}$. Similarly, on the 14 June water of the same temperature range varied in DO content from 1 to 3 $mg.l^{-1}$ during low tide and from 1 to 7 $mg.l^{-1}$ during high tide.

The findings in the Gamtoos estuary are similar to that found by Greb and Graczyk (1993) in a number of warm water and cold water streams in Wisconsin. Greb and Graczyk state that the most common occurrence of low DO occurred when temperatures were above 20^0C. On comparing the mean DO concentration of the warm water and cold water streams however, Greb and Graczyk state that the mean DO concentrations were similar regardless of water temperature. Furthermore, they found that 69% of the DO minimas occurred on storm days, where a storm day is defined as a period of 48 hours from the start of any measurable rainfall.

The rainfall-salinity response in the estuary during the four measurement periods was determined by the differences in freshwater input, with tidal exchanges being the main cause of mixing. Tidal exchanges largely determined the salinity structures in the lower estuary whilst spring high tides were more effective in penetrating the upper estuary.

Although the salinity profiles of the four intensive monitoring periods largely ignore station 5, which was a short distance up the Loeriespruit tributary, the freshwater input from the tributary influenced estuarine structures. Inputs were mostly small and with tidal mixing the extent of the freshwater input was generally not apparent much past stations 4 and 6. The input from the Loeriespruit depends on the natural overtopping of the Loerie dam as well as releases from the dam. Much of the released water is abstracted by farmers bordering the upper reaches of the tributary.

Wind-induced mixing does not appear to be a very important process in the Gamtoos estuary, except in extreme situations. This is because the estuary is fairly narrow, giving the wind little fetch over which to generate appreciable waves, except along the stretch from stations 6 to 7 when the wind is aligned with the channel direction. Such wind alignment accounted for less than 22% of each of the four measurement periods (based on 24 hour wind data). Salinity profiles at station 6 were examined together with wind data. Almost unbroken wind records were available throughout the study period for Patensie (Figure 2.1) and Port Elizabeth airport (obtained from the Weather Bureau, Pretoria). On examining wind data from both places together with the Gamtoos field observations, it appeared that the Port Elizabeth data was the most applicable to the Gamtoos study area; in addition the latter two are coastal regions. In the discussion to follow, wind data as obtained from Port Elizabeth is applied to the study area. The discussion is limited to periods when the wind was aligned with the channel orientation at station 6 (south-westerly and north-easterly).

Although a south-westerly wind prevailed for a 9 hour period preceding the salinity measurements at high tide on 30 November 1992, salinity structures showed no signs of wind-induced mixing. However, by low tide that afternoon some wind-induced mixing appears to have occurred following the 15 hour, near-constant wind direction which preceded the estuarine measurements. During this time the wind velocity ranged from 2.3 m.s^{-1} to 12.6 m.s^{-1}, with a mean of 7 m.s^{-1}. At low tide on 30 November although vertical salinity stratification was still evident at station 6, some degree of mixing had occurred at the surface. The salinity of the surface water was 11 (and bottom water 30), whereas at low tide on the preceding day and the day following these measurements the surface water had salinities of 1 and 3 respectively. On 4 and 5 December 1992 the high tide estuarine measurements were preceded by a station-aligned wind lasting 5.5 hours (mean wind velocity 9.5 m.s^{-1}) and 4.5 hours (mean wind velocity 4.6 m.s^{-1}) respectively. On both these dates, despite the wind, the salinity structures were tide dominated.

The measurements at low tide on 2 April 1993 were preceded by 10 hours of estuarine aligned winds averaging 10.4 m.s^{-1}. The salinity profile at station 6 at this time shows evidence of some surface mixing when compared to the profiles of the preceding and following days. On 2 April the surface water had a salinity of 22 whereas on the 1st it was 14 and on 3 April it was 18. At no other time during the April measurements did such winds persist for more than 4 hours and estuarine structures were tide dominated.

During the June 1993 measurements there was no evidence of wind mixing at station 6. The estuary aligned winds generally did not persist for any length of time, with two exceptions. On 10 June a light (2.6 m.s^{-1}) north-easterly wind swung to a south-westerly direction with a mean velocity of 12.8 m.s^{-1}. Although these winds continued for 11 hours (mainly through the night), a change in wind direction in the early hours of the next morning meant that by the time the daily estuarine readings were taken, salinity structures were tide dominated. Again on 16 June, a south-easterly wind blew for almost 10 hours, however the estuary at this time was dominated by a strong freshwater flow (see Figure 4.10) following heavy rain. Wind generally exerted a minor influence on mixing processes in the Gamtoos estuary, with such effects short-lived, while tidal influences dominated estuarine structures.

Unlike the immediate rainfall-salinity response within the estuary, the DO response in the Gamtoos estuary showed a delayed effect. On 26 November 1992 when a freshwater input was clearly evident in the salinity profile (Figure 4.3) the DO was above 7 mg.l^{-1} throughout the estuary at both high and low tide. But three days later at both high and low tides the bottom waters of stations 5 and 6 showed low DO values. By 4 December, a further three days later, when the freshwater input was decreasing (which was particularly noticeable during high tide) the DO was above the recommended minimum throughout the estuary. On 6 April, three days after rain occurred, low DO values were recorded in the bottom waters of stations 7, 8 and 9. On 7 June, again a few days after rain fell, a low DO was recorded in the bottom water at station 8.

The hypoxic conditions observed from time to time particularly following rainfall, in the Gamtoos estuary can be attributed to a number of factors:

During rainfall organic matter and other oxygen-demanding substances were washed into the estuary; increased throughflow from surrounding fields also occurs. On the land surface rainfall will come into contact with oxygen-demanding substances, thus by the time the surface runoff enters the estuary it would be characterised by a low oxygen content. Although increased turbulence and mixing, associated with increased stream discharge, favour reaeration of surface waters, the increased turbulence also results in the resuspension of sediments. In this way, large quantities of organic matter and other oxygen-demanding matter are released into the aquatic environment. Decomposition of organic matter within the estuary causes an increase in oxygen demand resulting in hypoxic conditions. The deeper scour holes (station 8, see Figure 2.4) upstream of the aggraded section serve as sinks for organic debris. This may account for the delay between rainfall and the drop in DO, as the organic matter undergoes decomposition, resulting in localised hypoxic conditions. Graczyk and Sonzogni (1991), on examining the DO content of four southern Wisconsin streams, found an immediate decline in DO concentration with an increase in stream flow. They stated that the drop in DO was caused by agricultural runoff during summer rainfall.

Hypoxic conditions in the Gamtoos estuary mostly occurred in the bottom waters of the upper estuary. Following the extremely high rainfall of June 1993 however, low DO conditions occurred throughout the estuary and persisted for some time. The reason the hypoxic conditions were more frequently observed in the upper estuary can be attributed to the restricted movement of the tidal waters upstream of the shallow, aggraded section. Apart from spring high tide, mixing and flushing in the upper estuary occurred at a lower rate when compared to the lower estuary. Furthermore, the surface agricultural drainage system which collects runoff and throughflow from the upper 1.2 m of soil from a 50 ha area, discharges into the estuary upstream of station 9. Analysis of water samples taken from the drainage pipe revealed that the discharge contained elevated levels of nutrients, as well as organophosphate and carbamate pesticides. Large quantities of organic foam were also frequently observed at the discharge site.

Pronounced salinity stratification has been found to inhibit the exchange of oxygen between surface and bottom waters and thus results in hypoxic conditions in near bottom waters (Gaston, 1985; Schroeder, 1985; Kuo and Neilson, 1987). In the Gamtoos estuary, low DO exhibited in the near-bottom waters of stations 5 and 6 on 1 December 1992 during both high and low tide and in stations 5 to 7 on 4 December corresponded to saline stratification. Similarly, the low DO content in the bottom waters at stations 5 and 7 during both high and low tide on 5 and 6 December, corresponded to saline stratification. Stations 8 to 10 showed homogeneous freshwater conditions (salinity <0.4) and a high oxygen content throughout. Upstream of station 7 the water had a salinity uniformly below 1 (low tide) and 4 at high tide. Salinity stratification and hypoxic conditions occurred on 6 April during low tide; whereas at high tide when almost vertically homogeneous salinity occurred in the upper estuary, all stations had DO contents above 4 $mg.l^{-1}$. The extremely high rainfall of June 10 to 13 ensured that the upper estuary was filled entirely with freshwater, with saline stratification in the lower estuary only. During this time, despite hypoxic conditions throughout the estuary, this hypoxia was confined to the near bottom waters in the lower estuary (saline stratification) whereas in the upper estuary the entire water column exhibited hypoxic conditions, particularly after 14 June.

Not only does salinity stratification inhibit the exchange of oxygen between surface and bottom waters, but the dense mats of a free-floating macrophyte, *Azolla filiculoides* (see section 4.3) in the upper estuary would further inhibit oxygen penetration to underlying layers. The lack of light penetration would, in addition hinder replenishment of oxygen by photosynthesis of aquatic plants in the near bottom waters of the estuary. The high flow, estimated as exceeding 100 $m^3.s^{-1}$, following the heavy rainfall in June 1993, was however sufficient to break-up and temporarily flush *Azolla* mats out of the estuarine system. *Azolla* reappeared in the estuary after this flushing, but was less extensive.

An increased turbidity caused by runoff during rainfall as well as bank and bed erosion will exacerbate hypoxic conditions within the estuary. As mentioned previously, the drop in light penetration results in a reduction in the rate of photosynthesis of aquatic plants, with a corresponding reduction in oxygen replenishment. Furthermore, any temperature increases associated with the rainfall runoff will adversely affect the DO content of the estuary.

The study conducted in the Gamtoos estuary was a broad-based study with many aspects to it, and as a result no aspect was studied in particular detail. Possible causes of hypoxic conditions in the Gamtoos estuary have been presented and tidal mixing has been cited as the main agent in oxygen replenishment. Kuo and Neilson (1987) examined hypoxic conditions in three Virginia estuaries and provide some detail on factors (other than those already mentioned) influencing oxygen replenishment. The Virginia estuaries were regarded as partially mixed as can the upper part of the Gamtoos estuary. The seasonal temperature range in the Virginia estuaries was from 4^0C in winter to 28^0C in summer and they were deeper than the Gamtoos estuary, being about 5 to 10 m with deeper holes in areas. Periods of hypoxia, particularly pertaining to the deeper holes were examined in their study. Kuo and Neilson provide a conceptual model of the DO budget in a deep hole within a partially mixed estuary, this is given in Figure 4.27. On examining the low DO content of deeper scour holes, Kuo and Neilson state that replenishment of DO occurs partly as a result of advective transport, which in partially mixed estuaries is driven by gravitational circulation. Furthermore, it is a longitudinal salinity gradient between the lower and upper estuary which sets up gravitational circulation.

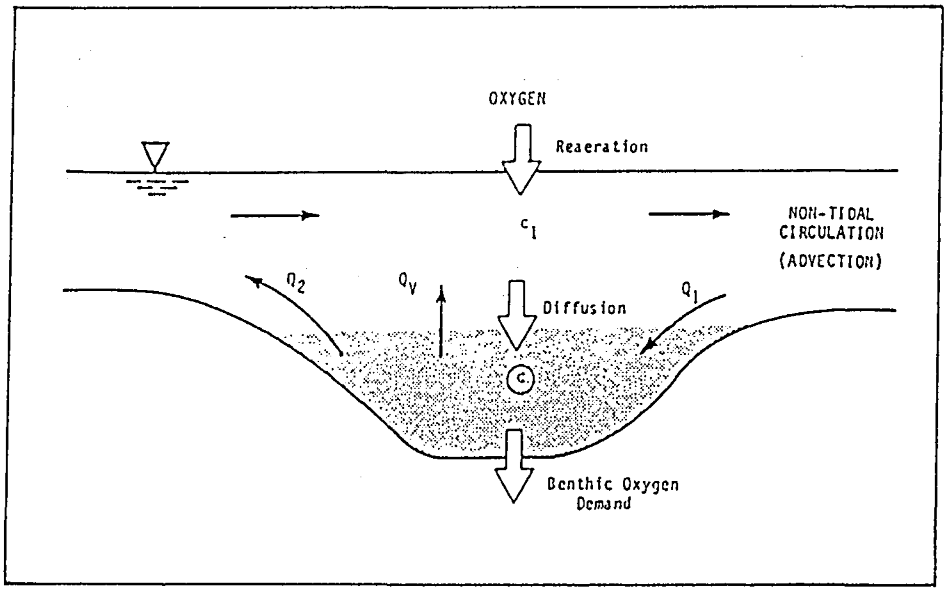

Figure 4.27 A conceptual model of the DO budget in a deep hole in a partially mixed estuary (from Kuo and Neilson, 1987).

4.10 Impact on Biota

An investigation into the response of biota within the Gamtoos estuary falls outside of the scope of this project, however, a brief literature review on the impact of physical changes on biota in aquatic environments is pertinent.

Day (1981; p. 198) defines an estuary as '*a partially-enclosed coastal body of water which is either permanently or periodically open to the sea and within which there is a measurable variation of salinity due to the mixture of sea water with freshwater derived from land drainage*'. Because of the 'measurable variation in salinity', estuarine biota comprises fauna and flora that can tolerate changes in salinity (i.e. euryhaline species). However rapid or extreme changes in conditions within the estuary will impact on the biota therein, particularly sessile species. Unnatural and rapid temperature fluctuations have been known to cause fish mortalities. Temperature changes can impact on various aspects of the life-cycle of fish, such as development of eggs and larvae, physiology, metabolism, behaviour and feeding (Alabaster and Lloyd, 1980; Lankford and Targett, 1994; Secor and Houde, 1995; McKenney and Celestial, 1995).

Changing salinity structures are important for estuarine biota, and Whitfield and Bruton (1989) blame excessive abstraction of freshwater from the Seekoei River catchment for the hypersaline conditions in the estuary which led to massive fish mortalities in March 1989. Furthermore they state that excessive damming or freshwater abstraction can lead to reduced biotic diversity, reduction in cues for fish migrating into the estuary and impeded downstream migration caused by dam walls. Marais (1988) investigated fish abundance in fourteen estuaries along the south and south-east coast of South Africa and found that fish numbers and mass were inversely correlated with salinity. Goncalves *et al.*, (1995) report that extreme salinity's (too high or too low) were a limiting factor in the development of *Rhithropanopeus harrisii* larvae. An investigation by Wooldridge and Bailey (1982) in the Sundays estuary found that an increase in copepod, *Pseudodiaptomus hessei* and *Acartia longipatella,* densities were associated with the freshwater input. Hilmer (1990), also working in the Sundays estuary, showed that chlorophyll-*a* values were elevated in the more highly stratified sections of the estuary (i.e. also dependent on freshwater input). Jerling and Wooldridge (1994) examined the occurrence of various copepods in the Gamtoos estuary and found that the copepod density of *Pseudodiaptomus hessei* and *Acartia longipatella* was negatively correlated with salinity, whereas *Acartia natalensis* showed a positive correlation. The correlation between salinity and copepod abundance however, may be indirectly related to a flush of nutrients associated with the freshwater input to the estuary.

Hellawell (1986) states that organic inputs to an aquatic environment cause a change in the biological community and relative abundance of organisms through hypoxic conditions. Furthermore the addition of nutrients can lead to the proliferation of certain organisms. Reyes and Merino (1991) examined the DO dynamics in a shallow lagoon in Cancun, Mexico. They found that resuspension of organic matter and subsequent decomposition of organic debris resulted in the consumption of DO and that turbidity limited the production of DO. As the lagoon system received a high organic input, yet had minimal exchanges with the open sea and was poorly flushed, the resultant hypoxic conditions and increased primary production resulted in eutrophication problems in the lagoon. Fichez *et al.*, (1992) relate algal blooms in the shallow, Great Ouse estuary, England, to the concentration of chlorophyll-*a* and discuss the related nitrate and phosphate cycling within the estuary.

Dauer *et al.*, (1992) working in the lower Chesapeake Bay found that low DO (<2 mg.l^{-1}) in near-bottom waters resulted in high benthic mortalities and lower species diversity. Gaston (1985) examining water in the Inner Shelf off Cameron, Louisiana states that hypoxic conditions led to a dramatic reduction in the populations of *Phoronis* sp., *Mediomstus californiensis* and *Cirratulus* cf. *filiformis*. Kuo and Neilson (1987) state that when the bottom waters of the Virginia estuaries are hypoxic, oyster larvae become stressed and if they rise to more oxygenated levels are then flushed out of the system. Graczyk and Sonzogni (1991) report that episodic reductions in DO cause stress to fish populations which can result in fish kills. Not only does a low DO stress an aquatic ecosystem but according to Lloyd and Swift (1976), a synergistic effect exists in the presence of certain toxins.

DO is also indirectly influenced by turbidity or depth of light penetration in estuarine water in that a reduction in light penetration causes a reduction in photosynthesis by aquatic plants. Introduction of suspended matter via surface runoff during rainfall events and resuspension of river bed sediments are the main, natural causes of an increase in turbidity and a corresponding drop in the depth of light penetration. Increased turbidity also causes impaired gill function in fish (Bruton, 1985) and smothering of benthic organisms (Dallas and Day, 1993). Whitfield and Paterson (1995) found that a high silt load and low DO were responsible for the mass mortality of fishes following a flood in the

Sundays estuary, Eastern Cape. Hecht and Van der Lingen (1992) however, state that whilst turbidity reduces the feeding efficiency of visual predators, it did not lead to a reduction in the abundance of fish in turbid estuaries. According to Cyrus and Blaber (1992) turbidity and salinity regimes were responsible for determining the seasonal distribution of many species of fish in the Embley estuary in northern Australia. Day (1981) and Adams and Talbot (1992) report that a high silt load has a negative impact on the growth and survival of *Zostera*, a subtropical sea grass. Similarly reduced water clarity was the cause of the disappearance of *Halodule wrightii* and *Thalassia testudinum* from experimental plots (Czerny and Dunton, 1995).

5. GROUNDWATER

This section deals with the results of the groundwater measurements and analyses. As past fluvial influences on the floodplain have led to a variation in soil properties and drainage conditions at each site, results are presented on a site by site basis. Site M is drained by a surface agricultural drainage system, site V is characterised by bar-and-swale topography, and site S was subject to frequent flooding during this investigation. This is followed by a discussion of specific chemical constituents as they occurred in the groundwater throughout the study area.

5.1 Soil Profiles

Figure 5.1 shows the position of the boreholes in the study area. Soil textures, identified according to particle size (as discussed in section 3.5.3), are shown in Figure 5.2. Table 5.1 gives a summary of the clay content found at each borehole. A visible change in soil does not necessarily indicate a change in soil type, as soil colour changes are indicative of oxidation/ reduction conditions and a fluctuating water table. In soil samples obtained from site T for example five changes in soil type

Percentage Clay	Borehole
90 - 100	D3, S2, S3, S4, T1, V1
70 - 89	D1, D2, D5, M3, S1, V2, V3
56 - 66	D4, D6, M1
40 - 50	M2, V5
<39	M4, V4

Table 5.1 Percentage clay found throughout the soil profile at the designated boreholes.

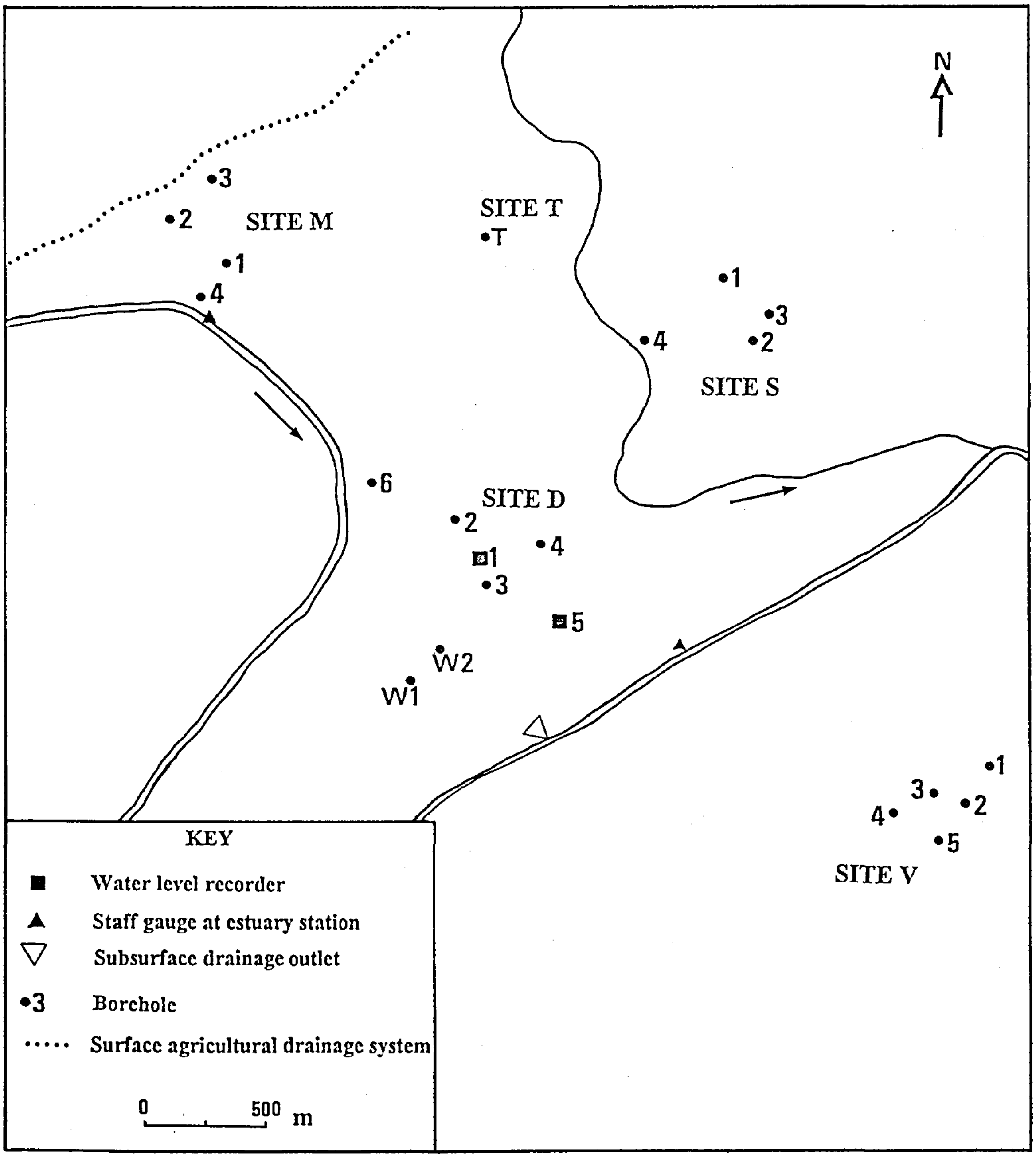

Figure 5.1 Position of the twenty boreholes at the five sites throughout the study area.

were detected, but later soil analysis showed that percentage mud varied between 55.6 and 98.5% and thus all five samples were classified as clay. As seen in Figure 5.2, clay predominates in the soil profiles with thin lenses of sandy clay and sandy clay loam soils.

