

**STUDIES ON THE RIVER-ESTUARY INTERFACE
REGION OF SELECTED EASTERN CAPE
ESTUARIES**

Report to the Water Research Commission

Edited by

AK Whitfield and AD Wood

**WRC Report No. 756/1/03
ISBN No. 1-86845-980-2
ISBN Set No. 1-86845-978-0**

February 2003

EXECUTIVE SUMMARY

The work described in this report is part of more comprehensive investigations undertaken by the Consortium for Estuarine Research and Management (CERM) for the Water Research Commission (WRC) to improve the predictive capabilities on the effects of changes in river flow on the physical and ecological conditions in estuaries. Focus of this particular research programme was aimed at increasing our knowledge of the river-estuary interface (REI) region of selected Eastern Cape estuaries, and the Gamtoos system in particular. The REI region is defined as that part of an estuary where river and estuarine waters mix, and where the vertically integrated salinity is usually less than 10‰.

The following global hypothesis, developed by the research team conducting this project, was used to guide the different studies:

"The structure and functioning of the river-estuary interface region is governed primarily by the quality, quantity and supply pattern of fresh water received. Furthermore, the interface region has strong influences on the physico-chemical as well as the biological structure and functioning of the entire system."

Estuarine hydrodynamics

Predictions of changes in the physical conditions in the Gamtoos Estuary under different river flow conditions and especially of the effects on salinity distributions in the estuary using the Mike-11 modelling system were undertaken. Several important results were obtained from the model, which played a major role in the complementary investigations on the ecological components of the study.

A large number of model simulations were undertaken to determine the salinity distributions in the Gamtoos Estuary for different river flows. The results are included in the report as graphs for high and low neap and spring tides. These simulations were also undertaken for pre-1996 flood and for post-1996 flood conditions, with significant differences in salinity distributions being found as a result of the scouring effect of these floods. The model results were also used to determine the volumes of water in the estuary below (and between) selected salinity levels of 5, 10, 15, 20, 25, 30 and 35 ‰ for different river flows under pre- and post flood conditions. In this manner the volume of available habitat (based on salinity) for aquatic organisms could be calculated.

Water column and porewater nutrients

Variations in the nutrient status of the Gamtoos Estuary could not be directly linked to riverine flow. Freshwater inputs under base flow conditions (0.35 to $1.6 \text{ m}^3 \text{ s}^{-1}$) did not influence mean estuarine nutrient concentrations significantly. However, phosphate concentrations were positively correlated with freshwater input ($r = 0.67$; $p < 0.05$). Investigations of nutrient concentrations between spring and neap tides showed that tidal movement also had no significant influence on nutrient status. As

freshwater inflow had no direct, significant influence on water column nutrient status, the amount of freshwater inflow required for consistent nutrient inputs to the estuary is unknown. However, it is predicted that a reduction in base flow will influence water column nutrient status through its influence on estuary mouth phase. Closed vs open mouth phases are likely to have different effects on the storage, release and exchange of nutrients within the estuary and between the estuary and sea.

Although some nutrient species were found to occur at higher concentrations in the upper reaches (vertically averaged salinity of 17 ‰ and below) of the Gamtoos Estuary, it was not possible to identify a REI confined within a certain salinity range for any of the nutrients measured. Overall, concentrations decreased towards the mouth in the case of nitrate, nitrite, dissolved organic carbon (DOC) and nitrogen (DON), particulate organic nitrogen (PON) and total particulate phosphorus (TPP). Plots of nutrients against salinity did not reveal strong conservative behaviour for most nutrients, which points towards biological transformation and/or non-point source additions of nutrients along the estuarine gradient.

Although freshwater inputs did not significantly influence the inorganic nutrient status of Gamtoos interstitial waters, sediment porewater quality did significantly affect benthic water nutrient status. Porewater concentrations of nitrate and nitrite were 5 to 10 times higher than respective water column concentrations, while no concentration gradients existed between the water column and porewaters for ammonium and phosphate. Nitrate, nitrite, and ammonium porewater concentrations were significantly correlated with respective concentrations in the bottom waters of the water column ($r = 0.58$, $p < 0.001$; $r = 0.24$, $p < 0.05$; $r = -0.26$, $p < 0.05$, respectively). Both porewater and bottom water phosphate concentrations were very low in the Gamtoos Estuary, indicating a phosphate-limiting environment in this system.