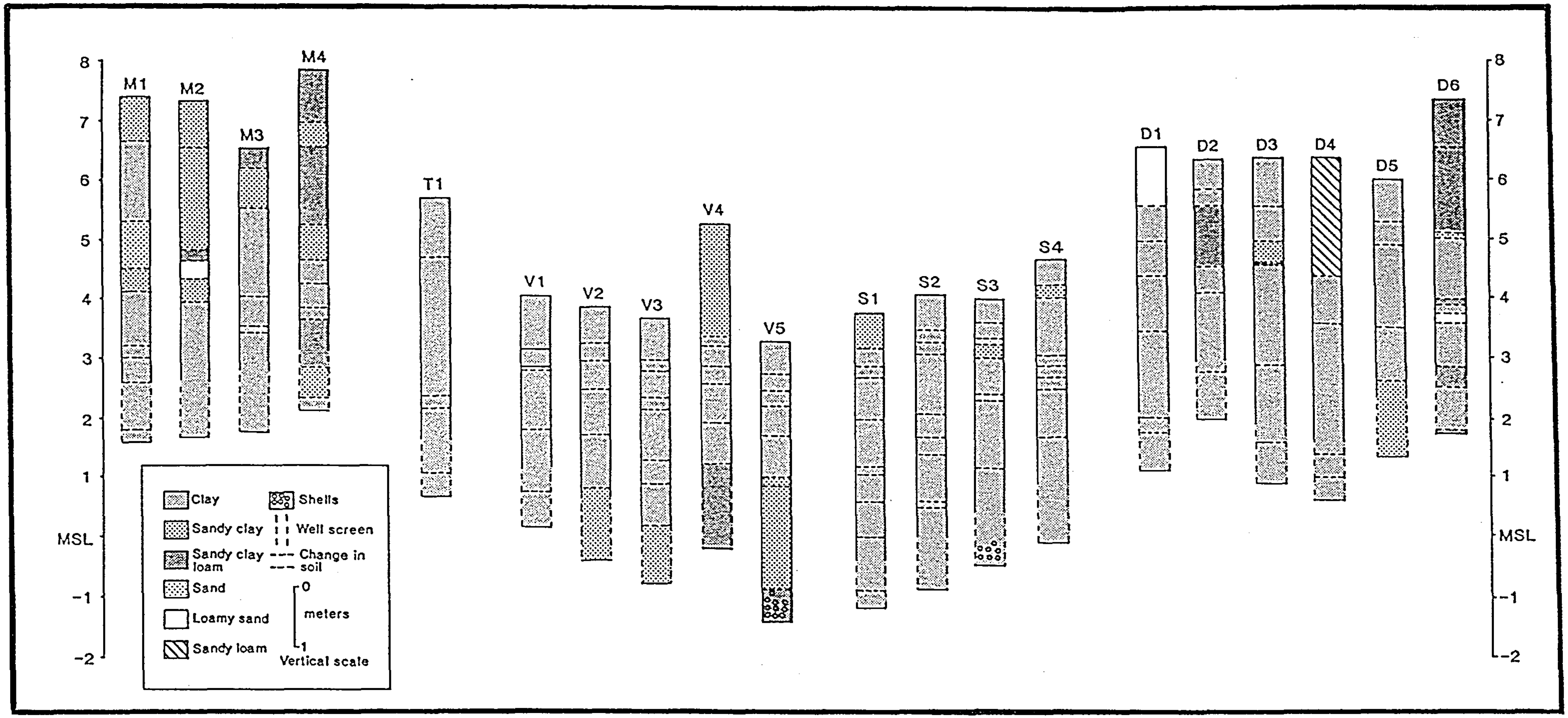

Figure 5.2 Presence of shells, soil texture and entire soil profiles, as determined from analysis of samples obtained during borehole installation, of the twenty boreholes installed in the study area. The depth of the boreholes relative to mean sea level (MSL) is also shown.

Conditions in the unsaturated soil profile (vadose zone) are important in that they determine a number of processes as illustrated in Figure 5.3, which ultimately influence:

- recharge of groundwater, and
- groundwater quality.

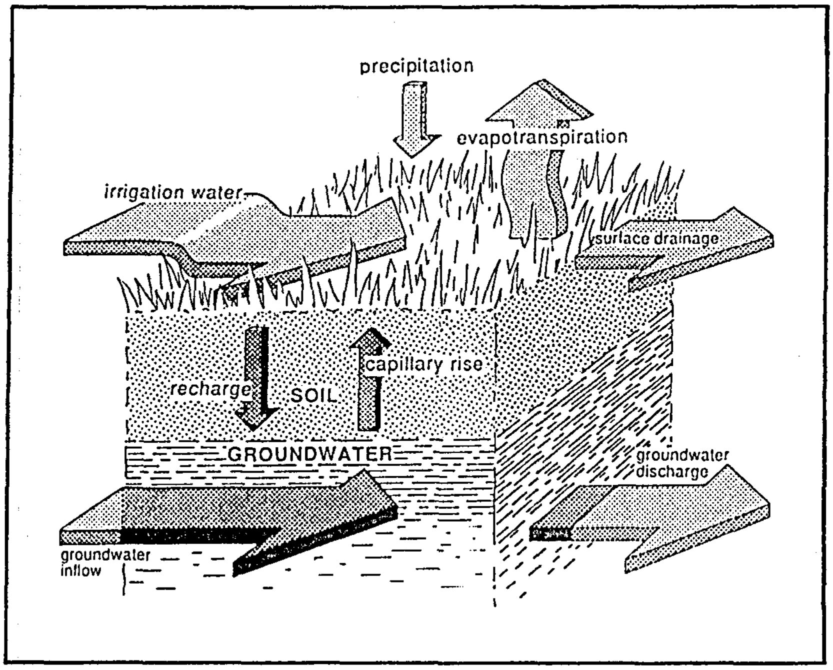

Figure 5.3 Principle of a water balance in an irrigated area (Bos and Wolters, 1994).

Recharge of Groundwater

The level of the water table varies as a function of the depth of the unsaturated zone, hydraulic conductivity, soil properties and other factors (Capone and Slater, 1990). When irrigation water is applied in excess of the crop requirements and in areas of non-uniform distribution of irrigation, larger amounts of deep percolation are generated (Bouwer, 1987; Wichelns and Nelson, 1989). Groundwater flow in excess of evapotranspiration losses will prevent the infiltration of brackish water into an aquifer during tidal flooding and flush salt and other solutes out of the sediment. Where clay lenses restrict the deep percolation of applied irrigation water, waterlogging may occur, making the installation of drainage systems necessary.

Groundwater Quality

Rice *et al.*, (1989) found that where land use varies, large variations in the ionic composition of water in the vadose zone occurs, which in turn contribute to variations in groundwater quality. Irrigation, fertilizer and pesticide inputs to groundwater will depend on the extent of surface runoff, moisture losses and concentration of salts during evaporation, direction of movement in the vadose zone, the quantity of water recharging the groundwater system and the direction of groundwater flow.

5.2 Saturated Hydraulic Conductivity

Hydraulic conductivity values (K) are given in Table 5.2 for those boreholes which met the stringent requirements of the Ernst equation, for the small-scale, in-situ determination of saturated hydraulic conductivity (section 3.6.4).

Borehole	Date of Aquifer Recovery Test	K-value (m per day)
V2	18/06/1994	0.04
V2	13/04/1994	0.03
D3	17/06/1994	0.45
D6	16/06/1994	0.93
M3	17/06/1994	0.38
T1	14/04/1994	0.11
S4	12/04/1994	0.84
13*	15/03/1994	0.44
32*	15/03/1994	0.53

Table 5.2 K-values for the Gamtoos boreholes, and *two auger holes tested in the survey of Botha (1994).

The K-values obtained in the Gamtoos floodplain are within the ranges given by Brassington (1990) for unconsolidated materials (Figure 5.4). Results are also in agreement with the findings of Oosterbaan and Nijland (1994) who state that the finer silt and clay particles of the backwater zone of a floodplain typically have K-values ranging from 0.1 to 0.5 m per day, whilst K-values are generally higher in the coarse soils of the levees bordering an estuary.

Hydraulic conductivity in m per day

10^{-5}	10^{-4}	10^{-3}	10^{-2}	10^{-1}	1	10	10^{2}	10^{3}	10^{4}

Relative hydraulic conductivity

Very low	Low	Moderate	High	Very high

Representative Materials
unconsolidated deposits

Massive clay	Silt, clay and mixtures of sand, silt and clay	Fine sand	Clean sand and sand and gravel	Clean gravel

Figure 5.4 K-values for unconsolidated sediments (from Brassington, 1990).

5.3 Borehole Sites

5.3.1 Site M

The general direction of groundwater movement, as deduced from relative RWLs at site M, is shown by the thick, curved arrows in Figure 5.5. Exceptions to this direction of groundwater flow occurred in June, October and November 1993 and April and June 1994. During these periods RWLs were generally elevated in all boreholes, and groundwater movement was as shown by the thin, straight arrows in Figure 5.5. The water table level relative to mean sea level, during both daily monitoring over consecutive days and routine monthly sampling is shown in Figure 5.6

Groundwater Quality at M

Borehole M4 is located 30 m from the estuary near station 9. The height of the river bank at M4 is approximately 7 m above the mean estuary water level whereas the borehole only extends 5.73 m below the ground surface. Whilst an indication of RWL could be obtained at M4, there was insufficient water in the borehole to obtain water samples, thus the discussion on water quality at site M is limited to boreholes M1, M2 and M3.

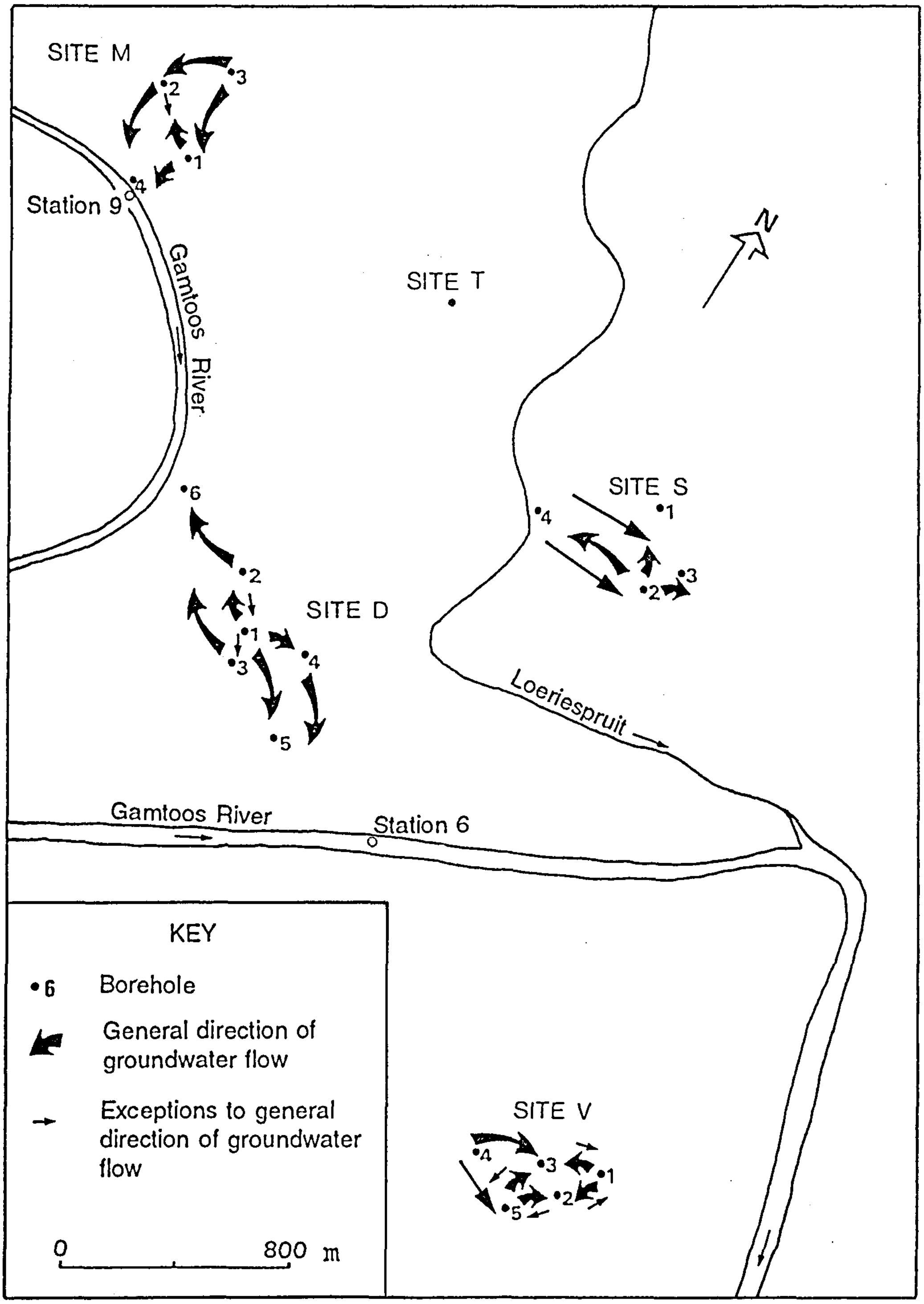

Figure 5.5 The general direction of groundwater flow (thick, curved arrows) and the exceptions to this general direction of flow (thin, straight arrows) at the borehole sites.

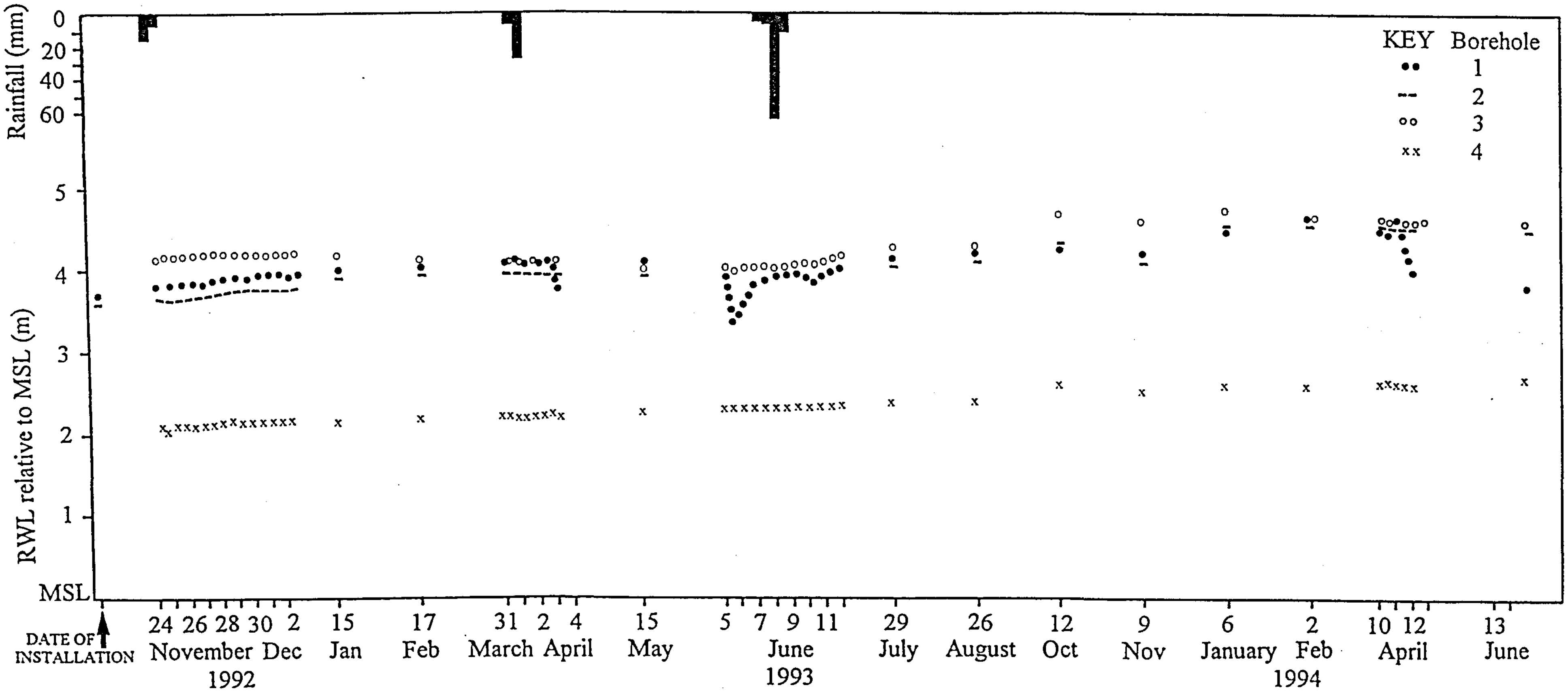

Figure 5.6 The water table level relative to mean sea level, during both daily monitoring over consecutive days and routine monthly sampling in boreholes at site M. Rainfall quantities and the days on which rain was recorded are also shown.

Groundwater quality at M differed from sites D, T, S and V as shown in Table 5.3, which compares the quality of groundwater from individual boreholes at site M with the mean values obtained at the other sites. The EC, sodium, chloride and sulphate concentrations in groundwater from boreholes at site M were much lower than that from boreholes at site D, T, S and V. However, the nutrient content (nitrate- and nitrite- as N, and total phosphorus as P) was greater in groundwater at site M than the other boreholes. The trilinear plot of water chemistry (Figure 5.7) shows that groundwater from all the boreholes except those at site M was chloride type water, and the diamond plot types them as chloride-sulphate, bicarbonate water whereas groundwater at M showed no dominant type.

WQ	M1	M2	M3	D	T	*S	V	**SEA WATER
EC $mS.m^{-1}$	162	264	266	1445	3559	5534	3762	-
Na $mg.l^{-1}$	334	482	597	3064	7052	13966	3762	10770
Cl $mg.l^{-1}$	337	371	402	4609	12773	22706	8482	19340
SO_4 $mg.l^{-1}$	127	177	165	1416	4939	7582	14304	2712
NO_3 -N $mg.l^{-1}$	6.04	27.08	23.95	0.10	0.10	<0.10	0.10	-
NO_2 -N $\mu g.l^{-1}$	719.50	378.69	886.34	93.00	30.47	8.51	82.17	-
P $mg.l^{-1}$	0.91	1.35	2.35	0.60	0.53	2.24	0.42	-

Table 5.3 Mean chemical composition of groundwater abstracted from boreholes at sites D, T, S and V, and compared with the individual borehole results for site M.

* Boreholes S1, S2 and S3 only.

** Kennish, 1986 (generally accepted value).

The Influence of the Surface Agricultural Drainage System on Groundwater at Site M

The agricultural drainage system, discussed in sections 2.2.7 and 4.2, was designed to drain the upper 1.2 m of the vadose zone and accept surface runoff. The drainage system has attained the purpose for which it was designed, since it has resulted in the lowering of the water table from that shown in Figure 2.7, to depths of between 2 and 3 m below the ground surface at boreholes M1, M2 and M3. Even following the rain of June 1993, the water table at the shallowest of boreholes at site M (M3) at no time rose above 1.76 m below the land surface. As stated in section 5.1, conditions in the vadose zone influence both the recharge and quality of groundwater. The low salt

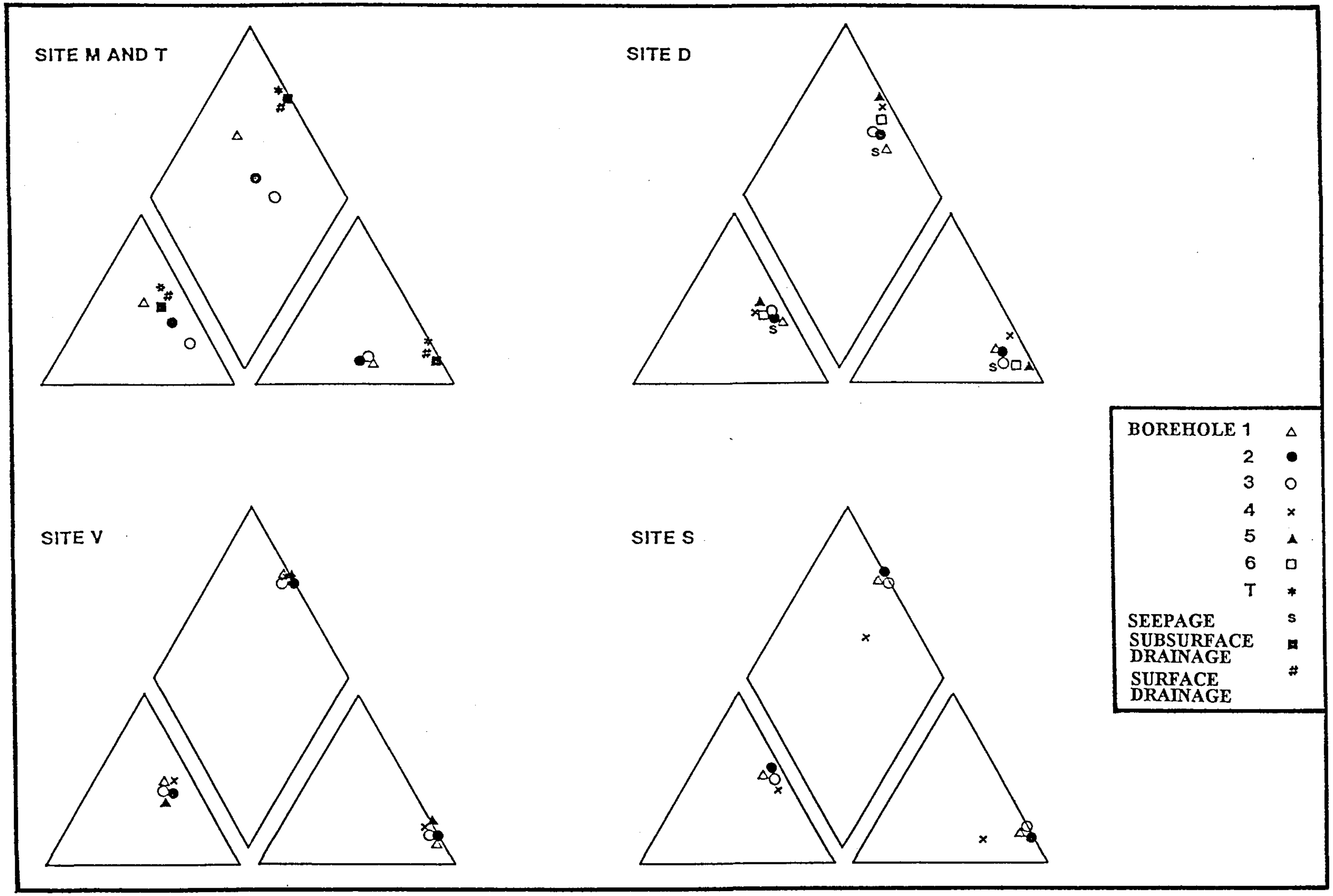

Figure 5.7 Trilinear plot of water chemistry (based on Figure 3.1) showing the grouping of similar types of groundwater, a seepage, the agricultural drainage discharge and discharge from the subsurface drain outlet.

concentration of groundwater at site M compared to that of the other boreholes in the Gamtoos floodplain (Table 5.3), may be partly attributed to the adequate flow of water (both surface runoff and water within the vadose zone) towards the drainage system.