Estuarine microalgae

A primary objective of this particular study was to assess the effects of Gamtoos River flow on the community structure and biomass of estuarine microalgae. The investigation showed that as flow increased the average nitrate in the estuary increased. A base flow of approximately $1.0 \text{ m}^3 \text{ s}^{-1}$ resulted in the highest microalgal biomass as indicated by phytoplankton and benthic microalgal chlorophyll-a concentrations. At most flows, the sites where phytoplankton chlorophyll-a ($\mu\text{g l}^{-1}$) was highest was near ($\pm 1.5 \text{ km}$) to the position where the vertically averaged salinity was 10 ‰ (i.e. mostly 12.5 to 14.5 km from the mouth). Subtidal and intertidal benthic chlorophyll-a values were highest at flows of 1.0 ($57.7 \text{ kg} \pm 0.4 \text{ SE}$) and $1.2 \text{ m}^3 \text{ s}^{-1}$ ($8.7 \text{ kg} \pm 0.1 \text{ SE}$) respectively. Benthic microalgal biomass was highest in regions where the salinity was 10 to 15 ‰, and was positively correlated with porewater phosphate concentrations.

Results from this study provide a preliminary tool that can be used to estimate the base flow for maximum phytoplankton biomass from known estuary volumes. If the total volume of the estuary is $3.6 \times 10^6 \text{ m}^3$, the estuary will be completely flushed in about 42 days at a flow rate of $1.0 \text{ m}^3 \text{ s}^{-1}$. To obtain maximum microalgal biomass a residence time of 3 spring tidal cycles (42 days) is required.

From this, if the estuary volume is known, then the base flow required can be determined from the formula;

$$\text{Base flow (m}^3 \text{ s}^{-1}) = (8.7 \times \text{Volume of estuary})/31536000$$

where; 8.7 = number of times an estuary is flushed per year at a residence time of 42 days

$$\text{Volume of estuary} = \text{volume in m}^3$$

$$31536000 = \text{number of seconds in a year}$$

The succession of phytoplankton groups as a result of base flow changes was also investigated. Phytoplankton were identified into five groups (diatoms, flagellates, dinoflagellates, chlorophytes and cyanophytes), counted and analysed in relation to flow rate. In both the Kromme and Gamtoos estuaries, flagellates increased as flow decreased while diatom numbers peaked at a flow of approximately $1.0 \text{ m}^3 \text{ s}^{-1}$. These two were the dominant groups of phytoplankton at most flows in the Gamtoos and at all flows in the Kromme Estuary. Chlorophytes, dinoflagellates and cyanophytes increased as flow decreased and were most abundant in the shallow upper reaches of the Gamtoos Estuary. Multivariate analyses were used to identify phytoplankton community structure in relation to flow. No differences in phytoplankton community structure were found inside the REI compared to outside of the REI.

Estuarine invertebrates

Studies on both the benthic invertebrates and zooplankton showed that patterns of species distribution between the mouth and upper Gamtoos estuary indicate clear zonation of animal assemblages. Pelagic and benthic assemblages in the REI region differ significantly in their species assemblages and abundance of component species compared to species composition farther downstream. Ecological communities of low salinity regions are therefore distinct in their biological structure.

Benthic communities of the REI are not only unique in their biological structure but display a different trophic organisation compared to assemblages in more saline waters. Filter-feeders dominate low salinity regions, possibly as a response to the high phytoplankton production in this sector, while surface-active deposit feeders dominate downstream in the lower estuary. In addition, food web components in the REI carry distinct carbon 'signatures' compared with organisms in other regions of the estuary. Carbon isotope ratios of primary producer's (phytoplankton, benthic microalgae, macrophytes) increase significantly and linearly with salinity. Similarly, gradients in carbon isotope composition are mirrored by equivalent relationships between $\delta^{13}\text{C}$ and salinity in consumers (prawns, amphiods, crabs, bivalves). This systematic shift in carbon composition across the salinity spectrum in both producers and consumers suggests that production and consumption of organic matter are spatially restricted to particular reaches of the estuary. Processing of energy through the food web within a salinity zone (e.g. in the REI) may therefore lead to limited transport and mixing of organic matter between salinity zones.

Carbon in selected top fish predators is ultimately derived from benthic microalgal production. The transfer of microalgal carbon to these higher trophic levels is channelled via two steps, a grazing of benthic microalgae by mugilid species, and a consumption of these taxa by piscivorous predators. These results stress the energetic importance of benthic microalgae in supporting fish production in the estuarine environment. In addition to benthic microalgae, which form the base of a clear trophic transfer route to piscivorous fish, a second energy pathway that originates from phytoplankton is important for higher trophic levels. Benthic invertebrates that assimilate mostly phytoplankton make a substantial contribution to the diet of benthivorous fish that are abundant in the REI region. Similarly, zooplankton grazing on phytoplankton is the intermediate trophic step that links pelagic primary production with pelagic zooplanktivorous fish. Since freshwater inflow promotes productivity of both benthic and pelagic microalgae in estuaries, supply rates of freshwater to estuaries are clearly a dominant structuring force for estuarine communities at several trophic levels.

Estuarine fishes

A primary objective of this study was to examine the fish assemblages in the REI region of the Great Fish and Kariega estuaries, and relate these structures to riverine influences in particular. Since the Kariega Estuary did not possess a significant REI region due to a lack of riverine flow, the geographical headwaters became a focus of research attention in this particular system.

The fish assemblages along the length of the Kariega and Great Fish estuaries exhibited a clear zonal trend, with the REI/headwater region forming a distinct but different grouping in both systems. Multivariate analyses, which examined the possible influence of various abiotic variables in structuring the fish assemblages associated with the different reaches in the two estuaries, indicated that a combination of variables produced the best correlations. Salinity has traditionally been regarded as one of the more important structuring forces in estuaries and was found to be an important factor in the Great Fish but not in the Kariega system.

In the Great Fish Estuary there were distinct trends between river flow rate and the species assemblages found within the different reaches of the system. Fish abundance in the REI region decreased with increasing riverine flow, with the highest abundance of fish being recorded under flow conditions $<10 \times 10^6 \text{ m}^3 \text{ month}^{-1}$. Species composition also changed, with the marine species contribution to the REI fish assemblage dropping from 60% under lower flow conditions to <30% during periods of elevated riverine flow ($>20 \times 10^6 \text{ m}^3 \text{ month}^{-1}$). A major increase in river flow results in an overall decline in fish abundance throughout the system, with flooding having the potential to cause extensive mortalities within the estuary.

The quality of freshwater entering an estuary may also affect the fish assemblages in the REI region. The high conductivity levels of the water in the Great Fish River may reduce the osmotic stress experienced by marine species in brackish water and therefore assist the penetration of these species into the upper reaches of the estuary and river. Conversely, floods reduce the high conductivity levels of river water and result in a decrease in marine species in the REI region and adjoining river.

Conservation and management

Important issues/principles addressed by this research programme, which will influence the conservation and management of Eastern Cape estuaries, and the REI region in particular, are outlined below.

Estuarine hydrodynamics

- Salinity distributions at different river flows underpin estuarine ecological investigations and enable quantification of ecological components. These distributions are also essential in the assessments of freshwater requirements of estuaries.
- Quantification of retention times is an important ecological tool, especially when making predictions on the impacts of pollutant spills or the effects of high nutrient inputs on estuaries.

Water column and porewater nutrients

- Freshwater inputs were not found to directly influence water column nutrient status of the Gamtoos Estuary but it is predicted that changes in flow regimes caused by managed activities will indirectly affect nutrient status in the system. Minimal baseflows of approximately $1.0 \text{ m}^3 \text{ s}^{-1}$ are required to help disperse and dilute nutrient rich waters caused by allochthonous (e.g. surface run-off) and autochthonous (e.g. sediment porewater) sources.
- Reduced baseflows may result in infilling of the estuary mouth with marine sands (possibly leading to closure), thus reducing hydrological mixing and accelerating the process of eutrophication. This scenario is predicted not as a result of increased allochthonous or autochthonous nutrient loads to the estuary, but because of reduced flushing and dilution effected by tidal exchange and freshwater flushes.

Estuarine microalgae

- Microalgae are the main primary producers in the food-web within the Gamtoos REI. To reach a maximum biomass, a base flow of $1.0 \text{ m}^3 \text{ s}^{-1}$ is required. In the Gamtoos estuary, this flow results in mineral nutrients being imported into the estuary and taken up by microalgae over a period of three spring tidal cycles, before being discharged to the sea.
- Permanently open estuaries require baseflow in order to maintain a productive REI region. The Kromme Estuary receives no baseflow and, as a result of this, primary productivity in the water column is negligible. Increased nutrient input to the Gamtoos Estuary (particularly phosphate) is likely to result in a eutrophic estuary characterised by dinoflagellate blooms. These blooms may be toxic.