5.3.2 Site T

The borehole at site T was originally installed to test the suitability of the vibracore equipment in the Gamtoos floodplain; it is situated 400 m from the Loeriespruit (Figure 5.1), in a field that had been abandoned by the farmer. Poor drainage in the field had caused waterlogging and salinization of the soil, rendering low crop yields. As borehole T1 stands alone, the direction of groundwater flow could not be determined at the site.

Response of RWLs to Rainfall

The lowest RWL recorded at borehole T (1.72 m below the land surface) was at the time of borehole installation (August 1992). Figure 5.8 shows the RWLs at borehole T, relative to mean sea level, during daily measurement periods and once-off monthly measurements. Following the rain on 24 November 1992 (15 mm) and 25 November (6 mm), a 6.7 cm rise in water table was measured three days later. Thereafter the water level declined steadily, as shown in Figure 5.8. Dry conditions followed these measurements with intermittent irrigation occurring in the vicinity of the borehole. The duration and quantity of irrigation as well as its proximity to the borehole varied during this period allowing little correlation to be drawn between response time of the RWL to irrigation inputs.

On 31 March 1993, the RWL at T was 1.66 m below the ground surface; during this measurement an adjacent field (approximately 100 m away) of seedling cabbages was under irrigation. The following day a 5 cm rise in water table was detected (the RWL was 1.61m below the ground surface). The RWL fluctuated during subsequent measurements. On 2 and 3 April, 8 mm and 27.4 mm of rain fell respectively. Irrigation continued in the cabbage field despite the rain. The RWL on the morning of 3 April was 1.66 m below the surface and rose 5.7 cm by the following morning, but by 5 April it had dropped 2.7 cm to 1.63 m below the surface. When measured on 6 April the water table had risen to 1.59 m.

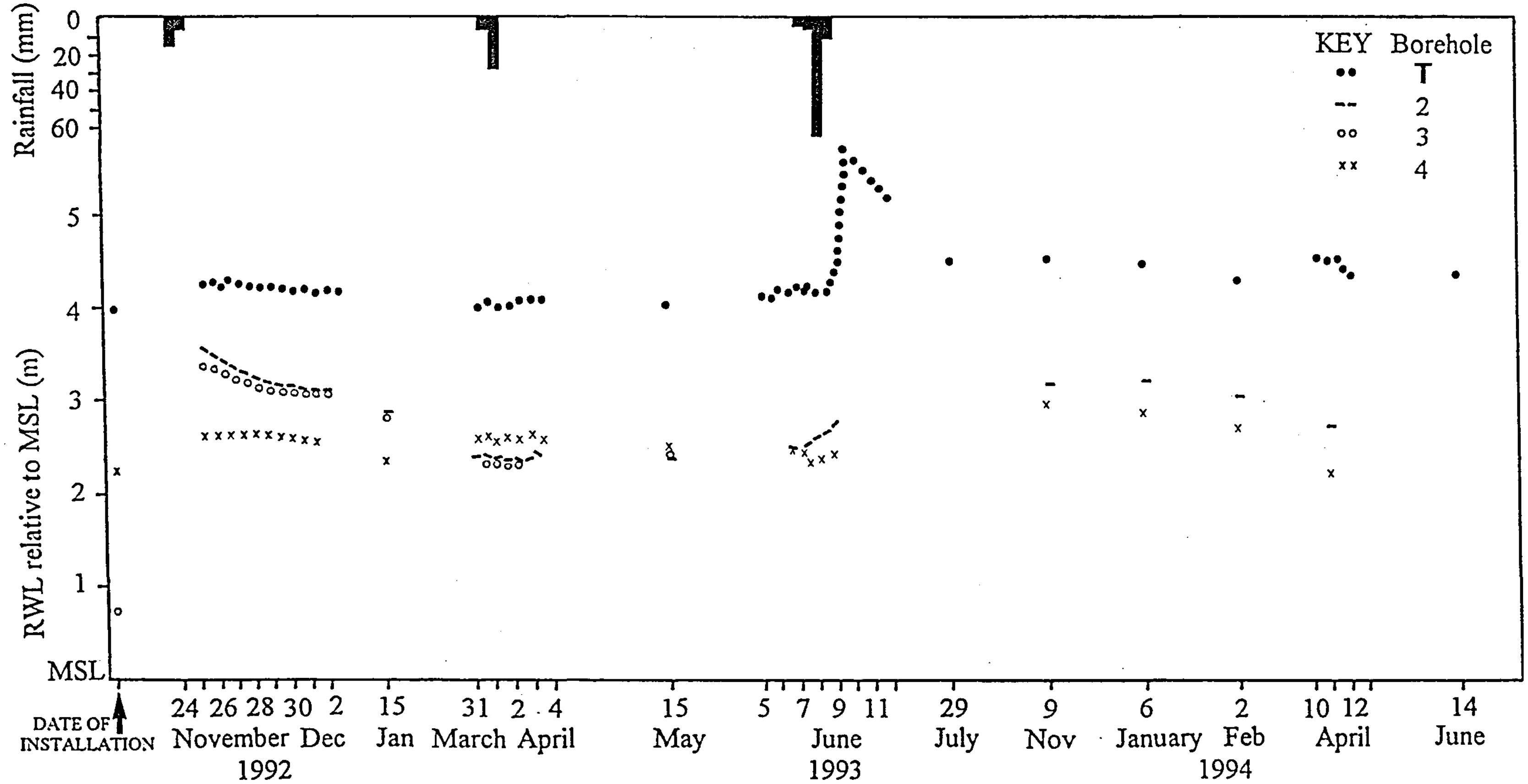

Figure 5.8 The water table relative to mean sea level, during both daily monitoring over consecutive days and routine monthly sampling in boreholes at sites T and S. Rainfall quantities and days on which rain was recorded are also shown.

On 5 June 1993, the RWL was 1.53 m below the ground surface (28.4 mm of rain fell on the 3rd). The RWL rose steadily to 1.45 m below the ground surface on the 9th. During the period 10 to 13 June an additional 87.2 mm of rain was recorded. Figure 5.8 shows the almost immediate response of the water table at borehole T to this rain, when on 14 June there was standing water at the borehole site. The water level subsequently dropped slowly and when measured again five days later (19 June) was 0.52 m below the land surface i.e. the rate of drop was 0.10 m per day. The estimate of hydraulic conductivity (K) determined from applying the Ernst equation to aquifer recovery test data was 0.10 m per day. Following the above mentioned June 1993 measurements, five once-off monthly measurements and four continuous days of measurement in April 1994 occurred during the subsequent twelve month period. During this time the RWL was on average 0.30 m higher (Figure 5.8) than during the sampling period prior to the June 1993 rain.

5.3.3 Site S

At the time of borehole installation, the level of the water table was very low compared to RWLs during the remainder of the study period. Rainfall during the six months preceding borehole installation totaled 158.5 mm (83.3 mm below the twenty year running mean for the area), and farmers were on reduced percentage allocation of water quotas as mentioned in section 2.2.6. The water table at S3 was a mere 0.74 m above mean sea level (MSL) compared to the average during subsequent routine sampling of 2.68 m above MSL. Should the water table drop below MSL saltwater intrusion could render the soils highly saline.

Flooding and Drainage at Site S

Prior to the onset of monitoring in November 1992, 15 mm and 6 mm of rainfall were recorded on 24 and 25 November respectively. This rainfall together with flooding of the Loeriespruit were responsible for flooding of site S. Access to boreholes was only possible by wading ankle deep through standing flood water in the fields.

The surface water had drained by 26 November when RWLs were again measured. The water table was 0.12 m below the ground surface at borehole S1, 0.53 m at S2, 0.68 m at S3 and 2.11 m at S4. The RWLs were not measured again until seven days later (3 December) by which time the water table had dropped to 0.66 m at S1, 0.89 at S2, 0.91 at S3 and had risen slightly in borehole S4 to 2.08 m. Four days later the water table at S1 had only dropped by a further 9 cm, by 7 cm at borehole S2 and by 2 cm at S3.

Borehole S1 was irretrievably damaged by cows in March 1993 and it is thus omitted from further discussion. The water table gradient at site S was so slight that drainage directions changed continuously as shown in Figure 5.5. When the RWL was higher in borehole S4 in relation to S2 and S3 (March-April 1993; May 1993) centripetal drainage occurred towards the vicinity of boreholes S2 and S3. RWLs in boreholes S2 and S3 were low in April 1993, and in S4 in January 1993 and April 1994. These results correlate with the findings of a drainage survey done in March 1994 by Botha (1994). Waterlogged soils and poor drainage exacerbated by the frequent flooding of site S, prompted the farmer (Mr M. Snyman) to approach the Department of Agriculture, Port Elizabeth for a solution to the problem. Botha's survey was conducted over two days during which the auger-hole method for the determination of saturated hydraulic conductivity as described by Oosterbaan and Nijland (1994) was applied. Thirty-two holes were augered throughout the problem area to water table depth. The position of the holes, land surface level and depth to water table at each hole were surveyed, to the nearest cm. Figure 5.9, based on Botha's findings, shows the depth to the water table and the direction of groundwater movement at site S during March 1994. Botha applied the Ernst equation (section 3.6.4) to data obtained from auger-holes 13 and 32 (the positions of which are shown on Figure 5.9): these yielded hydraulic conductivity values of 0.44 m per day and 0.53 m per day respectively. An aquifer recovery test conducted on borehole S4 on 12 April 1994 yielded a hydraulic conductivity of 0.84 m per day.

Apart from the November 1992 inundation, flooding occurred again in June 1993 following a 24 hour record rainfall. There was no rain in July and 43 mm fell in August. The land was again under water in September 1993 following 143.7 mm of rain (i.e. total rainfall during September). Although only 3 mm of rain fell on 4 October, the fields were still flooded on 12 October, thereby preventing sampling. No measurements were taken until 9 November by which time the standing water at S2 and S4 had dropped to 0.91 m and 1.74 m below the ground surface respectively.

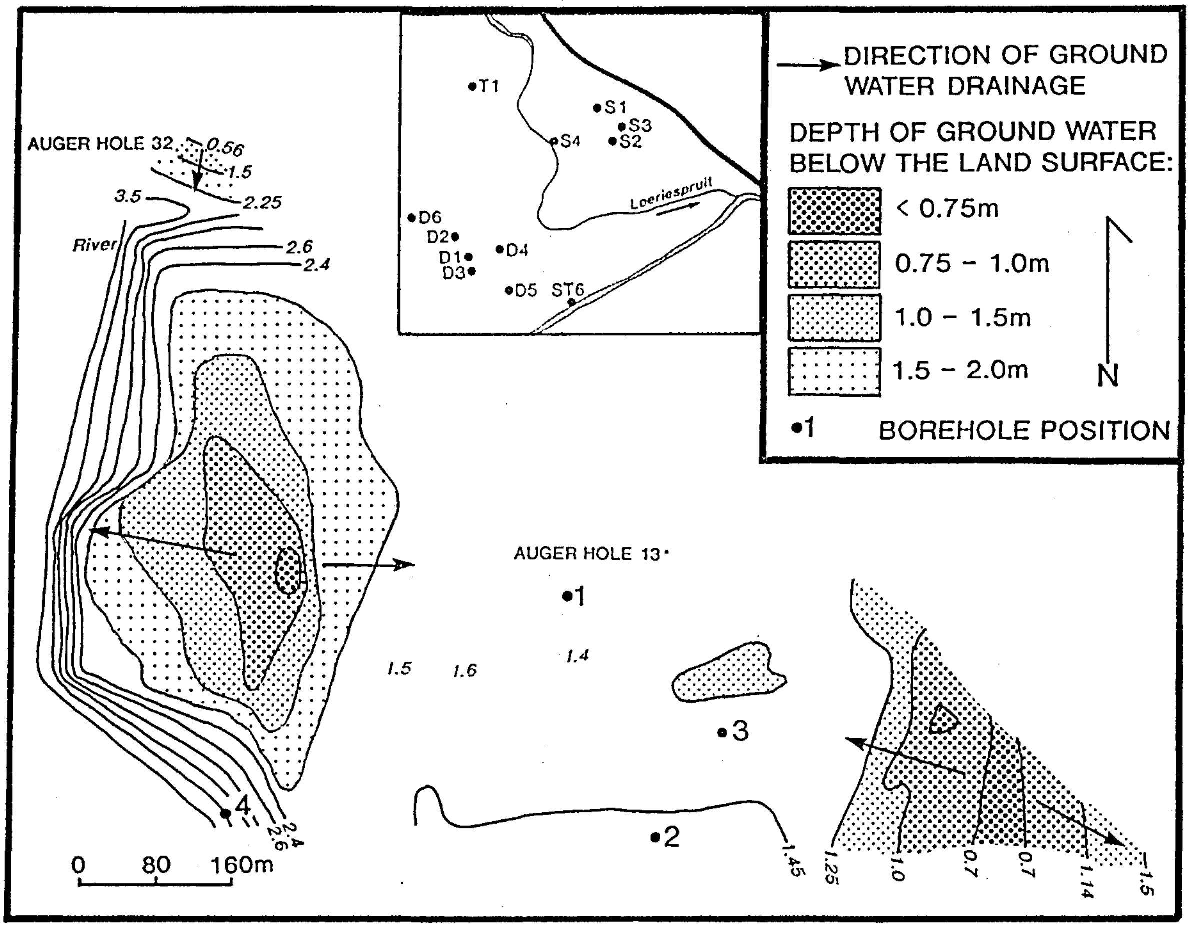

Figure 5.9 Depth of the water table and groundwater flow directions at site S during March 1994 (based on the findings of Botha, 1994).

Groundwater Quality at Site S

Shells found at depth during the installation of borehole S3, as well as the high EC, sodium, chloride and sulphate content of groundwater abstracted from boreholes S1, S2 and S3 (given in Table 5.3) indicate that the groundwater quality at site S was influenced by past marine activity. As was the situation at site V, discussed in section 5.3.4. These results may have been influenced by the depth at which groundwater was abstracted from boreholes S1, S2 and S3. Although the RWLs in the boreholes were between 2 to 3 m above mean sea level as shown in Figure 5.8, Figure 5.2 shows that the screen position of the boreholes was below mean sea level. Furthermore, groundwater flow frequently occurred towards the three boreholes as shown in Figure 5.5 which would account for elevated salt concentrations during that period.

Groundwater quality at borehole S4 was markedly different to that of S1, S2 and S3, as shown in Table 5.4. Initially the elevation of the land surface in the vicinity of borehole S4 was high relative to the Loeriespruit and the other boreholes, allowing adequate surface runoff. However, because of drainage problems experienced in the central part of site S, a number of shallow trenches leading excess water towards the Loeriespruit were dug. One such trench was located adjacent to borehole S4, this caused water to pool in the trench next to the borehole. A corresponding deterioration in groundwater quality at borehole S4 is shown in Table 5.5. Results given in Table 5.5 show however that total alkalinity and the nutrient content of groundwater at borehole S4 was variable. The nitrate, nitrite and total phosphorus content of groundwater obviously varied in response to the time and quantity of fertilizer application.

5.3.5 Site V

Topography to the west of the estuary shows evidence of meander scrolling in the form of bar-and-swale terrain, with numerous water filled depressions. The past migration of estuarine waters across the floodplain would result in the deposition of saline sediments. Shells were found at depth during the installation of boreholes V2, V3, V4 and V5. Shells taken from V5 at a depth of 4 m were identified as *Natica gualteriana*, which today exists in tropical Indo-Pacific regions and on the east coast of South Africa as far south as Port Alfred (Zhang *et al.*, 1996a). Zhang *et al.*, (1996b) collected a number of fossils throughout the Gamtoos valley and after having them dated concluded that the Gamtoos paleoestuarine environment was previously similar to that of present day Natal i.e. the sea temperature was 5^{0}C warmer and the average air temperature 3.5^{0}C warmer than at present in the area.

Land use at site V differs from that in the Loerie Flats (sites T, D and M). In the area west of the estuary but north of the N2 highway (site V - Figure 2.2) there is less intense rotational cropping; fodder crops and grazing predominate. These did not require the same quantity of fertilizer and pesticide application as the other sites. It was thus decided to monitor such an area (site S is also mainly fodder cropping) to investigate any related difference in groundwater characteristics.

Borehole	EC ($mS.m^{-1}$)	Sodium ($mg.l^{-1}$)	Chloride ($mg.l^{-1}$)	Sulphate ($mg.l^{-1}$)
V1	4006	8798	17977	4493
V2	4051	9015	15092	4060
V3	4103	8203	13310	4391
V4	3845	9383	14545	5572
V5	2328	5970	7859	3159
D1	865	1942	2237	960
D2	733	1789	1659	625
D3	652	1317	1853	492
D4	1598	3634	4125	2736
D5	3320	7659	12724	2699
D6	680	1311	2456	546
S1	4140	10600	16280	6970
S2	5745	13626	24477	6762
S3	5560	15047	21866	8688
S4	449	855	1189	263
T	3559	7052	12773	4939

Table 5.4 Mean EC, sodium, chloride and sulphate values of groundwater abstracted from individual boreholes at sites V, D, S and T.

Parameter mg.l^{-1} unless otherwise specified	3/12/92	15/01/92	06/04/93	15/05/93	9/11/93	2/02/94
EC mS.m^{-1}	258	274	447	491	578	600
Tot. Alkalinity	422	706	593	875	650	807
Calcium	114	90	109	162	152	241
Magnesium	166	193	250	372	518	688
Sodium	450	455	596	1007	1104	1326
Potassium	8.4	10.7	8.7	13.0	15.0	25.0
Chloride	480	410	688	1186	1621	2687
Sulphate	144	157	185	313	318	371
Nitrate-N	0.15	0.15	0.10	1.48	0.10	0.39
Nitrite -N $ug.l^{-1}$	3	4	26	203	29	29
Total PO_4-P	0.07	0.27	0.10	0.30	0.09	0.10

Table 5.5 Deterioration of groundwater quality detected at borehole S4 during routine sampling from 3 December 1992 to 2 February 1994.

Drainage at Site V

The level of the water table in boreholes at site V, relative to mean sea level is shown in Figure 5.10. Generally groundwater movement at site V occurred as shown by the thick, curved arrows in Figure 5.5, i.e. towards a water table depression between boreholes V2 and V3. Groundwater drainage was spatially and temporally variable and deviations from the flow paths occurred in December 1992 and on 14 June 1993, when drainage occurred towards two centres located at V1 and V5. In February 1993 drainage occurred towards V3 and V5, whilst on 5 June 1993 there was a depressed

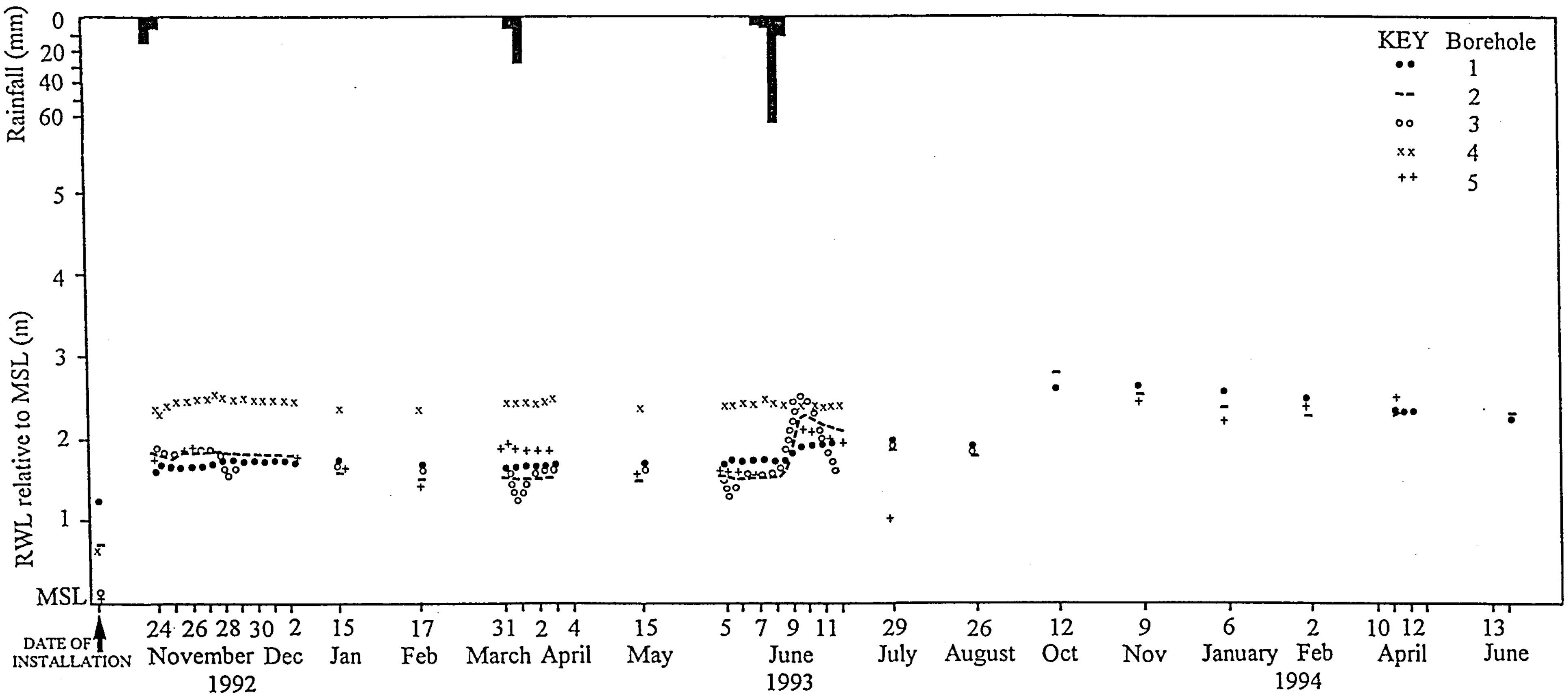

Figure 5.10 The water table level relative to mean sea level, during both daily monitoring over consecutive days and routine monthly sampling in boreholes at site V. Rainfall quantities and the days on which rain was recorded are also shown.

area in the vicinity of V2. The slight hydraulic gradient at the site resulted in frequent changes in the direction of groundwater flow and led to standing water in areas of inadequate drainage. Further evidence of sluggish groundwater movement is the low K-value obtained at borehole V2 (0.03 m per day) which was an order of magnitude slower than borehole T (0.11 m per day) which exhibited the second slowest K-value of the boreholes tested.