Estuarine invertebrates

- The REI region is a unique and important invertebrate habitat for overall estuarine structure and function. Within the estuarine ecosystem, low salinity reaches have unique species assemblages (i.e. biological structure) and largely internal energy pathways (i.e. biological processes) that also contribute substantially to estuarine ecological diversity.
- If the extent of the REI in estuaries is diminished as a consequence of freshwater abstraction, then the unique 'goods and services' derived from this region are diminished. Although a full salinity gradient may be evident, the overall functional value of an estuary for invertebrates may be substantially reduced because of limited habitat remaining in the REI as a consequence of excessive riverine abstraction. This leads to the recommendation that the magnitude or size of the REI must also be considered when determining freshwater requirements of individual estuaries.

Estuarine fishes

- The presence of significant numbers of important recreational and commercial linefish species in the REI region of a freshwater enriched estuary, and their conspicuous absence in a freshwater deprived one, has considerable management implications. If juvenile dusky kob (*Argyrosomus japonicus*) and spotted grunter (*Pomadasys commersonnii*) are dependant on the REI region of estuaries, then estuaries have to be managed to ensure that this region is not reduced or impacted on by excessive freshwater abstraction.
- The results of this study show that the quality and quantity of water entering an estuary influences the composition, abundance and distribution of fishes within that system. Indications are that perennial base flow conditions are conducive to a rich and diverse ichthyofauna in all reaches of an estuary and that total cessation of riverine inputs results in a significant change in the fish assemblages within the headwaters of an estuary. In addition, this study has aluded to conductivity levels in river water being an important factor regulating the use of the freshwater reaches adjacent to estuaries by marine species. The effect that water abstraction or transfer may have on the conductivity levels of water entering estuaries must therefore be considered when managing the resource.

Conclusions

Prior to the initiation of this multi-disciplinary and multi-institutional research programme, the influence of river flow rates on biotic and abiotic components in the river-estuary interface zone was poorly understood. Permanently open Eastern Cape estuaries were the focus of this study because the catchments of these systems are a primary freshwater source when it comes to supplying both urban and rural users. A Water Research Commission sponsored research programme has since been initiated to examine the REI region of temporarily open/closed estuaries along the South African coast.

For the purposes of the Eastern Cape REI study, the following primary conclusions emerged:

1. Elevated river flow rates into the Gamtoos Estuary increased the size of the REI zone both longitudinally and in terms of volume. A flow rate of between 0.8 and $1.2 \text{ m}^3 \text{ s}^{-1}$ produced a maximum phytoplankton biomass in the estuary but there was no clear relationship between measurable nutrient content in the water column and phytoplankton biomass.
2. River flow rate and the size of the REI were shown to have a major effect on the distribution and community structure of aquatic invertebrates. Benthic invertebrates of the Gamtoos REI region had a different species composition and abundance to those found in the more saline estuarine reaches. Similarly, estuary-associated fishes in the Great Fish Estuary showed a distinct longitudinal zonation pattern that was related to salinity changes, with the REI region playing an especially important nursery role for certain angling fish species.
3. From the above we can conclude that the maintenance and productivity of the REI zone depends on an adequate supply of river water (measured in terms of both quantity and quality) entering the estuary. In future the management and implementation of the Water Reserve, as required by the National Water Act (No. 36) of 1998, should address not only the requirements of the lower and middle estuarine reaches but also the very important REI region which, in turn, affects the entire system.