Groundwater Quality at Site V

Groundwater quality is inversely related to groundwater movement. Table 5.4 shows the high levels of sodium, chloride and sulphate at site V, and the trilinear plot of water quality (Figure 5.7) classifies groundwater at V as a chloride-sulphate type. Groundwater quality at specific boreholes at site V is discussed further in section 5.6.

5.3.5 Site D

The six boreholes at site D covered the largest area of the borehole groupings, a distance of 920 m separated boreholes D5 and D6, with boreholes D1 to D4 spaced between these two (Figure 5.1). D6 was situated approximately 70 m from the estuary (near station 8). The shortest distance from D5 to the estuary (near station 6) was approximately 320 m. Water quality at site D was variable. Borehole D6 bordered a field which was unused for most of the duration of the study period until February 1994 as a result of previous poor crop yields from the land.

Drainage at Site D

The level of the water table relative to mean sea level, during both daily monitoring over consecutive days and routine monthly sampling of boreholes at site D is shown in Figure 5.11. Rainfall quantities and the days on which rain fell are also shown in the figure. Drainage generally occurred towards the estuary, moving outward from an elevated water table in the vicinity of boreholes D1 and D3 as shown in Figure 5.5. Deviations from this general direction of groundwater flow occurred in December 1992, January and November 1993 and February and April 1994, when an elevated water table was in the vicinity of D2. Groundwater quality and direction of flow at D4 and D5 differed from that of the other boreholes at site D, flowing southward towards the estuary. Despite the similarity in cropping practices, frequency of irrigation and nature of crops throughout site D

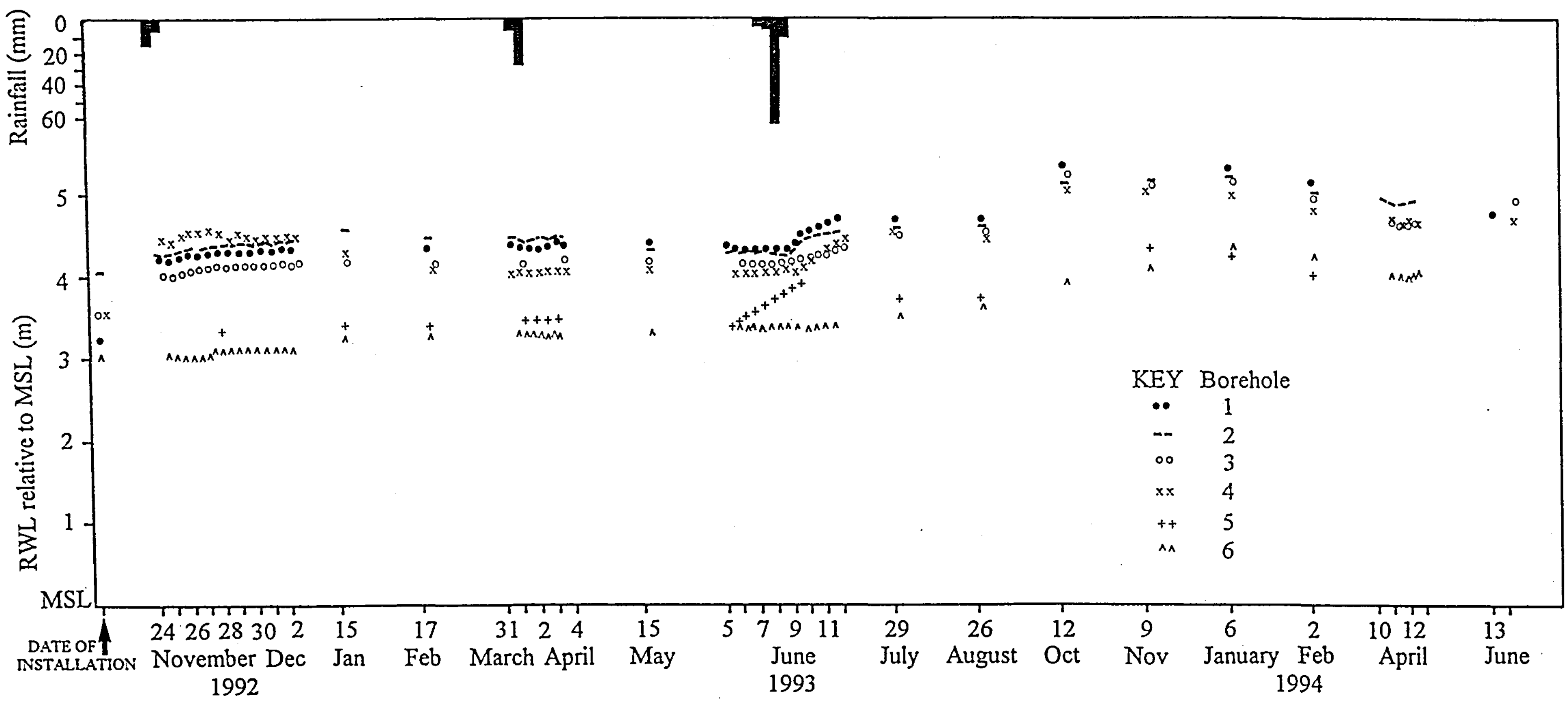

Figure 5.11 The water table level relative to mean sea level, during both daily monitoring over consecutive days and routine monthly sampling in boreholes at site D. Rainfall quantities and the days on which rain was recorded are also shown.

(with the exception of an unused field near D6), groundwater was markedly more saline in these two boreholes.

Groundwater Discharge

A rough estimate of groundwater discharge to the estuary from the Loerie Flats was obtained using Darcy's equation (section 3.6.4). The estimate is by no means statistically valid and merely serves to give an indication of the groundwater input to the estuary. Data used in the calculations were as follows: an average K-value based on the K-values given in Table 5.2 was applied; actual distances and elevation heads between boreholes were used for specific dates (dates were chosen close to the time of K determination and when a complete data set was available). From this an average was obtained for the Loerie Flats. Throughout the study period the boreholes at site D intercepted a depth of approximately 4 m of water i.e. the difference between depth to RWL from the ground surface and depth of the borehole. The distance from station 9, following the circumference of the estuary/land border to the Loeriespruit-Gamtoos confluence is 7.7 km. Thus for a depth of 4 m, the cross-sectional area of 30 800 m^2 (7700 m x 4 m) was applied to Darcy's equation. The groundwater discharge to the estuary over the 7.7 km stretch was a conservative estimate of 276 m^3 per day. Clearly a similar discharge from the west bank and the entire expanse of the estuary, as well as deeper groundwater inputs would yield a substantial groundwater discharge to the estuary.

Material having a low hydraulic conductivity, such as muddy sediments, exhibit a great deal of spatial variability and will thus influence the accuracy of estimates of water flux (Nuttle and Harvey, 1995). The K-value of a soil profile can vary with depth and spatially within a soil layer. K-values at a particular point can also change over time, although this occurs mainly in clays that are subject to alternate periods of wetting and drying (Bos, 1994). Table 5.1 shows that soil samples obtained in the Gamtoos floodplain from a profile of sixteen of the twenty boreholes had a clay content greater than 56%. Thus should similar tests be repeated at a later stage results could vary but would probably be within a specific range. Further compounding the estimation of groundwater discharge is that the rate and direction of discharge will change in response to the hydraulic gradient to the estuary.

5.4 Seepages

Ritzema (1994a, p. 1103) defines a seepage as the '*slow movement of water through small cracks, pores or interstices of a material, in or out of a body of surface or subsurface water*'. Seepages were observed on numerous occasions along the banks of the Gamtoos estuary, particularly between stations 7 and 8. Whilst they were mostly observed in this area, it is noted that thick bank vegetation may have obscured the observation of seepages in other areas. One such seepage, approximately 70 m from borehole D6, was sampled on 9 November 1993. Table 5.6 provides a comparison of groundwater obtained from the three closest boreholes to the seepage (D1, D2 and D6) on 9 November to that of the seepage. Similarities in the chemical composition of the seepage and the average constituents found in groundwater at D1, D2 and D3 during the study period, are evident on comparing Tables 5.4 and 5.6 and groundwater type as shown in Figure 5.7. The groundwater flow paths shown in Figure 5.5 indicate that the groundwater had moved from an area of recharge in the vicinity of boreholes D1 and D3 towards the estuary near borehole D6. Thus any seepage will have characteristics similar to that of the aquifer through which it had moved.

WQ	EC mSm^{-1}	Calcium $mg.l^{-1}$	Magnesium $mg.l^{-1}$	Sodium $mg.l^{-1}$	K $mg.l^{-1}$	Chloride $mg.l^{-1}$	Sulphate $mg.l^{-1}$	Nitrate-N $mg.l^{-1}$	Nitrite-N $\mu g.l^{-1}$	Total P $mg.l^{-1}$
D1	652	243	419	1359	2	1463	795	0.1	0.0	0.72
D2	630	503	696	1520	89	1818	687	0.1	0.01	0.07
D6	706	380	729	1596	75	1706	449	0.1	764	0.5
Seepage	595	281	391	1317	15	1460	324	9.1	260	0.2

Table 5.6 Chemical composition of seepage flow compared with groundwater abstracted from nearby boreholes on 9 November 1993.

5.5 The Subsurface Drainage System

MacEwan *et al.*, (1992) and Beck (1984, cited in Nakamoto and Hassler, 1992) give reference to losses of 16% and 10% respectively of total annual crop production in Murray valley, Australia and San Joaquin valley, California. Crop losses were caused by saline salt build up following waterlogging. Where waterlogged lands or poor drainage and a shallow water table exist, subsurface drainage systems are installed to overcome the problem. The systems are designed to prevent crop damage from a saline, high water table by draining excess water from the root zone. Salts are leached out of the soil by the drainage water thereby reducing or preventing build up of salts in the soil profile (Oosterbaan, 1988). Soil nutrients however are removed simultaneously in this manner. The drier soils lead to improved soil aeration and stabilized soil structure allowing the soil to be easily worked.

A subsurface drainage system culminating in a flowing gravity outlet (Figure 2.8) was located on a farm in the Loerie Flats. To determine the groundwater quality within the drainage system, two boreholes (W1 and W2 - Figure 5.1) were installed on 17 January 1994 within the bounds of the drainage system. Boreholes W1 and W2 were only sampled on one occasion (2 February 1994) before financial constraints prevented further sampling. Thus while results are not statistically valid in this area, they provide further insight to groundwater quality and movement within the Loerie Flats.

In the study area, the subsurface drain outlet was almost inaccessible as it discharged near the base of a sheer river bank through dense undergrowth and reeds. Results of analysis of the subsurface drain discharge, together with groundwater analyses (on 2 February 1994) from the two boreholes within the system (W1 and W2), and D5 (in the neighbouring field), are given in Table 5.7. The discharge from the drain outlet on the day of sampling was 0.03 $l.s^{-1}$. The drain discharges into the estuary just upstream of station 6. Although some mixing and dilution of nutrients will have occurred downstream from the drain outlet, elevated nutrient levels were measured on occasions at station 6 (section 4.7). It is possible that these high nutrient levels are related to the subsurface drain input, however as the drain and estuary were not sampled at the same time there is no data to support the supposition.

Although no long-term discharge data was obtained, factors generally influencing the quantity of water collected by subsurface drainage systems include antecedent moisture conditions, land use, the volume of rainfall and irrigation inputs, soil characteristics and the location of the drainage system (Reid and Parkinson, 1984; Wichelns and Nelson, 1989). Wichelns and Nelson (1989) attributed an estimated 27% of water collected in a subsurface drainage system to lateral subsurface inputs from neighbouring non-drained fields. This would account for the similarity in water type of the subsurface drain discharge and that of groundwater from boreholes D4 and D5 in the Gamtoos study area. The salt content of the subsurface drainage effluent is indicative of the quality of water recharging the aquifer which in the vicinity of the system was 12 883 $mg.l^{-1}$ (total dissolved solids). Rice *et al.*, (1989) however found that water from agricultural drains in the fine-textured soils of Salt River valley, south central Arizona, had a salt content of a few hundred $mg.l^{-1}$. A comparison of water quality of two tributaries of the San Joaquin River, California, by Nakamoto and Hassler (1992) revealed that the Salt Slough had a significantly higher conductivity (183 $mS.m^{-1}$), total alkalinity, turbidity and lower DO than the Merced River (conductivity 21 $mS.m^{-1}$). They attributed the difference to the degraded quality of drain water entering Salt Slough from tile drained fields whereas the Merced River received only surface agricultural runoff.

WQ	*EC	Ca	Mg	Na	K	Cl	SO_4	NO_3	#NO_2	P
Drain	1952	1125	2503	3528	34	8703	2039	0.1	27	0.3
W1	1751	584	1987	3134	56	16012	7257	0.1	7.5	0.1
W2	1572	489	1635	4176	58	7672	136	x	8.4	0.3
D5	2790	1829	4840	7944	318	13890	2647	0.1	181	0.1

Table 5.7 Chemical composition of the discharge from the subsurface drainage system compared with groundwater from boreholes W1, W2 and D5 on 2 February 1994. Units are $mg.l^{-1}$ throughout, except for *EC which is given in $mS.m^{-1}$ and #NO_2 which is given in $ug.l^{-1}$. x - no data available.

5.6 Groundwater Quality Variables

5.6.1 Sodium and Chloride

Groundwater containing high sodium and chloride concentrations may indicate contact with water of marine origin (Bouwer, 1978). The chloride content of groundwater varies depending on climate, geology and other conditions. Chloride has the tendency to accumulate as it is not easily removed by natural processes and is not significantly influenced by exchange and adsorption (unlike sodium) and biotic activity (Davis and De Wiest, 1966).

Of the groundwater tested, that obtained from sites S and V contained the highest sodium and chloride. Groundwater from borehole S3 had an average sodium concentration of 15 047 $mg.l^{-1}$, borehole S2 of 13 626 $mg.l^{-1}$ and S1 10 600 $mg.l^{-1}$. At site V the sodium concentration of groundwater ranged from 5 970 $mg.l^{-1}$ in borehole V5 to 9 383 $mg.l^{-1}$ in borehole V4. The chloride content of groundwater at site S was high in boreholes S1 (16 280 $mg.l^{-1}$), S2 (24 477 $mg.l^{-1}$) and S3 (21 866 $mg.l^{-1}$). The chloride content of groundwater at site V ranged from 7 859 mg l^{-1} (V5) to 17 977 mg l^{-1} in borehole V1. As shown in Table 5.4 the sodium and chloride concentration of groundwater abstracted from borehole T was also high (7 052 and 12 773 $mg.l^{-1}$ respectively). The elevated sodium and chloride levels at site S and V could be partly attributable to the position of the borehole screen, Figure 5.2 shows that the screen depth was below mean sea level in all but borehole V1. Thus saltwater intrusion during previous dry periods as well as past marine influences, and in the case of site V - the previous meandering of the river across the floodplain, would result in elevated sodium and chloride levels in groundwater.

Sodium has a dispersant effect on soil properties which in turn will hinder soil drainage. According to Bos (1994) well structured clay and clay loam soils have transmissivities ranging from 0.5 to 2 m per day whereas similar soil types which are poorly structured range from 0.002 to 0.2 m per day. The hydraulic conductivities determined at site S ranged from 0.4 to 0.8 m per day, whereas those determined at borehole V2 on two occasions were 0.03 and 0.04 m per day respectively.

5.6.2 Sulphate

Soils taken from site T during borehole installation were dark in colour indicating waterlogged conditions and the presence of iron sulphide which often occurs in intertidal sediments having a high organic content (Kennish, 1986). Groundwater samples from T frequently had a strong hydrogen sulphide smell and were black in colour indicating the presence of sulphate reducing bacteria and anoxic conditions (Holden, 1986). Sulphates present in the groundwater are reduced by the action of the bacteria to form hydrogen sulphide which then combines with iron oxides to form ferrous sulphide. It is the ferrous sulphide which imparts the black colouration to sediments and groundwater.

In general, the sulphate concentration in groundwater varies considerably depending on geologic and soil conditions. As most sulphate compounds are readily soluble in water, groundwater associated with sedimentary rocks (particularly organic shales) may have a high sulphate concentration. Past flooding in the Gamtoos floodplain will have introduced water of marine origin to the land, and as sea water contains a high sulphate concentration, this would impart a high sulphate content to the soil. According to Davis and De Wiest (1966) groundwater which has passed through sediments derived from igneous and metamorphic rocks generally has a sulphate content of less than 100 $mg.l^{-1}$. Although sulphates are removed from groundwater by the action of sulphate reducing bacteria, the sulphate content of groundwater abstracted from borehole T was high with a mean of 4 939 $mg.l^{-1}$. Groundwater at site S (excluding borehole S4, which borders on the Loeriespruit) and site V also had very high sulphate concentrations (as shown in Table 5.4). Many fertilizers contain sulphate (such as ammonium sulphate nitrate - which was commonly used in the study area), and are thus responsible for elevated sulphate levels in groundwater. Whilst groundwater at site M contained elevated nutrient levels, the suphate content was low (mean 158 $mg.l^{-1}$). The high sulphate content of the groundwater at sites T, S and V, given the high sodium and chloride content, low hydraulic conductivity and low nutrient content, can be partly attributed to the nature of the sediments rather than fertilizer application alone.

5.6.3 Nutrients

Application of Fertilizers in the Loerie Flats

It was difficult to estimate the quantity of fertilizers and pesticides applied throughout the study area as the farmers became suspicious and reluctant to talk on the subject for fear of reprisals. This was exacerbated by the complexity of changes in crop type throughout the study area at any time (section 2.2.5).

Nitrate and Nitrite as Nitrogen

Under natural conditions, i.e. prior to any artificial application of nitrogenous fertilizers, nitrogen can be incorporated into living matter and later converted to ammonia by soil bacteria. Ammonia may then be nitrified to nitrite and then nitrate. This nitrate in the soil may then be used by crops while it can accumulate and undergo subsequent leaching from the soil or be converted to nitrogenous gases by denitrification or volitization (if converted to ammonia); (Burt and Haycock, 1991).

There are generally two categories of processes which influence the leaching of nitrate from the soil root zone. Those that influence the movement of water and thus transport of nitrate in the soil i.e. soil type (Sabol *et al.,* 1987; Ritter *et al.*, 1990), hydraulic conductivity and depth of the unsaturated zone. Those that influence the nitrogen turnover and thus the nitrate concentration in the leachate percolating to the groundwater zone include: redox conditions in the soil profile, crop cover, timing, quantity and type of fertilizer (Cuttle, 1989; Strebel *et al.*, 1989; Heng *et al.*, 1991; Tredoux, 1993). The type of fertilizer applied will determine the amount of effective nitrogen available (Bareket, date unknown), furthermore the susceptibility to leaching will also vary, for example nitrate fertilizers are more susceptible to leaching than urea and ammonium compounds (Ritter, 1989). According to Mills (1985; cited in Tredoux, 1993) the use of certain pesticides can also inhibit denitrification resulting in a build-up of nitrate in the soil and groundwater.

The findings of the Gamtoos study are similar to those of Sonnen *et al.*, (1987) in that they found no general trends or relationships in nitrate concentration and irrigation applications, rainfall, RWLs or borehole depths. Whilst Schmidt and Sherman (1987) and Ritter *et al.*, (1990) found the total loading of nitrogen to be the most dominant factor influencing the quantity of nitrate in the soil profile and available for leaching. Denver (1986, cited in Ritter *et al.*, 1989) found the highest nitrate concentrations in groundwater in west-central and south-west Delaware occurred in soils which were well drained. Sonnen *et al.*, (1987) attributed a portion to the presence of shales, which contain nitrate and exchangeable ammonium ions. Nitrate-N values in groundwater in the Gamtoos study ranged from 0.1 $mg.l^{-1}$ to a mean of 27.1 $mg.l^{-1}$ in borehole M2. The elevated nitrate and nitrite levels in groundwater at site M can be attributed to elevated loading (exact quantities are unknown) of nitrogen-containing fertilizers in the fields. Conditions which most favour the leaching of nitrate from the soil are frequent changes in crop type, interspersed with lying fallow with crop residues left in the fields as occurred in the Gamtoos study area at the sites M, D and T. Studies in California (Ritter, 1989) indicate that the nitrate levels in groundwater in normal irrigated croplands will range from 25 to 30 $mg.l^{-1}$ (nitrate as N). Weaver (1993) found that nitrate (as N) values in shallow groundwater in the Hex River valley irrigation area ranged from 0.2 to 33.2 $mg.l^{-1}$.

Phosphorus

Generally natural groundwater contains less than 0.1 $mg.l^{-1}$ phosphorus (Bouwer, 1978). This is attributed to the low solubility of apatite which is a natural source of phosphorus to groundwater. The phosphorus content of groundwater is determined in part by soil chemistry and the rate of leaching. Phosphate fixation, particularly in clay soils, occurs on moving through the soil profile (Matthess, 1982; Nicholaichuk *et al.*, 1991). Table 5.8 gives the total phosphorus ($mg.l^{-1}$) content of groundwater in those boreholes which exceeded a mean of 1 $mg.l^{-1}$. Of the boreholes having a mean below 1 $mg.l^{-1}$, elevated values were observed on occasions, these include borehole T (2.46 $mg.l^{-1}$ on 15 January 1993), D4 (2.73 $mg.l^{-1}$ on 1 April 1993), D5 (3.89 $mg.l^{-1}$ on 1 April 1993), V3 (2.57 $mg.l^{-1}$ on 6 April 1993), V4 (3.11 $mg.l^{-1}$ on 6 April 1993) and V5 (5.53 $mg.l^{-1}$ on 6 April 1993 and 3.19 $mg.l^{-1}$ on 2 February 1994). These results show that phosphorus containing compounds were applied in the area between boreholes D4 and D5 prior to 1 April 1993 and between boreholes V3, V4 and V5 prior to 6 April 1993.

Groundwater values in the Gamtoos floodplain were somewhat elevated in comparison to the highest value of 0.11 $mg.l^{-1}$ and consistent values below 0.05 $mg.l^{-1}$ in the Hex River valley irrigation area (Weaver, 1993).

Borehole	**Mean Total Phosphorus (as P) ($mg.l^{-1}$)**
D1	1.22
M2	1.35
M3	2.35
S1	2.88
S2	2.49
S3	2.00

Table 5.8 Mean total phosphorus (as P) concentration of groundwater in boreholes which exceeded a mean of 1 $mg.l^{-1}$.

6. DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

Although some of the findings of this study have already been discussed in chapters 4 and 5, the main findings are summarised in this chapter and are placed in the context of the original study objective. The original study objective, stated in chapter 1, was to examine the influence of agricultural practices in a small section of the floodplain on the estuary and examine the land-based inputs to the estuary via surface runoff or subsurface throughflow. The specific aims are restated from chapter 1 and discussed individually. The extent to which the aims have been achieved, as well as shortcomings are discussed. The implications that these findings have for the management of the Gamtoos estuary are briefly discussed in section 6.4. A summary of conclusions and recommendations for future research then concludes the chapter.

Before the results are discussed they must be placed in the context of the conditions which existed during the measurement period, that is variable rainfall. The initial measurements of November/ December 1992 followed a particularly dry period during which farmers were on reduced irrigation quotas. The mean annual rainfall in 1992 was over 25% below the twenty year running mean for Hankey. The main deviation in rainfall occurred over a seven month period from March to September, when rainfall was 34% below the twenty year running mean for the same months. During these dry months farmers were on 15% of their normal irrigation quotas, but this was increased to 25% following the 1 September review. The study then entered a period of above average rainfall in 1993 (almost 52% above the long-term mean). Also during this time on 12 June, a record maximum quantity of rain fell within a 24 hour period (in comparison to the 102 years and 61 years of data available for Hankey and Jeffreys Bay respectively). The rainfall of 1994, although above average, was closer to the long-term mean. Although the findings are discussed in the context of the variable rainfall, applicability to other rainfall conditions is discussed in section 6.4.

6.1 Discussion of the aim to: Investigate the Groundwater Dynamics and Subsurface Drainage Patterns in Relation to Geological Structures and the Effects of Rain, Irrigation Water and Tidal Variations in the Adjacent Estuary, as well as aspects such as Vegetation Cover, Wind and Solar Radiation (Temperature).

The investigation of groundwater dynamics in this study refers to shallow (< 6 m) groundwater. The groundwater investigation comprised five sites across a small section of the floodplain. The results presented in chapter 5 and the discussion below are based on data obtained from the twenty boreholes established at the five sites. The boreholes tapped predominantly clay material in the alluvial floodplain.

6.1.1 Water Table Response to Rain, Irrigation and Solar Radiation.

When examining water table response to surface inputs (rainfall and irrigation) James and Fenton (1993) found that a thirty day response time, following rainfall, occurred where water tables were at a depth of between 130 and 340 cm below the soil surface in a silt clay loam in Iowa, USA. Furthermore Capone and Slater (1990) state that in Suffolk County, New York, the water table response to changes in rainfall can be detected after about six months. They noted however that in certain areas a more immediate response occurred, with the time delay varying as a function of depth of the saturated zone, hydraulic conductivity and soil porosity. With regard to water losses to the groundwater zone, Chiew and MacMahon (1991) found that in the irrigation area of Riverine Plain, south-eastern Australia, only 15% of irrigation water and 6% of rainfall recharged the shallow aquifer with the remainder being lost to plant consumption and evaporation.

Solar radiation will determine air temperature and potential evapotranspiration, and wind will also influence evaporation losses. Evaporation losses (calculated from Tables given in Midgley *et al.*, 1994a; 1994b) are high in the Gamtoos valley, resulting in periods of moisture deficit, and during this time a proportional decrease in groundwater recharge will occur. Figure 6.1 shows the total irrigation input (on full irrigation quotas), on a monthly basis throughout the Gamtoos irrigation area (Joubert, 1992) and the mean monthly rainfall based on the long-term means (given in Table

4.3). Based on these general conditions, a moisture deficit will occur from October to February (Figure 6.1). This moisture deficit could account for seasonal differences in the elevation of the water table.

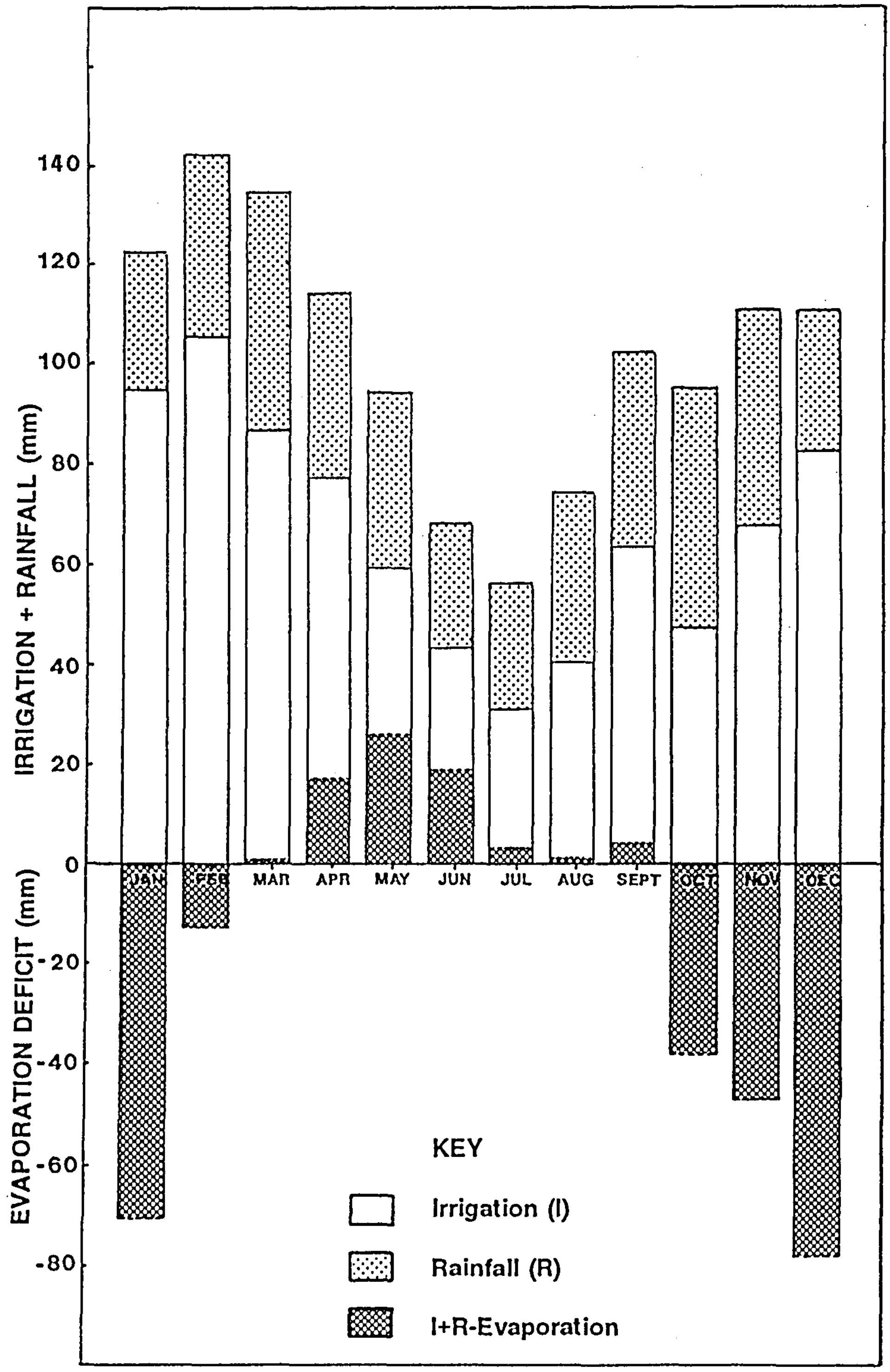

Figure 6.1 Monthly irrigation input on full quotas, average rainfall and evaporative losses in the Gamtoos region.

Figures 5.6, 5.8, 5.10 and 5.11 show that the water table level throughout the study area was at its lowest at the time of borehole installation (i.e. during August to November 1992). This can be attributed to the below average rainfall during 1992 (Table 4.3) and the reduced irrigation quotas during this time (section 2.2.6). At no time during the subsequent measurement period did the water table drop to the low levels of 1992 (borehole installation). As mentioned earlier, the study then entered a period of above average rainfall during 1993 and 1994, and the exceptionally heavy rainfall of June 1993 was sufficient to mask any possible seasonal trend in water table level which may have been related to the water balance of inputs minus evaporative losses. Although a delayed response in the water table level generally followed rainfall in the Gamtoos floodplain, Figures 5.6, 5.8, 5.10 and 5.11 show an immediate increase in the level of the water table after the June rainfall. Most of the boreholes exhibited their highest recorded water levels (during this study) within four days of the June rainfall. During the latter six months of 1993, the water table throughout the study area was generally higher than that prior to the June 1993 rainfall. By early 1994, however the water table was starting to show a slight decline, and this trend continued in most boreholes until measurements ceased in April 1994. Although all measurements essentially stopped in April 1994, a continuous recording data logger on borehole D5 yielded data up to 15 June 1994 (Figure 6.2). Figure 6.2 not only shows the fluctuation in the level of the water table with corresponding rainfall quantities, but also shows a general increase in the water table level from 12 April to 15 June. Although there is no supporting data available from the other boreholes, the increase in water table shown in Figure 6.2 may be a result of lower evaporative losses as depicted in Figure 6.1.

In the Gamtoos study area, irrigation varied both spatially and temporally over small areas (0.1 ha). In certain situations a rise in RWL corresponded with irrigation and on other occasions did not. The hydraulic gradient between most of the boreholes was very slight. With the constantly changing surface inputs coupled with variable recharge rates, groundwater in the area exhibited highly dynamic and variable subsurface flow directions (Figure 5.5). The translatory movement of water through the soil profile is one of many factors responsible for a fluctuation of RWL in response to surface inputs (Deverel and Fio, 1991). This is further affected by antecedent moisture conditions and hydraulic conductivity. Soil textures (Figure 5.2), previous flood paths, clay lenses and shelly zones differ at each borehole site and thus each borehole will respond differently. The hydraulic gradient between the estuary and boreholes at close proximity to the estuary was greater and groundwater in such areas showed a preferential drainage towards the estuary.

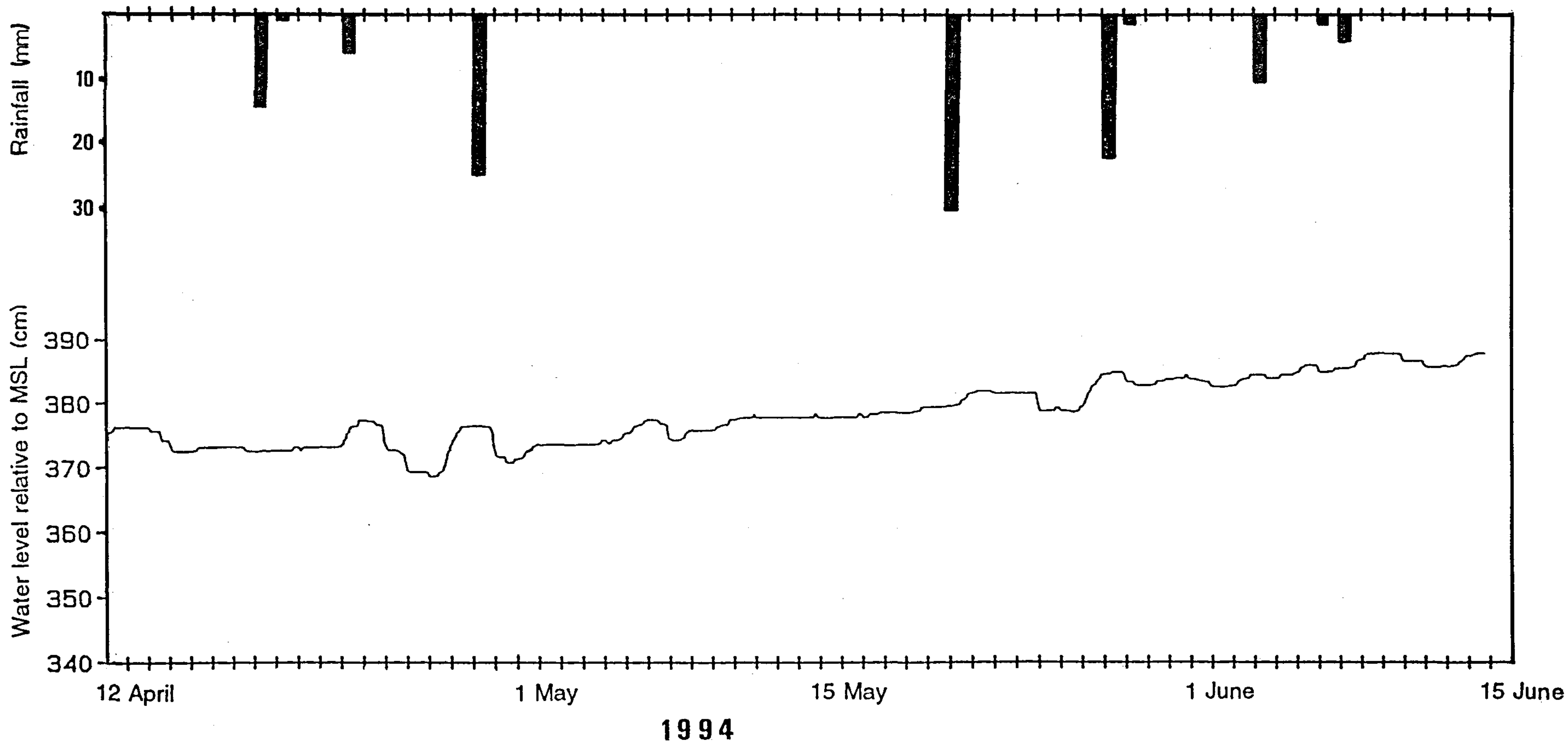

Figure 6.2 Rainfall quantities, days on which rainfall was recorded and the corresponding variation in height of the water table at borehole D5, logged on an hourly basis from 12 April 1993 to 15 June 1993.

The vegetation cover at sites S and V consisted mostly of fodder crops, whereas at sites M, D and T it was market vegetables. Differences in vegetation cover will determine the type, timing and quantity of fertilizer applied, and to a lesser extent the timing and quantity of irrigation. In this study, no trends or differences related to differing vegetation type were identified, and thus does not warrant any further discussion. Aspects other than vegetation cover (soil properties, hydraulic conductivity, area of irrigation, hydraulic gradient) appeared to have had a greater influence on groundwater characteristics.

Other Factors Influencing RWLs

An immediate increase in RWL during sizeable rainfall or irrigation inputs can occur due to air entrapment. An inverted zone of saturation is created at the land surface, and the advancing wet front traps air between itself and the water table thereby causing a sudden elevation in the RWL. This phenomenon is typically observed in shallow unconfined aquifers. Once the trapped air escapes the rise in water table then dissipates (Freeze and Cherry, 1979). Atmospheric pressure effects can also account for fluctuations in RWLs (Freeze and Cherry, 1979), however this was not observed during the study period as some boreholes showed an increase in RWL whilst neighbouring boreholes exhibited a drop in RWL during successive measurements on the same day. If the change in RWL were due to an increase or decrease in air pressure - all boreholes would exhibit a corresponding drop or rise in RWL accordingly.

6.1.2 Water Table Response to Tidal Fluctuations

During tidal fluctuations in surface water bodies, pressure waves propagate inland through adjacent aquifers causing hydraulic gradients and groundwater levels to fluctuate (Serfes, 1991). The characteristics of the pressure waves are a function of the tidal period, tidal amplitude and aquifer transmissivity. Erskine (1991) examined the effect of tidal fluctuation on a coastal aquifer in East Anglia, UK and found that the time lag of the water table response increased linearly with the distance of the monitoring boreholes from the sea. Tidal efficiency decreased exponentially with distance from the sea. Furthermore Erskine states that in unconfined aquifers, damping extinguished fluctuations about 400 m from the sea, whereas Nielsen (1990) documented considerable damping over a distance of 10 m at a beach north of Sydney, Australia.

In the Gamtoos study area no correlation was found between water table level and tidal fluctuation. The absence of any tidal effect in the study area could be attributed to the combined effects of low tidal amplitude and period and distance of boreholes from the estuary. The tidal lag time in the vicinity of station 9 (near sites M and D) is 2 hours and 27 minutes (section 4.5), and at this point the tidal amplitude is somewhat attenuated being 55 cm at spring tide and 25 cm during neap tide. Furthermore aquifer transmissivity throughout the floodplain is low ranging from 0.03 m per day (borehole V2) to 0.93 m per day (borehole D6). A low hydraulic conductivity of sediments hinders the transmission of pressure waves through the aquifer, and any pressure effects would be further reduced in shallow unconfined aquifers. Figure 6.3 shows the variation in the RWL in boreholes at sites M and D during hourly monitoring on 25 November 1992. Hourly monitoring, daily readings and continuous RWL data showed no relationship between the water table and tidal fluctuations.

Tidal fluctuations will, however, influence the hydraulic gradient between the water table and the estuary. With an increase in the hydraulic gradient there will be a proportional increase in groundwater discharge to the estuary. During the study period the level of the water table in the floodplain was generally at an elevation of between 2 to 4 m above the water level in the estuary. An exception occurred at site V where the water table fluctuated between 1 to 3 m above mean sea level. This would thus favour a continual preferential groundwater discharge into the estuary during both high and low tide, with the proportional flux varying with tide. Staver and Brinsfield (1996) report that the groundwater discharge into the Wye River estuary varied widely in the short-term as a result of changes in hydraulic gradient caused by tidal fluctuations. In the long-term, seasonal changes in groundwater recharge rates (i.e. an elevated water table during the rainy season) determined groundwater discharge.

6.2 Discussion of the aim to: Estimate the Input of Fertilizers, Herbicides and Pesticides on the Agricultural Lands, and Assess the Resultant Input of Chemical Products into the Estuary.

An estimate of the input of fertilizers, herbicides and pesticides on the agricultural lands could not be determined. The farmers were not forthcoming with such information for fear of reprisals. Regardless of the input of such compounds on the land, of greater importance is whether such compounds enter the estuary in harmful amounts. Inputs to the estuary from land-based activities include surface runoff, discharge from the surface agricultural system (discussed in section 4.2 and 4.6.4), discharge from the subsurface drainage system (discussed in section 5.5), and groundwater inputs to the estuary (discussed in sections 5.3 and 5.5). These inputs have been discussed at length in the designated sections. Certain aspects of the findings are however briefly repeated in the discussion to follow, whilst the fate of such compounds in the estuary is discussed in section 6.3.

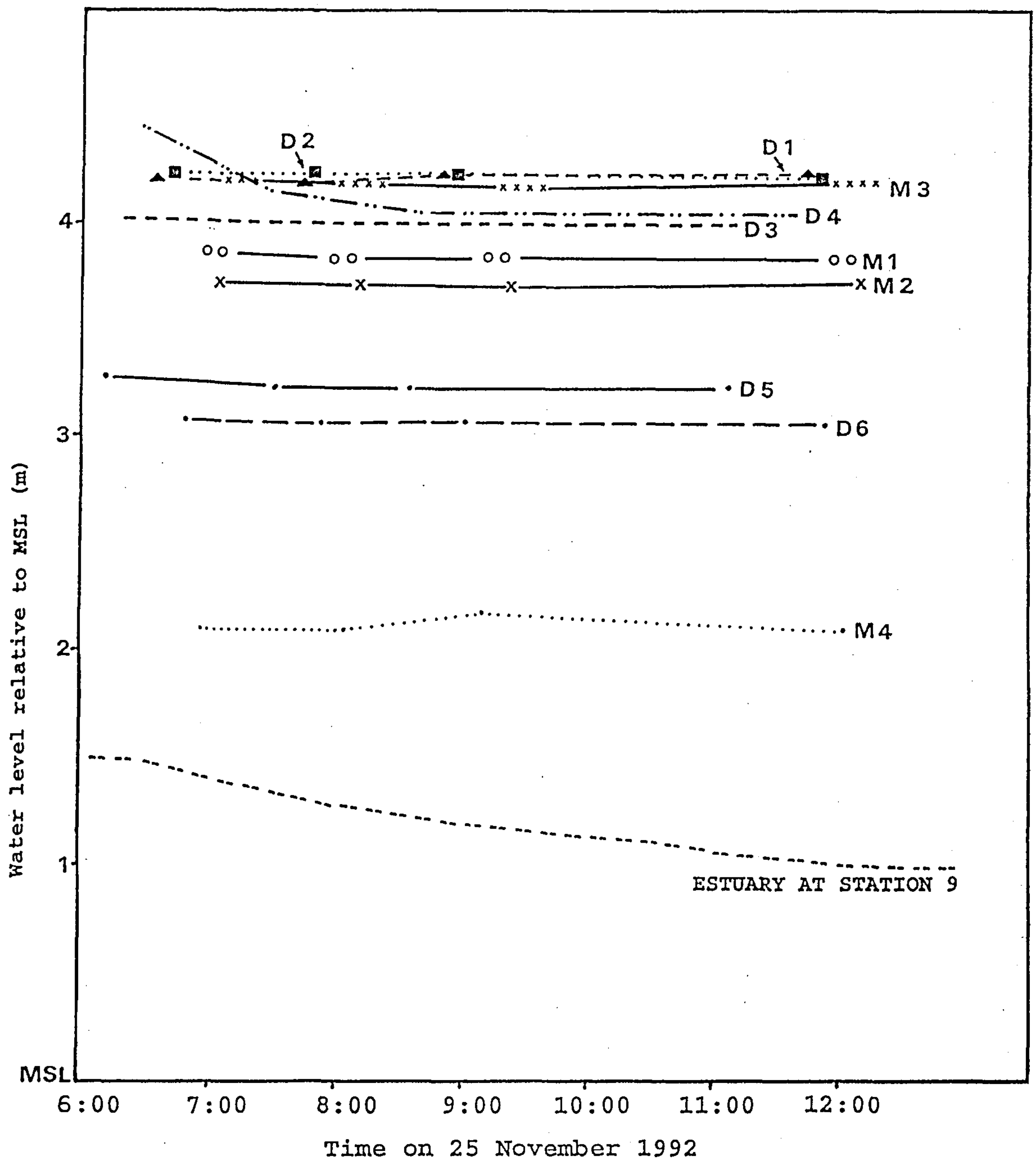

Figure 6.3 Variation in RWL in boreholes at sites M and D during hourly monitoring on 25 November 1992. The change in water level at estuary station 9 is also shown.

6.2.1 Inputs into the Estuary from the Surface Agricultural Drainage System

Surface runoff from the fields will provide an input to the estuary. From the results presented in chapter 4 it is seen that discharge from the surface agricultural drainage system constitutes a clearly identifiable point source of pollution to the Gamtoos estuary. The discharge from the drainage pipe comprises runoff from an area of approximately 50 ha and throughflow from the upper 1.2 m of soil. Analysis of water samples obtained at the drainage pipe give an indication of the chemical inputs to the estuary. Results show that the concentrations of nitrate-N, sulphate and potassium in the drainage water were frequently present in very high levels. Similarly the nitrite-N content often exceeded the recommended limit of 0.06 $mg.l^{-1}$, and all of the samples obtained at the drainage pipe exceeded the recommended total phosphorus limit for the protection of aquatic life. Conservative estimates of the nutrient loading to the estuary were obtained. These estimates do not include the flush of nutrients into the estuary during rainfall. On an annual basis approximately 75 kg of phosphorus, 129 kg of nitrate and 16 kg of nitrite entered the estuary from the surface agricultural drainage system alone. In addition to these nutrient inputs to the estuary, the pesticide screening test yielded positive results in all of the surface agricultural drainage samples analysed. A mixing zone exists around the discharge point of the surface agricultural drainage pipe resulting in a dilution of the pollutants entering the estuary. Such dilution increases with distance from the discharge point and thus brought the concentration of nutrients measured at certain times in the estuary to within recommended limits.

6.2.2 Variation in Groundwater Quality

No seasonal trends in groundwater quality were observed. Groundwater at borehole S4 did however, show a gradual deterioration in quality during the study period as a result of the installation of an adjacent drainage trench (as discussed in section 5.3.3). Apart from this, temporal variations in groundwater quality were mostly determined by land-surface application of fertilizers. For example, on 15 May 1993, 14 June and 29 July 1993, groundwater in all of the boreholes at site M exhibited elevated nitrate levels. In April 1993, elevated total phosphorus values were found in groundwater at boreholes V2, V3, D1, D4, S2 and S3. Boreholes S2 and S3 also had elevated total phosphorus values on 15 January 1993. Elevated nitrite levels occurred in groundwater at boreholes V1, V4 and V5 in April 1993 and throughout site V as well as D2 and D4 in June 1993. A seepage into the estuary near station 8, showed similarities to groundwater abstracted from nearby boreholes D1, D2, D3 and D6. Similarly the discharge from a subsurface drainage system showed corresponding characteristics to that of groundwater abstracted from boreholes within the drainage system bounds as well as neighbouring fields. Apart from these similarities groundwater at the five sites generally exhibited some differences.

6.2.3 The Nutrient Input into the Gamtoos Estuary

The input of nutrients to the Gamtoos estuary will be determined not only by groundwater quality in the surrounding area, but also according to the total groundwater discharge to the estuary. As discussed in section 5.3.5, groundwater discharge to the estuary will vary with changes in the hydraulic gradient as well as with changes in hydraulic conductivity.

Millham and Howes (1994) examined the rate and pattern of groundwater discharge into Little Pond, a shallow coastal embayment on the southern shore of Cape Cod, Massachusetts. The tidal inlet to Little Pond was restricted by the presence of a sill as a result of the development of a flood-tidal delta, this raised the embayment level thereby reducing the tidal range. Groundwater discharge to the embayment was dependent on the hydraulic gradient, with an increase in groundwater discharge when the water level within the embayment was low. As the nutrient levels in the groundwater were high an increased groundwater input resulted in increased loading of the embayment. Millham and Howes found a complex management situation whereby when the lagoon was opened to allow tidal flushing, this accelerated nutrient removal from the embayment, but the resultant lowered water level led to an increase in the nutrient input via groundwater discharge.

Contaminant concentration in groundwater will determine the resultant input to coastal marine ecosystems. Thus, an indication of the input of nutrients to the Gamtoos estuary, via groundwater flow is obtained by examining the nutrient content of groundwater abstracted from the boreholes, as well as the discharge from the subsurface drainage system and seepage. As groundwater quality varies throughout the area so too the proportion of nutrients entering the estuary via irrigation return flow will vary spatially and temporally. Groundwater at site M for example, had a higher nutrient content than groundwater at sites W, D, S and T and it therefore appears as if irrigation return flow in the vicinity of site M would provide a proportionally greater nutrient input to the estuary.

An indication of the magnitude of nutrient input to the estuary is obtained by multiplying the estimated groundwater discharge to the estuary with the mean nutrient concentration in the groundwater. The groundwater discharge to the estuary (section 5.3.5) from a 7.7 km expanse in the Loerie Flats was estimated at 276 m^3 per day. The mean nitrate (as N) concentration of groundwater at all the boreholes at site D, throughout the study period, was $\leq$ 0.1 $mg.l^{-1}$. Assuming a groundwater concentration of 0.1 $mg.l^{-1}$ nitrate-N, with a groundwater discharge of 276 m^3 per day, then 10 $kg.a^{-1}$ of nitrate-N would enter the estuary from this area. However, at site M where the nutrient content of groundwater was greater, applying the mean nitrate-N concentration of groundwater from borehole M3 (27.08 $mg.l^{-1}$) would result in an input of 2.7 tons of nitrate on an annual basis.

In a similar manner, by multiplying the lowest and highest concentrations of nitrite (as N) and total phosphorus (as P) obtained in groundwater within the Loerie Flats, by the average daily groundwater discharge to the estuary would yield an annual input ranging from 1.2 to 89 kg of nitrite-N to the estuary. The annual input of phosphorus to the estuary would be between 10 and 400 kg. These are conservative estimates and it must be borne in mind that the nutrient loads given here represent the input from an area which comprises less than 20% of the total area bordering the estuary. With regard to pesticides, discussed in section 4.8, findings are based on a small data set in which no organophosphate and carbamate pesticides were detected in groundwater, or the subsurface drainage system or the estuary. Being a small data set however, this aspect requires further investigation (see section 6.6).

6.3 Discussion of the aim to: Investigate Estuarine Processes such as Tidal Action, Water Stratification and the Influence of Freshwater Input, Particularly in terms of Mixing and Removal of Chemical Products, and Possible Effects on Water Quality and Biota.

The physical properties and structures within the estuary over tidal cycles and in response to freshwater inputs are described and discussed in detail in section 4.6, and are not repeated here, however, a brief summary of the main findings follows. The effects of estuarine processes and inputs to the estuary on biota are covered in the form of a literature review in section 4.10. The discussion to follow focuses on three aspects:

- Estuarine stratification and tidal action (section 6.3.1)
- The effect of inputs into the estuary (6.3.2), and
- The fate of the inputs into the estuary (6.3.3).

6.3.1 Estuarine Stratification and Tidal Action

Distinct differences in salinity stratification were exhibited between the lower and upper estuary, and in the response to rainfall. Stratified conditions favour estuarine circulation resulting in mixing, with subsequent flushing of the estuary. The aggraded section upstream of station 7 limited the extent of the tidal influence since the more dense saline water remains at the bottom of the water column. Following rain, the resultant increase in river flow meant that the physical structure of the upper estuary, apart from during spring high tide, was dominated by riverine processes with tidal influences limited to the lower estuary. During these periods the established longitudinal salinity gradient was greater than the vertical salinity gradient, and such conditions usually hinder estuarine circulation and increase residence times. Tidally-induced mixing during spring high tide was however sufficient to mix the increased freshwater flow with the saline bottom waters with resultant flushing from the estuary. During the extremely high rainfall of June 1993 the rainfall was such that freshwater flow dominated throughout the estuary. Saline conditions however persisted at the bottom of the water column near the mouth due to the presence of flood-tidal deltas.

6.3.2 The Effect of Inputs Into the Estuary

According to Petts (1984, p. 253) '*whilst short durations of exposure to low-quality water may prove tolerable, long-duration exposure of the same magnitude can result in deleterious effects*'. In the Gamtoos estuary, no long-term periods of low quality water were observed, however, short-term periods of low quality water did occur and these are discussed. There is the potential for these periods of low quality water to persist, and the implications of longer-term low quality periods is discussed in section 6.4.

Despite the elevated nutrient input to the estuary via irrigation return flow, the nutrient content of the estuary was routinely low because of sufficient dilution. Furthermore nitrates entering the estuary via groundwater discharge are subject to volatilization and denitrification within the sediments prior to entering the estuary (Slater and Capone, 1987), and sedimentation of nitrate-N may also occur (Stanley, 1993). Elevated values of nitrate-N were however found in the Gamtoos estuary on three occasions, these being on 15 June 1993 at station 5 where the nitrate-N content was 0.26 $mg.l^{-1}$; on 26 August 1993 values of 0.26 $mg.l^{-1}$ (station 5), 0.84 $mg.l^{-1}$ at station 6 and 0.22 $mg.l^{-1}$ at station 9 were recorded. On 12 October a nitrate-N value of 0.30 mg l^{-1} was recorded at station 9, this elevated value could have been caused by the high nitrate input of 12.14 $mg.l^{-1}$ discharging from the agricultural drainage pipe upstream of station 9. Although no supporting data are available, it could be speculated that the subsurface agricultural drain discharging into the estuary just upstream of station 6 could have accounted for the elevated nitrate values at station 6. The mean total phosphorus concentrations throughout the estuary (which are given in Table 4.7) show that the phosphate concentrations in the estuary were routinely high, with extreme values being recorded on some days. The elevated values were found more frequently in the upper estuary than in the lower reaches.

As water is not released into the Gamtoos estuary from the Kouga dam, a freshwater input to the estuary will occur either as a result of inputs from the Groot River or following rain. The quality of water which enters the Gamtoos River from the Groot River is variable depending on its origin (see section 4.4.2). During rainfall organic debris and other oxygen-demanding material were washed into the estuary. Furthermore, interaction of rainfall with oxygen-demanding substances on the land surface will render a low oxygen input (irrigation return flow) to the estuary. Such inputs to the estuary, together with the release of oxygen-demanding matter from estuarine sediments, resulted in increased turbidity and a lowering of the oxygen content of estuarine waters. The hypoxic conditions and in certain instances anoxic conditions tended to be more frequently observed in the bottom waters of the upper estuary, regardless of tide. Localised, deeper scour holes served as sinks for decomposing organic matter.

6.3.3 The Fate of Inputs into the Estuary

The fate of the inputs (discussed in section 6.3.2) to the estuary is determined by estuarine processes. Geomorphological characteristics of the estuary such as the aggraded section upstream of station 7 and the constriction at the tidal inlet further influence estuarine functioning. Increasing sedimentation at the tidal inlet over the past four years has resulted in reduced ocean-estuary exchanges, with ocean tides attenuated by 50 to 60% of that in the adjacent sea. These factors account for the substantial differences that were found in the salinity structures throughout the estuary during spring and neap tides. The lower reaches of the estuary were mostly well mixed during spring high tides by the input of sea water through the tidal inlet, with flushing of the estuary during low tide. Wind was generally not an important mixing agent (section 4.9) because of the narrow estuary channel, though in certain sections orientated parallel to the wind reasonable waves could be generated at times.

Because the average freshwater input at the head of the estuary is low (estimated at less than 1 $m^3.s^{-1}$) the estuarine structures are generally tide-dominated with periods of change following rainfall. Inflows in excess of 1 $m^3.s^{-1}$ effectively flushed out much of the upper reaches of the estuary, but in the lower 14 km the cross-section broadens and deepens, and the freshwater tended to exit in the surface layers. As a result of the low freshwater inflow to the estuary and because of the presence of sand banks in the upper estuary, most of the problem conditions occurred upstream of station 7. Because tidal flushing was limited in the upper estuary, pollutants discharging into this region such as occurs at the agricultural drainage pipe remained near the surface, being mixed further into the water column as tidal mixing became more effective further downstream. Freshwater inflow is thus necessary to remove pollutants further downstream. A volume inflow of 10 $m^3.s^{-1}$ would be sufficient to flush out the upper reaches within a tidal period. Pollutants would thus be removed to a position where tidal transport could then abstract them into the ocean.

Although elevated nutrient levels were measured in the estuary from time to time, the freshwater and marine input to the estuary and the tidal mixing were adequate for the dilution of the quantities and concentrations entering the estuary. Similarly hypoxic conditions did not persist for any length of time in the estuary. Despite the physical restrictions to the movement of the marine waters and a lower rate of mixing and replenishment in the upper estuary, movement of the main oceanic tidal wedge, particularly during spring high tides, was sufficient for the replenishment of DO. In the event of prolonged dry conditions however, with decreased freshwater flow, or with increased constriction at the mouth, a nutrient build-up could occur in the upper estuary.

6.4 Implications for Management of the Gamtoos Estuary

The study has provided some insight into the changes which occurred in the Gamtoos estuary in response to irrigation return flow and freshwater inputs. These findings have management implications for the Gamtoos estuary and possibly other estuaries in the Eastern Cape and are discussed in this section. Firstly, whilst this study did not entail a detailed examination of sedimentation at the estuary mouth, Figure 2.3 shows the substantial constriction at the inlet by flood-tidal deltas in August 1992. Furthermore, a number of studies (Institute for Coastal Research, unpublished data; Reddering and Esterhuizen, 1984; Reddering and Rust, 1990) indicate that tidal exchanges through the inlet have decreased considerably over the last three to four years. Reduced tidal exchanges will result in reduced mixing and flushing of nutrients and pesticides from the estuary. Even during the record rainfall of June 1993 (see section 4.4.1), the whole estuary was not flushed out, and the mouth was not scoured open. This can be attributed to the presence of the Kouga dam higher in the catchment, which absorbed a portion of the runoff which would have been available for the flushing of the estuary. Prior to the heavy June rainfall the dam was 66% full, a week later it had risen to 80% and it was only after the good rains in September (total: 143.7 mm) that overtopping of the dam occurred.

With the decrease in freshwater input, and if the tidal inlet closes, given the nutrient input there is a strong possibility of eutrophication of the Gamtoos estuary. An accumulation of nutrients could result in a situation similar to that of the Peel-Harvey estuarine system of Western Australia, where the input of phosphorus-rich irrigation return flow led to eutrophication problems within the estuary (Birch *et al.*, 1984). The Harvey estuary is shallow (~2m deep) and has a strongly seasonal rainfall and river flow.

Because of the seasonal (winter) freshwater input the salinity range is extreme, ranging from freshwater to 50. Insufficient marine flushing exacerbated the problem leading to a build-up of phosphorus stores both in the water and estuarine sediments, which caused massive blooms of the blue-green alga *Nodularia*.

The second factor to be commented on is the application of Jezewski and Roberts' (1986) freshwater requirements of the Gamtoos estuary. Jezewski and Roberts clearly state that their values (section 4.4.3) are mere estimates based on a simplistic model. The comment to follow is not a criticism of their findings, which served as an important reference basis, but an application of their values to the conditions found in the Gamtoos estuary during this study. Jezewski and Roberts state that 36.8×10^6 $m^3.a^{-1}$ is required to keep the tidal inlet open and flush accumulated sediment out of the estuary. A consistent flow of 1.16 $m^3.s^{-1}$ throughout the year would produce this total. The estimated flow of the estuary is around this value. However as mentioned in the preceding discussion, tidal exchanges have decreased over the last few years as a result of increasing constriction at the inlet (see section 4.5) and a sustained low flow in the estuary, as occurred during 1992, would not be sufficient to prevent sedimentation. A short period of strong freshwater flow to the mouth is deemed necessary. During dry years dam levels are usually also low and with water being scarce it is understandable that sizeable volumes cannot simply be released from the dam. Nevertheless, it must be ensured that the tidal inlet remains sufficiently open, even if it has to be done mechanically, to allow ocean-estuary exchanges.

Not only does the flow of freshwater to the sea prevent the build-up of hypersaline conditions within the estuary, but it acts as a cue for certain marine organisms to enter the estuary (recruitment). Likewise certain estuarine organisms must have a marine phase to complete their full development cycle (Wooldridge, 1994). Temporary and partial closure of the mouth would thus also inhibit such cues and this would result in a reduction in biotic diversity and a decline in the marine species within the estuary (Reddering, 1988; Whitfield and Bruton, 1989). Thus, not only is a flushing event required to keep the tidal inlet sufficiently open, be it a release from the dam or manual opening of the mouth, but the timing of such an event should be best suited to the biological requirements of the estuary.

Thirdly from a management option, although it would be difficult to reduce the pollutant inputs to the land, nor could irrigation return flow to the estuary easily be avoided, any future drainage systems should discharge as far downstream in the estuary as possible. This would prevent a build-up of nutrients in the upper reaches of the estuary where a lower rate of mixing and flushing occurs.

The general results presented here should also be applicable to other estuaries where intensive agriculture is practised in the floodplain with a concomitant input of fertilizers and pesticides. However, it is important to realise the differences that do exist, and each system must be considered on its own merits.

6.5 Conclusions

(1) The Gamtoos estuary, an impounded system, experiences a low and variable rainfall and the mean freshwater flow is estimated at $< 1\ m^3.s^{-1}$. Furthermore the estuary receives substantial agricultural inputs via irrigation return flow during both dry and wet conditions.

(2) The estuary extends 20 km inland of the tidal inlet and the constriction of the mouth means that ocean-estuary exchanges are restricted. There is an aggraded section in upper estuary which further limits the extent to which the saltwater wedge can move upstream, so that whereas substantial mixing occurs in the lower estuary it occurs to a lesser extent in the upper estuary.

(3) Despite the agricultural inputs and temporal periods of hypoxic conditions, nuisance macrophytes and high turbidity in the aquatic environment, the Gamtoos estuary has shown a degree of resilience and water quality is apparently restored after a few days. It is therefore important that the tidal inlet remains open to allow tidal flushing, particularly given the variable rainfall and lack of freshwater release from the Kouga dam.

6.6 Recommendations for Future Research

(1) The use of a pesticide screening method in this study was advantageous from the point of view that costly pesticide analyses were avoided. The screening test used here has shown some positive results, and according to Van der Walt (1996) has the advantage that it is a very sensitive test, yet does not yield any false negative results. Since positive results were found during this study, it is recommended that further research be done to examine the exact quantities and types of pesticides entering the Gamtoos estuary. Furthermore the positive results indicate that the expense of such an investigation would be justified.

In addition to yielding information on the input of pesticides to the estuary, the analysis of a spectrum of pesticides following positive screening results would also highlight whether future pesticide studies country-wide should in fact first opt for the less expensive screening approach. This would allow for the identification of problem areas, thereby making more effective use of scarce research funds.

(2) Following on from this Gamtoos study, it would be desirous to model the system in order to provide a predictive capacity of estuarine behaviour, for the purpose of effective management. Before this could be done a basic practical problem should be addressed, that is the need for a gauging station on the Gamtoos River. At present there is no gauging station at any point on the 75 km extent of the river, and the lack of flow data was a limiting factor in this study. In order to model the Gamtoos estuary, knowledge of freshwater inputs to the system is required. An ideal position for such a gauging station would be at the Boschhoek railway crossing, as it marks the approximate position of the freshwater input to the estuary (the uppermost extent of the tidal influence is just downstream of this point). It is also suitable because the river flows in a straight path for a fair distance, which is a prerequisite for a gauging station, and there is road access to the site.

The Gamtoos floodplain and the estuary are both complex systems which would not be uncomplicated to model. Wuttke *et al.*, (1991) overcame some of the complexities of their model by splitting it into several submodels. Before a system is modelled, the question should be asked: What is the primary requirement from this model? A vast range of models with various applications have been reported for: the prediction of the flushing capacity (Merino *et al.*, 1990), tidal circulation within an estuary (Wang and Craig, 1993), estuarine stratification (Sharples *et al.*, 1994), residence times (Miller and McPherson, 1991), salinity distribution response to freshwater inputs (Cifuentes *et al.*, 1990; Thollapalli, 1991), nutrient cycling within an estuary (Kimmerer *et al.*, 1993), or with biological implications (Gupta *et al.*, 1994; Hopkinson and Vallino, 1995) or the potential for eutrophication (Madden and Kemp, 1996). Similarly, a vast range of groundwater modelling studies have been reported for: the time lag between surface application of fertilizers and resultant leachate inputs to groundwater (Kim *et al.*, 1993), groundwater vulnerability to contamination by pesticides (Villeneuve *et al.*, 1990), groundwater discharge into an adjacent water body (Nuttle and Harvey, 1995) or the nutrient transport into an estuary via groundwater discharge (Millham and Howes, 1994).

Future modelling of the Gamtoos estuary would require a model that can predict the residence time of nutrients and the potential that exists for eutrophication should sustained dry periods prevail or the tidal inlet close. Seemingly suitable models, which are reported as successful in estuarine studies having certain elements in common with the Gamtoos system, cannot simply be applied without some modification to suit local conditions (Kimmerer *et al.*, 1993). Furthermore certain models only provide a good correlation with field data under specific conditions (Giese and Jay, 1989), for example the model used by Merino *et al.*, (1990) assumes that rainfall is the main force determining estuarine salinity and flushing, when field conditions changed to a situation where rainfall was not a dominant factor the model proved inappropriate for the system.

Quantification of the nutrient inputs to the estuary would be required and thus a dual modelling system could be used, one to model nutrient loading of the estuary via groundwater discharge and a second to model estuarine hydrodynamics. The groundwater discharge and related nutrient input to the estuary could be predicted using a model similar to that used by Millham and Howes (1994). The model was based on estimations of hydraulic conductivity and a measured hydraulic gradient. The type of data required by the model could be obtained in the Gamtoos area, although the findings of Millham and Howes were based on a vast collection of data. Nuttle and Harvey (1995) however state that it is essentially difficult to accurately determine hydraulic conductivity, and where an estimated hydraulic conductivity is low it also exhibits some degree of spatial variability. They further argue that models based on estimations of hydraulic conductivity and a measured hydraulic gradient are less accurate than their model. Their model was based on the product of estimated specific yield at the sediment-water interface and the observed change in hydraulic head. Nuttle and Harvey thus state that these factors coupled with the possibility of preferential flow pathways are sources of error in the prediction of groundwater discharge. The model of Nuttle and Harvey however required a range of complex field measurements.

The model of Miller and McPherson (1991) provided answers in an estuary in south-western Florida for the sort of questions being asked of the Gamtoos estuary. Although it is stated above that models cannot simply be applied to new areas, Miller and McPherson claim that their model can be used in many estuaries but is better suited where the estuarine geometry is simple. The model of Miller and McPherson would be suitable for a system such as the Gamtoos estuary for the following reasons:

- The data required for the calibration and application of the model includes: determination of the cross-sectional area at a number of points covering the extent of the estuary, salinity data at a number of fixed sites under a range of freshwater and tidal conditions and an estimate of daily freshwater input.

- It is most suitable for a long, narrow estuary in which the main freshwater input occurs near the head of the estuary. Furthermore the estuary in which Miller and McPherson applied the model had a mean depth of about 3 m.

- The model determines the effect of river inputs and tidal flushing on the transport of water and waterborne constituents through a series of sequential, one dimensional volume elements, and uses a simple mixing equation to predict changes in salinity which in turn is used to determine residence times.

- The residence time of a point source input (such as an agricultural drainage outlet or freshwater tributary) at any position along the estuary can be determined along with time series constituent concentration profiles at any point.

Much of the data required for the calibration and application of the Miller and McPherson model, is already available for the Gamtoos estuary or would be easy to obtain. Given the vulnerability of the Gamtoos estuary and problems facing management, a model having a predictive capacity is required and thus it is recommended that such a modelling project be undertaken in the Gamtoos estuary in the future.

7. REFERENCES

ADAMS, J.B. AND TALBOT, M.M.B., 1992. The influence of river impoundment on the estuarine seagrass *Zostera capensis* setchell, *Botanica marina*, 35, 69-75.

ALABASTER, J.S. AND LLOYD, R., 1980. *Water Quality Criteria for Freshwater Fish*, Butterworth, London, 297pp.

APHA, 1989. *Standard Methods for the Examination of Water and Wastewater*, 18th ed., AMERICAN PUBLIC HEALTH ASSOCIATION, Washington DC.

ASHTON, P.J., APPLETON, C.C. AND JACKSON, P.B.N., 1986. Ecological impacts and economic consequences of alien invasive organisms in southern African aquatic systems, In: *The ecology and management of biological invasions in southern Africa: Proceedings of the National Synthesis Symposium on the Ecology of Biological Invasions*, by I.A.W. MacDonald, F.J. Kruger and A.A. Ferrar (eds.), Oxford University Press, Cape Town, 324pp.

ASHTON, P.J., CLAASSENS, G.C.D., DE A ESTEVES, F., GROBLER, D.C., HEYNIKE, J., JARVIS, A.C., SEAMAN, M.T., SILBAUER, M.J., THORNTON, J.A. AND WIECHERS, H.N.S., 1984. Eutrophication, In: Limnological criteria for management of water quality in the southern hemisphere, by R.C. Hart and B.R. Allanson (eds.), *South African National Scientific Programme Report* No. 93, Pretoria, 134-152.

ASHTON, P.J. AND WALMSLEY, R.D., 1976. The aquatic fern Azolla and its Anabaena symbiont, *Endeavour*, 35, 39-43.

AUBREY, D.G. AND SPEER, P.E., 1985. A study of non-linear tidal propagation in shallow inlet estuarine systems, Part 1: Observations, *Estuarine, Coastal and Shelf Science*, 21, 185-205.

BACK, W., 1966. Hydrochemical facies and ground water flow patterns in the northern part of Atlantic Coastal Plain, *US Geological Survey Project Paper* 498-A.

BAJWA, M.S. AND JOSAN, A.S., 1989. Effect of gypsum and sodic irrigation water on soil and crop yields in a rice-wheat rotation, *Agricultural Water Management*, 16, 53-61.

BAKER, W.C. AND HORTON, T., 1990. Runoff and the Chesapeake Bay, *EPA Journal*, 16, 6, 13-16.

BAREKET, M., date unknown. *Potassium Nitrate in Crop Nutrition*, Halfa Chemicals, Israel, 5-35.

BECK, L.A., 1984. Case history: San Joaquin Valley, *California Agriculture*, 38, 16-17.

BENTLEY, K., 1992. *Eastern Province Herald*, 18 November 1992.

BIELESKI, R.L. AND LAUCHLI, A., 1992. Phosphate uptake, efflux and deficiency in the water fern Azolla, *Plant, Cell and Environment*, 15, 665-673.

BIRCH, P.B., 1982. Phosphorus export from coastal plain drainage into the Peel-Harvey estuarine system of western Australia, *Australian Journal of Marine and Freshwater Research (East Melbourne)*, 33, 1, 23-32.

BIRCH, P.B., GABRIELSON, J.O. AND HODGKIN, E.P., 1984. A Peel-Harvey estuarine system review of study progress, *Water (Sydney)*, 11, 2, 17-20.

BLACK, C.A. (ed.), 1965. *Methods of Soil Analysis*, No. 9 in the series Agronomy, American Society of Agronomy, Madison, USA.

BOKUNIEWICZ, H., 1980. Groundwater seepage into Great South Bay, New York, *Estuarine and Coastal Marine Science*, 10, 437-444.

BONN, H.L., MCNEAL, B.L. AND O'CONNOR, G.A., 1985. *Soil Chemistry*, Wiley, New York, 153-261.

BOS, M.G., 1994. Basics of groundwater flow, In: *Drainage Principles and Applications*, by H.P. Ritzema (ed.), ILRI Publication 16, 2nd ed., Netherlands, 225-262.

BOS, M.G. AND WOLTERS, W., 1994. Influences of irrigation on drainage, In: *Drainage Principles and Applications*, H.P. Ritzema (ed.), ILRI Publication 16, 2nd ed., Netherlands, 513-532.

BOTHA, A.,1994. Personal communication, Department of Agriculture, Port Elizabeth.

BOUWER, H., 1978. *Groundwater Hydrology*, McGraw-Hill, New York, 467pp.

BOUWER, H., 1987. Effect of irrigated agriculture on groundwater, *Journal of Irrigation and Drainage Engineering*, 113, 1, 4-15.

BRASSINGTON, R., 1990. *Field Hydrology*, Halsted, New York.

BRUTON, M.N., 1985. The effects of suspensoids on fish, *Hydrobiologia*, 1, 221-241.

BURT, T. AND HAYCOCK, N., 1991. Farming and nitrate pollution, *Geography*, 60-63.

CAPONE, D.G. AND BAUTISTA, M.F., 1985. A groundwater source of nitrate in nearshore sediments, *Nature*, 313, 214-216.

CAPONE, D.G. AND SLATER, J.M., 1990. Interannual patterns of water table height and groundwater derived nitrate in nearshore sediments, *Biogeochemistry*, 10, 277-288.

CAVELAARS, J.C., VLOTMAN, W.F. AND SPOOR, G., 1994. Subsurface drainage systems, In: *Drainage Principles and Applications*, by H.P. Ritzema (ed.), ILRI Publication 16, 2nd ed., Netherlands, 827-930.

CHAMBERS, R.M., FOURQUREAN, J.W., HOLLIBAUGH, J.T. AND VINK, S.M., 1995. Importance of terrestrially-derived particulate phosphorus to phosphorus dynamics in a West Coast estuary, *Estuaries*, 18, 3, 518-526.

CHAPMAN, H.C., 1986. Canal irrigation schemes in perspective, *SA Waterbulletin*, 12, 2, 6-10.

CHIEW, F.H.S. AND MACMAHON, M.C., 1991. Groundwater recharge from rainfall and irrigation in the Campaspe River Basin, *Australian Journal of Soil Research*, 29, 5, 651-670.

CIFUENTES, L.A., SCHEMEL, L.E. AND SHARP, J.H., 1990. Qualitative and numerical-analyses of the effects of river inflow variations on mixing diagrams in estuaries, *Estuarine, Coastal and Shelf Science*, 30, 4, 411-427.

CUTTLE, S.P., 1989. Land use changes and inputs of nitrogen to Loch Leven, Scotland: A desk study, *Agricultural Water Management*, 16, 119-135.

CYRUS, D.P. AND BLABER, S.J.M., 1992. Turbidity and salinity in a tropical northern Australian Estuary and their influence on fish distribution, *Estuarine, Coastal and Shelf Science*, 35, 345-563.

CZERNY, A.B. AND DUNTON, K.H., 1995. The effects of in situ light reduction on the growth of two subtropical seagrasses *Thalassia testudinum* and *Halodule wrightii*, *Estuaries*, 18, 2, 418-427.

DALLAS, H. F. AND DAY, J.A., 1993. The effect of water quality variables on riverine ecosystems: A review, *Water Research Commission Report* TT 61/93, Freshwater Research Institute, University of Cape Town.

DARDEAU, M.R., MODLIN, R.F., SCHROEDER, W.W. AND STOUT, J.P., 1992. Estuaries, In: *Biodiversity of southeastern United States/Aquatic communities*, by C.T. Hackney, S.M. Adams and W.A. Martin (eds.), Wiley and Sons, New York, 615-744.

DAUER, D. M., RODI, A.J. AND RANASINGHE, J.A., 1992. Effects of low oxygen on the macrobenthos of the Lower Chesapeake Bay, *Estuaries*, 15, 384-391.

DAVIS, S.N. AND DE WIEST, R.J.M., 1966. Hydrogeology, Wiley, New York, 463 pp.

DAY, J.H. (ed.), 1981. *Estuarine Ecology with particular reference to Southern Africa*, Balkema, Cape Town, 411pp.

DENVER, J.M., 1986. Hydrogeology and geochemistry of the unconfined aquifer, west-central and southwestern Delaware, Report of Investigations No. 41, *Delaware Geological Survey*, University of Delaware, Newark, Delaware.

DEPARTMENT OF WATER AFFAIRS AND FORESTRY, 1992. Paul Sauer Dam, Gamtoos Government water scheme, Communication Services, Department of Water Affairs and Forestry, Pretoria, 6pp.

DEPARTMENT OF WATER AFFAIRS AND FORESTRY, 1993a. *Water Quality Guidelines for Agricultural Use*, Volume 4, Department of Water Affairs and Forestry, Pretoria, 286pp.

DEPARTMENT OF WATER AFFAIRS AND FORESTRY, 1993b. *Water Quality Guidelines for Industrial Use*, Volume 3, Department of Water Affairs and Forestry, Pretoria, 286pp.

DEVEREL, S.J. AND FIO, J.L., 1991. Groundwater flow and solute movement to drain laterals, western San Joaquin valley, California, 1. Geochemical Assessment, *Water Resources Research*, 27, 9, 2233-2246.

DRISCOLL, F.G., 1986. *Groundwater and Wells*, 2nd ed., Johnson Division, Minnesota, 1089pp.

DUDA, A.M., 1983. Municipal point source and agricultural nonpoint source contributions to coastal eutrophication, *Water Resources Bulletin (Urbana Ill)*, 18, 3, 397-407.

DU PREEZ, D.R. AND BATE, G.C., 1991. Problems involved in the determination of chlorophyll *a* by HPLC, *South African Journal of Science*, 87, 6, 265-267.

DU PREEZ, I., 1993. Personal communication, Land-owner in the Loerie Flats.

DURBAN CORPORATION, 1991. *Survey Handbook*, 6th ed., Durban Institute of Topographical and Engineering Surveyors of South Africa, City Engineers Department. 152-164.

DZYDEVICH, S.V., SHULGA, A.A., SOLDATKIN, A.P., HENDJII, A.M.N., JAFFREZICRENAULT, N. AND MARTELET, C., 1994. Conductometric biosensors based on cholinesterases for sensitive detection of pesticides, *Electroanalysis*, 6, 9, 752-758.

EMMERSON, W.D. 1985. The nutrient status of the Swartkops River estuary, Eastern Cape, *Water SA*, 11, 4, 189-198.

EMMERSON, W.D., 1989. The nutrient status of the Sundays River estuary, South Africa, *Water Research*, 23, 1059-1067.

EMMERSON, W.D. AND ERASMUS, T., 1987. The nutrient status of the Kromme River estuary, South Africa, *Hydrobiologia*, 148, 87-96.

ERSKINE, A.D., 1991. The effect of tidal fluctuations on a coastal aquifer in the UK, *Ground Water*, 29, 4, 556-562.

FAIRCHILD, D.M. (ed.), 1987. *Ground Water Quality and Agricultural Practices*, Lewis, Chelsea, 402pp.

FAO, 1985. Water Quality for Agriculture, Irrigation and Drainage paper 29, Rev. 1, FAO, Rome, 174pp. As cited in *Drainage Principles and Applications*, by H.P. Ritzema (ed.), ILRI Publication 16, 2nd ed., Netherlands, 533-600.

FICHEZ, R., JICKELLS, T.D. AND EDMUNDS, H.M., 1992. Algal blooms in high turbidity, a result of the conflicting consequences of turbulence on nutrient cycling in a shallow water estuary, *Estuarine, Coastal and Shelf Science*, 35, 577-592.

FREEZE, R.A. AND CHERRY, J.A., 1979. *Groundwater*, Prentice-Hall, New Jersey, 604pp.

GAMTOOS IRRIGATION BOARD, Department of Agricultural Technical Services 1: 4 000 map of the drainage and water table levels in the Loerie Flats area.

GARVINE, R.W., MCCARTHY, R.K. AND WONG, K-C., 1992. The axial salinity distribution in the Delaware estuary and its weak response to river discharge, *Estuarine, Coastal and Shelf Science*, 35, 157-165.

GASTON, G.R., 1985. Effects of hypoxia on macrobenthos of the Inner Shelf off Cameron, Louisiana, *Estuarine, Coastal and Shelf Science*, 20, 603-613.

GEOLOGICAL MAP, 1: 250 000 sheet 3324.

GIESE, B.S. AND JAY, D.A., 1989. Modelling tidal energetics of the Columbia River estuary, *Estuarine, Coastal and Shelf Science*, 29, 6, 549-571.

GILLIOM, R.J., ALEXANDER, R.B. AND SMITH, R.A., 1985. Pesticides 1975-1980, *US Geological Survey Water-Supply Paper* 2271, 1-23.

GLOTFELTY, D.E., TAYLOR, A.W., ISENEE, A.R., JERSEY, J. AND GLENN, S., 1984. Atrazine and simazine movement to Wye River Estuary, *Journal of Environmental Quality (Madison Wisconsin)*, 13, 1, 115-121.

GOMME, J.W., SHURVELL, S., HENNINGS, S.M. AND CLARK, L., 1991. Hydrology of pesticides in a chalk catchment: Surface waters, *Journal of Irrigation, Water and Environmental Management*, October 1991.

GONCALVES, F., RIBEIRO, R. AND SOARES, A.M.V.M., 1995. Laboratory study of effects of temperature and salinity on survival and larval development of a population of *Rhithropanopeus harrisii* from the Mondego River estuary, Portugal, *Marine Biology*, 121, 639-645.

GOODRICH, D.M., BOICOURT, W.C., HAMILTON, P. AND PRITCHARD, D.W., 1987. Wind-induced destratification in Chesapeake Bay, *Journal of Physical Oceanography*, 17, 2232-2240.

GOSCHEN, W.S. AND SCHUMANN, E.H., 1995. Upwelling and the occurrence of cold water around Cape Recife, Algoa Bay, South Africa, *South African Journal of Marine Science*, 16, 57-67.

GRACZYK D. J. AND SONZOGNI, W.C., 1991. Reduction of dissolved oxygen concentration in Wisconsin streams during summer runoff, *Journal of Environmental Quality*, 20, 2, 445-451.

GREB, S.R. AND GRACZYK, D.J., 1993. Dissolved oxygen characteristics of Wisconsin Streams, *Water Science Technology*, 28, 575-581.

GREEN, R.E., GOSWAMI, K.P., MUKHTAR, M. AND YOUNG, H.Y., 1977. Herbicides from cropped watersheds in stream and estuarine sediments in Hawaii, *Journal of Environmental Quality (Madison Wisconsin)*, 6, 2, 145-154.

GRIFFIN, D.A. AND LEBLOND, P.H., 1990. Estuary/ocean exchange controlled by spring-neap tidal mixing, *Estuarine, Coastal and Shelf Science*, 30, 275-297.

GUPTA, S.J., LONSDALE, D.J. AND WANG, D.P., 1994. The recruitment patterns of an estuarine copepod - a biological-physical model, *Journal of Marine Research*, 52, 4, 687-710.

HALLBERG, G.R., 1986. From hoes to herbicides, agriculture and groundwater quality, *Journal of Soil and Water Conservation*, Nov-Dec., 357-364.

HAUGHTON, S.H., 1937. *Geology of portion of the coastal belt near the Gamtoos valley, Cape Province*: An explanation of sheet nos. 151 north and 151 south (Gamtoos River), Department of Mines, Geological Survey, Government Printer, Pretoria.

HECHT, T. AND VAN DER LINGEN, C.D., 1992. Turbidity-induced changes in feeding strategies of fish in estuaries, *South African Journal of Zoology*, 27, 3, 95-106.

HELLAWELL, J.M., 1986. *Biological Indicators of Freshwater Pollution and Environmental Management*, Elsevier Applied Science, New York, 546pp.

HENDJII, A.M.N., JAFFREZICRENAULT, N., MARTELET, C., CLECHET, P., SHULGA, A.A., STRIKHA, V.I., NETCHIPORUL, L.I., SOLDATKIN, A.P. AND WLODARSKI, W.B., 1993. Sensitive detection of pesticides using a differential Isfet-based system with immobilized cholinesterases, *Analytica Chimica Acta*, 281, 1, 3-11.

HENG, L.K., WHITE, R.E., BOLAN, N.S. AND SCOTTER, D.R., 1991. Leaching losses of major nutrients from a mole-drained soil under pasture, *New Zealand Journal of Agricultural Research*, 34, 3, 325-334.

HEROLD, J., 1996. Salinization of the Coerney River, *Water Research Commission Report*. (In press).

HEYDORN, A.E.F. AND GRINDLEY, J.R. (eds.), 1981. Estuaries of the Cape Part II: Synopsis of available information on individual systems, Report no. 7 Gamtoos (CMS48), *CSIR Research Report* 406, 40pp.

HEYDORN, A.E.F. AND TINLEY, K.L., 1980. Estuaries of the Cape Part 1: Synopsis of the Cape Coast, natural features, dynamics and utilisation, *CSIR Reasearch Report* 380, Stellenbosch, 1980, 96pp.

HILMER, T., 1990. Factors influencing the measurement of primary production in small estuaries, Unpublished PhD Thesis, University of Port Elizabeth, Port Elizabeth.

HOLDEN, P.W., 1986. *Pesticides and Groundwater Quality Issues and Problems in Four States*, National Academy Press, Washington DC, 124pp.

HOORN, J.W. AND VAN ALPHEN, J.G., 1994. Salinity control, In: *Drainage Principles and Applications*, by H.P. Ritzema (ed.), ILRI Publication 16, 2nd ed., Netherlands, 533-600.

HOPKINSON, C.S. AND VALLINO, J.J., 1995. The relationships among man's activities in watersheds and estuaries: A model of runoff effects on patterns of estuarine community metabolism, *Estuaries*, 18, 4, 598-621.

INSTITUTE FOR COASTAL RESEARCH, Unpublished data, University of Port Elizabeth.

JAMES, H.R. AND FENTON, T.E., 1993. Division S-5 - Soil genesis, morphology and classification water tables in paired artificially drained and undrained soil catenas in Iowa, *American Journal of the Soil Science Society*, 57, 774-781.

JAY, D.A. AND SMITH, J.D., 1990. Residual circulation in shallow estuaries, 2. Weakly stratified and partially mixed, narrow esturaries, *Journal of Geophysical Research*, 95, C1, 733-748.

JENKINS, G.M. AND WATTS, D.G., 1968. Spectral analysis and its applications, Holden-Day, San Francisco, 525pp.

JERLING, H.L. AND WOOLDRIDGE, T.H., 1994. The mesozooplankton of a freshwater-starved estuary, In: *Changes in Fluxes in Estuaries*, by K.R. Dyer and R.J. Orth (eds.), Olsen and Olsen, Fredensborg, Denmark, 301-306.

JEZEWESKI, W.A. AND ROBERTS, C.P.R., 1986. Estuarine and lake freshwater requirements, *Department of Water Affairs Technical Report* No. TR 129, Government Printer, Pretoria, 2-21.

JOHANNES, R.E., 1980. The ecological significance of the submarine discharge of groundwater, *Marine Ecology - Progress Series*, 3, 365-373.

JOUBERT, P., 1992. Die uitwerking van watertekorte in die Gamtoosvallei, Gamtoos Besproiingsraad, Opgestel vir die Koordinerende Komitee vir Besproeiingsvoorligting.

KENNISH, M.J., 1986. *Ecology of Estuaries*, Vol. 1, Physical and Chemical Aspects, CRC Press, Florida.

KIM, C.S., HOSTETLER, J. AND AMACHER, G., 1993. The regulation of groundwater quality with delayed responses, *Water Resources Research*, 29, 5, 1369-1377.

KIMMERER, W.J., SMITH, S.V. AND HOLLIBAUGH, J.T., 1993. A simple heuristic model of nutrient cycling in an estuary, *Estuarine, Coastal and Shelf Science*, 37, 2, 145-159.

KIRK, J.T.O. AND AKHURST, E.G.J., 1984. Turbidity and suspensoids, In: Limnological criteria for management of water quality in the southern hemisphere, by R.C. Hart and B.R. Allanson (eds.), *South African National Scientific Programme Report* No. 93, Pretoria, 108-133.

KONIKOW, L.F. AND PERSON, M., 1985. Assessment of long term salinity changes in an irrigated stream - aquifer system, *Water Resources Research (Washington DC)*, 21, 11, 1611-1624.

KUMARAN, S. AND MORITA, M., 1995. Application of a cholinesterase biosensor to screen for organophosphorus pesticides extracted from soil, *Talanta*, 42, 4, 649-655.

KUNISHI, H.M., 1988. Sources of nitrogen and phosphorus in an estuary of the Chesapeake Bay, *Journal of Environmental Quality (Madison Wisconsin)*, 17, 2, 185-188.

KUO, A. Y. AND NEILSON, B.J., 1987. Hypoxia and salinity in Virginia Estuaries, *Estuaries*, 10, 277-282.

LANKFORD, T.E. AND TARGETT, T.E., 1994. Suitability of estuarine nursery zones for juvenile weakfish (Cynoscion regalis): Effects of temperature and salinity on feeding, growth and survival, *Marine Biology*, 119, 611-620.

LARGIER, J.L., 1986. Structure and mixing in the Palmiet estuary, South Africa, *South African Journal of Marine Science*, 4, 139-152.

LARGIER, J.L. AND SLINGER, J.H., 1991. Circulation in highly stratified Southern African estuaries, *Southern African Journal of Aquatic Sciences*, 17, 103-115.

LLOYD, R. AND SWIFT, D.J., 1976. Some physiological responses by freshwater fish to low dissolved oxygen, high carbon dioxide, ammonia and phenol with particular reference to water balance, In: *Effects of pollutants on aquatic organisms*, by A.P.M. Lockwood (ed.), Cambridge University Press, London.

MACEWAN R.J., GARDNER, W.K., ELLINGTON, A., HOPKINS, D.G. AND BAKKER, A.C., 1992. Tile and mole drainage for control of waterlogging in duplex soils of south-eastern Australian, *Australian Journal of Experimental Agriculture,* 25, 865-878.

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 No. KV 45/93, 217pp.

MACKAY, H.M. AND SCHUMANN, E.H., 1990. Mixing and circulation in the Sundays River estuary, South Africa, *Estuarine, Coastal and Shelf Science*, 31, 203-216.

MACKAY, H.M. AND SCHUMANN, E.H., 1991. Tidal and long-period water level variations in the Sundays River estuary, South Africa, *South African Journal of Science*, 87, 597-600.

MADDEN, C.J. AND KEMP, W.M., 1996. Ecosystem model of an estuarine submersed plant community - calibration and simulation of eutrophication responses, *Estuaries*, 19, 2B, 457-474.

MARAIS, J.F.K., 1988. Some factors that influence fish abundance in South African estuaries, *South African Journal of Marine Science*, 6, 67-77.

MATTHESS, G., 1982. *The Properties of Groundwater*, Wiley, New York, 406pp.

MAYERS, J.R. AND ELKINS, N.R., 1990. Potential for agricultural pesticide runoff to a Puget Sound estuary: Padilla Bay, Washington, *Bulletin of Environmental Contamination and Toxicology (New York)*, 45, 2, 215-222.

MCKENNEY, C.L. AND CELESTIAL, D.M., 1995. Interactions among salinity, temperature and age on growth of the estuarine mysid *Mysidopsis bahia* reared in the laboratory through a complete life cycle. 1. Body mass and age-specific growth rate, *Journal of Crustacean Biology*, 15, 1, 168-178.

MEDYANTSEVA, E.P., VERTLIB, M.G., BUDNIKOV, G.K., BABKINA, S.S AND EREMIN, S.A., 1995. Amperometric biochemical sensor based on immobilized cholinesterase in the immunoassay of pesticides, *Journal of Analytical Chemistry*, 50, 7, 719-722.

MERINO, M., CZITROM, S., JORDAN, E., THOME, P. AND MORENO, O., 1990. Hydrology and rain flushing of the Nichupte Lagoon System, Cancun, Mexico, *Estuarine, Coastal and Shelf Science*, 30, 223-237.

MIDGLEY, D.C., PITMAN, W.V. AND MIDDLETON, B.J., 1994a. Surface Water Resources of South Africa 1990, Users Manual, *Water Research Commission Report* No. 298/1/94.

MIDGLEY, D.C., PITMAN, W.V. AND MIDDLETON, B.J., 1994b. Surface Water Resources of South Africa 1990, Volume IV Drainage regions E, G, H, J, K, L, Appendices, *Water Research Commission Report* No. 298/4.1/94.

MIDGLEY, D.C., PITMAN, W.V. AND MIDDLETON, B.J., 1994c. Surface Water Resources of South Africa 1990, Volume IV Drainage regions E, G, H, J, K, L, Book of Maps, *Water Research Commission Report* No. 298/4.2/94.

MILLER, R.L. AND MCPHERSON, B.F., 1991. Estimating estuarine flushing and residence times in Charlotte Harbor, Florida, via salt balance and a box model, *Limnology and Oceanography*, 36, 3, 602-612.

MILLHAM, N.P. AND HOWES, B.L., 1994. Nutrient balance of a shallow coastal embayment: I. Patterns of groundwater discharge, *Marine Ecology Progress Series*, 112, 155-167.

MILLS, H.A., 1985. Effect of selected agricultural chemicals on denitrification and the potential for nitrate pollution, Paper presented at Conference, Nitrates dans les Eaux, Paris. As cited in: A preliminary investigation of the nitrate content of groundwater and the limitation of the nitrate input, by G. Tredoux, *Water Research Commission Report* No. 368/1/93.

NAKAMOTO, R.J. AND HASSLER, T.J., 1992. Selenium and other trace metals in Bluegills from agricultural return flows in the San Joaquin Valley, California, *Archives of Environmental Contamination Toxicology*, 22, 1, 88-98.

NEWSON, M., 1994. *Hydrology and the River Environment*, Clarendon, Oxford, 221 pp.

NICHOLAICHUK, W., BEST, K., GROVER, R., AND WHITING, J., 1991. Herbicide, nutrient and drainage from irrigated fields, Hydrological Interactions between Atmosphere, Soil and Vegetation, Symposium held during the Twentieth General Assembly of the International Union of Geodesy and Geophysics, *IAHS Publication* No. 204, 467-476.

NIELSEN, P., 1990. Tidal dynamics of the water table in beaches, *Water Resources Research*, 26, 9, 2127-2134.

NUTTLE, W.K. AND HARVEY, J.W., 1995. Fluxes of water and solute in a coastal wetland sediment. 1. The contribution of regional groundwater discharge, *Journal of Hydrology*, 164, 1, 89-107.

OOSTERBAAN, R.J., 1988. Agricultural criteria for subsurface drainage: A systems analysis, *Agricultural Water Management*, 14, 79-90.

OOSTERBAAN, R.J., 1994. Agricultural drainage criteria, In: *Drainage Principles and Applications*, by H.P. Ritzema (ed.), ILRI Publication 16, 2nd ed., Netherlands, 635-687.

OOSTERBAAN, R.J., AND NIJLAND, H.J., 1994. Determining the saturated hydraulic conductivity, In: *Drainage Principles and Applications*, by H.P. Ritzema (ed.), ILRI Publication 16, 2nd ed., Netherlands, 435-476.

PEARCE, M. W. AND SCHUMANN, E.H., 1994. Water quality of an agricultural drainage system in the Gamtoos floodplain, In: *Proceedings of the 50 years of Water Engineering in South Africa: A tribute to Des Midgley Conference*, University of the Witwatersrand, Johannesburg, 14-15 July 1994, 563-571.

PETTS, G.E., 1984. *Impounded Rivers: Perspectives for Ecological Management*, Wiley, Chichester.

PORT ELIZABETH MUNICIPALITY, Water resources and supply pipelines map.

RACTLIFFE, G., 1991. The effects of suspended sediments on the macroinvertibrate community structure of a River ecosystem, Unpublished Honours project, University of Cape Town, South Africa, 57pp.

RAINS, D.W. AND TALLEY, S.N., 1979. Use of *Azolla* in North America, In: *Nitrogen and Rice*, International Rice Research Institute, Los Banos, Philippines, 419-431.

REAY, W.G., GALLAGHER, D.L. AND SIMMONS, G.M., 1992. Groundwater discharge and its impact on surface water quality in a Chesapeake Bay inlet, *Water Resources Bulletin*, 28, 6, 1121-1134.

REDDERING, J.S.V., 1988. Prediction of the effects of reduced river discharge on the estuaries of the south eastern Cape Province, South Africa, *South African Journal of Science*, 84, 726-730.

REDDERING, J.S.V. AND ESTERHUIZEN, K., 1984. Sedimentation in the Gamtoos estuary, ROSIE (Research on Sedimentation In Estuaries) Report 7, University of Port Elizabeth, 85pp.

REDDERING, J.S.V. AND RUST, I.C., 1990. Historical changes and sedimentary characteristics of Southern African estuaries, *South African Journal of Science*, 86, 425-428.

REDDERING, J.S.V. AND SCARR, N.G., 1990a. Sediment dispersal in the Gamtoos estuary, Institute for Coastal Research Report Number 24, University of Port Elizabeth.

REDDERING, J.S.V. AND SCARR, N.G., 1990b. The physical characteristics of the Gamtoos estuary, I.C. Rust (ed.), Institute for Coastal Research Report Number 23, University of Port Elizabeth.

REDDY, B.S.R. AND RAO, V.R., 1994. Seasonal variation in temperature and salinity in the Gautami-Godavari estuary, *Proceedings of the Indian Academy of Science - Earth and Planetary Sciences*, 103, 1, 47-55.

REID, I. AND PARKINSON, R.J., 1984. The nature of the tile drain outfall hydrograph in heavy clay soils, *Journal of Hydrology*, 72, 289-305.

REYES, E. AND MERINO, M., 1991. Diel dissolved oxygen dynamics and eutrophication in a shallow well-mixed tropical lagoon (Cancun, Mexico), *Estuaries*, 14, 372-381.

REYNDERS, A., 1992. Personal communication, Water Research Commission, Pretoria.

RICE, R.C., BOWMAN, R.S. AND BOUWER, H., 1989. Ionic composition of vadoze zone water in an arid region, *Ground Water*, 27, 6, 813-822.

RITTER, W.F., 1989. Nitrate leaching under irrigation in the United States, *Journal of Environmental Science and Health*, A24, 4, 349-378.

RITTER, W.F., CHIRNSIDE, A.E.M. AND SCARBOROUGH, R.W., 1990. Soil nitrate profiles under irrigation on coastal plain soils, *Journal of Irrigation and Drainage Engineering*, 116, 6, 738-751.

RITTER, W.F., MEMBER, A.S.C.E., HUMENIL, K.F.J. AND SKAGGS, R.W., 1989. Irrigated agriculture and water quality in east, *Journal of Irrigation and Drainage Engineering*, 115, 5, 807-821.

RITZEMA, H.P. (ed.), 1994a. *Drainage Principles and Applications*, ILRI Publication 16, 2nd ed., Netherlands, 1125pp.

RITZEMA, H.P., 1994b. Subsurface flow to drains, In: *Drainage Principles and Applications*, by H.P. Ritzema (ed.), ILRI Publication 16, 2nd ed., Netherlands, 263-303.

SABOL, G.V., BOUWER, H. AND WIERENGA P.J., 1987. Irrigation effects in Arizona, *Journal of Irrigation and Drainage Engineering*, 113, 1, 30-48.

SCHMIDT, K.D. AND SHERMAN, I., 1987. Effect of irrigation on groundwater quality in California, *Journal of Irrigation and Drainage Engineering*, 113, 1, 16-29.

SCHROEDER, W.W., 1977. The impact of the 1973 flooding of the mobile Bay System on the hydrography of Mobile Bay and east Mississippi Sound, *Northeast Gulf Science*, 1, 2, 68-76.

SCHROEDER, W.W. (Convenor), 1985. The occurrence of hypoxic and anoxic conditions in estuaries and coastal environments, Abstracts for the 8th Biennial Estuarine Research Conference, *Estuaries*, 8, 2B, 43A-47A.

SCHROEDER, W.W., DINNEL, S.P. AND WISEMAN, W.J., 1990a. Salinity stratification in a river-dominated estuary, *Estuaries*, 13, 145-154.

SCHROEDER, W.W., WISEMAN, W.J. AND DINNEL, S.P., 1990b. Wind and river induced fluctuations in a small, shallow, tributary estuary, In: Residual Currents and Long-term Transport, by R.T. Cheng (ed.), Springer-Verlag, New York, *Coastal and Estuarine Studies*, 38, 481-493.

SCHULZE, B.R., 1986. *Climate of South Africa*, 6th ed., Part 8, General Survey, Weather Bureau, Department of Transport, Government Printer, Pretoria.

SCHUMANN, E.H. AND BRINK, K.H., 1990. Coastal trapped waves off the coast of South Africa: Generation, propagation and current structures, *Journal of Physical Oceanography*, 20, 1206-1218.

SCHUMANN, E.H. AND DE MEILLON, L., 1993. Hydrology of the St. Francis Bay marina, South Africa, *Transactions of the Royal Society of South Africa*, 48, 323-337.

SCHUMANN, E.H., LARGIER, J.L. AND SLINGER, J.H., 1997. Estuarine hydrodynamics, In: *Estuaries of South Africa*, by D. Baird and B. Allanson, Cambridge University Press, Chapter 3.

SCHUMANN, E.H. AND MARTIN, J., 1994. Programs to read estuarine data, Department of Oceanography, University of Port Elizabeth.

SCHUMANN, E. H. AND PEARCE, M.W., 1997. Freshwater inflow and estuarine variability in the Gamtoos estuary, South Africa, *Estuaries,* 20, 1.

SCHUMANN, E.H., ROSS, G.J.B. AND GOSCHEN, W.S., 1988. Cold water events in Algoa Bay and Cape south coast, South Africa, in March/April, 1987, *South African Journal of Science*, 84, 579-584.

SECOR, D.H. AND HOUDE, E.D., 1995. Temperature effects on the timing of striped bass egg production, larval viability and recruitment potential in the Patuxent River (Chesapeake Bay), *Estuaries,* 18, 3, 527-544.

SERFES, M.E., 1991. Determining the mean hydraulic gradient of ground-water affected by tidal fluctuations, *Ground Water*, 29, 4, 549-555.

SHARPLES, J., SIMPSON, J.H. AND BRUBAKER, J.M., 1994. Observations and modeling of periodic stratification in the Upper York River estuary, Virginia, E*stuarine, Coastal and Shelf Science*, 38, 3, 301-312.

SHONE, R.W., NOLTE, C.C. AND BOOTH, P.W.K., 1990. Pre-Cape rocks of the Gamtoos area - a complex tectonostratigraphic package preserved as a horst block, *South African Journal of Geology*, 93, 4, 616-621.

SKLADAL, P., 1992. Detection of organophosphate and carbamate pesticides using disposable biosensors based on chemically modified electrodes and immobilized cholinesterase, *Analytica Chimica Acta*, 269, 281-287.

SLATER, J.M. AND CAPONE, D.G., 1987. Denitrification in aquifer soil and nearshore marine sediments influenced by groundwater nitrate, *Applied and Environmental Microbiology*, 52, 1292-1297.

SONNEN, M.B., THOMAS, J.L. AND GUITJENS, J.C., 1987. Irrigation effects in six western states, *Journal of Irrigation and Drainage Engineering*, 113, 1, 57-68.

SOUTH AFRICAN TIDE TABLES, 1992. South African Naval Hydrographer, Tokai, Cape Town.

SOUTH AFRICAN TIDE TABLES, 1993. South African Naval Hydrographer, Tokai, Cape Town.

STANLEY, D.W., 1993. Long-term trends in Pamlico River estuary nutrients, chlorophyll, dissolved oxygen and watershed nutrient production, *Water Resources Research*, 29, 8, 2651-2662.

STAVER, K.W. AND BRINSFIELD, R.B., 1996. Seepage of groundwater nitrate from a riparian agrosystem into the Wye River estuary, *Estuaries*, 19, 2B, 359-370.

STAVER, L.W., STAVER, K.W. AND STEVENSON, J.C., 1996. Nutrient inputs to the Choptank River estuary: Implications for watershed management, *Estuaries*, 19, 2B, 342-358.

STREBEL, O., DUYNISVELD, W.H.M. AND BOTTCHNER, J., 1989. Nitrate pollution of groundwater in western Europe, *Agriculture, Ecosystems and Environment*, 26, 189-214.

SUBUDHI, B.P.R. AND WATANABE, I., 1981. Differential phosphorus requirements of *Azolla* species and strains in phosphorus limited continuous culture, *Soil Science and Plant Nutrition*, 27, 2, 237-247.

TETT, P., KELLY, M.G. AND HORNBERGER, G.M., 1977. Extraction of chlorophyll *a* and pheophytin a in methanol, *Limnology and Oceanography*, 22, 3, 579-580.

THOLLAPALLI, A., APRIL, G.C., RANEY, D.C., SCHROEDER, W.W., MCCORMICK, B.C. AND LU, Z., 1991. Computer simulation of the hydrodynamics and salinity behavior of Weeks Bay, Alabama, at equatorial tide conditions, *Water Science and Technology*, 24, 6, 165-169.

TOERIEN, D.K. AND HILL, R.S., 1989. The Geology of the Port Elizabeth area: Explanation of sheet 3324, scale 1:250 000, Department of Mineral and Energy Affairs, Geological Survey, Pretoria, 35pp.

TREDOUX, G., 1993. A preliminary investigation of the nitrate content of groundwater and the limitation of the nitrate input, *Water Research Commission Report* No. 368/1/93, 76pp.

TYSON, P.D., 1986. *Climatic Change and Variability in Southern Africa,* Oxford University Press, Cape Town.

VAN DER WALT, 1996. Personal communication, Department of Agriculture, Pretoria.

VAN ZYL, M., 1994. A study of the factors influencing groundwater levels and direction of groundwater movement in the Gamtoos valley, Unpublished Honours Thesis, University of Port Elizabeth, 55pp.

VILLENEUVE, J.P., BANTON, O. AND LAFRANCE, P., 1990. A probabilistic approach for the groundwater vulnerability to contamination by pesticides - the vulpest model, *Ecological Modelling,*51, 1, 47-58.

WANG, X.H. AND CRAIG, P.D., 1993. An analytic model of tidal circulation in a narrow estuary, *Journal of Marine Research*, 51, 3, 447-465.

WATANABE, I., BERJA, N.S. AND DEL ROSARIO, D.C., 1980. Growth of *Azolla* in paddy fields as affected by phosphorus fertilizer, *Soil Science and Plant Nutrition*, 26, 2, 301-307.

WEATHER BUREAU, 1994. Rainfall records, Weather Bureau, Pretoria.

WEAVER, J.M., 1993. A preliminary survey of pesticide levels in groundwater from a selected area of intensive agriculture in the western Cape, *Water Research Commission Report* No 268/1/93.

WHITE, J.C., 1989. Sources and distribution of pesticide residues in an agricultural area: Vaalharts irrigation scheme, Unpublished Internal Report, Hydrological Research Institute, Department of Water Affairs and Forestry.

WHITFIELD, A.K. AND BRUTON, M.N., 1989. Some biological implications of reduced water inflow into eastern Cape estuaries: A preliminary assessment, *South African Journal of Science*, 85, 691-694.

WHITFIELD, A. K. AND PATERSON, A.W., 1995. Flood-associated mass mortality of fishes in the Sundays estuary, *Water SA*, 21, 385-389.

WICHELNS, D. AND NELSON, D., 1989. An empirical model of the relationship between irrigation and the volume of water collected in subsurface drains, *Agricultural WaterManagement*, 16, 293-308.

WONG, K-C., 1995. The hydrography at the mouth of Delaware Bay: Tidally averaged distribution and intratidal variability, *Estuarine, Coastal and Shelf Science*, 41, 719-736.

WOOLDRIDGE, T.H., 1994. The effect of periodic inlet closure on recruitment in the estuarine mudprawn, *Upogenia africana* (Ortmann), In: *Changes in Fluxes in Estuaries*, by K.R. Dyer and R.J. Orth (eds.), Olsen and Olsen, Fredensborg, Denmark, 329-333.

WOOLDRIDGE, T.H. AND BAILEY, C., 1982. Euryhaline zooplankton of the Sundays River estuary and notes on trophic relationships, *South African Journal of Zoology*, 17, 151-163.

WUTTKE, G., THOBER, B. AND LIETH, H., 1991. Simulation of nitrate transport in groundwater with a 3-dimensional groundwater model run as a subroutine in an agroecosystem model, *Ecological Modelling*, 57, 3, 263-276.

ZAGORC-KONCAN, J., DULAR, M. AND SOEMEN, J., 1991. Evaluation of dissolved oxygen balance in two shallow turbulent Slovene streams, *Water Research*, 25, 11, 1357-1363.

ZHANG, P., 1995. Evolution of the Gamtoos River floodplain, South Africa, Unpublished MSc Thesis, University of Port Elizabeth, Port Elizabeth.

ZHANG, P., RUST, I.S. AND RYNHOUT, M.S., 1996a. The Eemian paleoclimate in estuarine deposits of the Gamtoos River valley, Eastern Cape. (In prep).

ZHANG, P., RUST, I.S. AND RYNHOUT, M.S., 1996b. Late Quaternary paleoestuarine deposits from the Gamtoos River valley, South Africa: Evidence for 8.2 m raised sea level and climatic warming, *Journal of Paleontology*, (In press).

ZHANG, P., SCHUMANN, E.H. AND SHONE, R.W., 1995. Effects of a storm-surge co-incident with a high spring tide in the Gamtoos estuary, May 1992, *South African Journal of Science*, 91, 412-414.