TABLE OF CONTENTS

Executive Summary	i
Table of Contents	viii
Chapter 1: Introduction to the study and the global hypothesis	1
Chapter 2: The effects of different run-off conditions on the salinity distributions in the Gamtoos Estuary	7
Summary	7
2.1. Introduction	7
2.2. Model design and calibration	8
2.2.1. The Mike 11 modelling system	8
2.2.1.1 The hydrodynamic module	8
2.2.1.2 Survey data collected after the flood of November 1996	8
2.2.3. Additional field data	9
2.2.3.1 Tidal variations	9
2.2.3.1.1 Water level recordings during neap tide	15
2.2.3.1.2 Water level recordings during spring tide	18
2.2.3.2 River inflow	18
2.2.3.2.1 Simulated Monthly Flow Data for the period 1975 to 1989	18
2.2.3.3 Salinity concentrations at neap tide	18
2.2.3.4 Salinity distributions at spring tide	24
2.2.4. Model calibration	24
2.2.4.1 Tidal variations	24
2.2.4.2 Comparison of water level variations for pre- and post flood conditions	24
2.2.4.3 Salinity distribution	25
2.3. Model simulation results	25
2.3.1. Flow through the mouth	25
2.3.2. Salinity distributions at constant low flow conditions (pre-flood)	30
2.3.2.1 Longitudinal profiles	30
2.3.2.2 Salinity concentration variations with time (pre-flood conditions)	30
2.3.3. Comparison of salinity distributions for pre- and post-flood conditions	30
2.3.3.1 Salinity distributions at different constant low river flows	30
2.3.3.2 Dry year simulations for pre- and post-flood conditions	41
2.3.4. River flow and salinity concentration correlations	42
2.3.5. Monthly averaged salinity distributions in the Gamtoos Estuary	52
2.3.6. Retention times	53
2.3.6.1 Reduction in tracer concentrations at a river flow of $0.1 \text{ m}^3/\text{s}$	64
2.3.6.2 Reduction in tracer concentrations at a river flow of $1.0 \text{ m}^3/\text{s}$	64
2.3.6.3 Reduction in tracer concentrations at a river flow of $5.0 \text{ m}^3/\text{s}$	64

2.3.6.4 General	64
2.4. Conclusions and recommendations.....	64
2.5 Acknowledgements	65
2.6 References	65
Chapter 3: Water column, interstitial and boundary layer nutrient status in and around the river- estuary interface (REI) zone of the Gamtoos Estuary	66
Summary	66
3.1 Introduction	67
3.2 Materials and methods	69
3.2.1 Sampling and analysis	69
3.2.2 Water column.....	69
3.2.3 Bottom waters.....	71
3.2.4 Rainfall and flow	71
3.2.4.1 Water column study.....	71
3.2.4.2 Bottom waters study.....	71
3.2.5 Data analyses	71
3.3 Results	72
3.3.1 Water column.....	72
3.3.1.1 Physico-chemical status	72
3.3.1.2 Nutrient status.....	72
3.3.1.2.1 Under base flow conditions (0.4 to $1.6 \text{ m}^3 \cdot \text{s}^{-1}$)	72
3.3.1.2.2 During the freshet ($8 \text{ m}^3 \cdot \text{s}^{-1}$)	79
3.3.1.2.3 C:N:P ratios.....	79
3.3.1.2.4 Point sources	80
3.3.1.2.5 Conservative / non-conservative behaviour of nutrients	82
3.3.1.2.6 A river-estuarine interface for nutrients.....	87
3.3.1.2.7 Short-term variability in nutrient concentrations.....	89
3.3.1.3 Comparison to other Eastern Cape estuaries	89
3.3.2 Bottom and porewaters.....	89
3.3.2.1 Bottom waters.....	89
3.3.2.2 Sediment	101
3.4 Discussion	110
3.4.1 Water column.....	110
3.4.1.1 Nutrient sources and the REI	110
3.4.1.2 Eutrophication in the Gamtoos Estuary	112
3.4.2 Bottom waters.....	114
3.4.3 Conclusions and management implications	116
3.4.3.1 Water column.....	116
3.4.3.2 Bottom waters.....	117

3.5 Acknowledgements	117
3.6 References	118
Chapter 4: Structure and dynamics of estuarine microalgae in the Gamtoos Estuary in relation to the river-estuary interface (REI) region	122
Summary	122
4.1 General introduction	123
4.2 Effect of river flow on microalgal biomass and distribution in the Gamtoos Estuary	124
4.2.1 Introduction	124
4.2.2 Study area	124
4.2.3 Materials and methods	126
4.2.3.1 Flow	126
4.2.3.2 Sampling	128
4.2.3.3 Light attenuation	128
4.2.3.4 Nitrate - N	128
4.2.3.5 Phytoplankton chlorophyll-a	129
4.2.3.6 Plotting	129
4.2.3.7 Benthic chlorophyll-a	129
4.2.4 Results	130
4.2.5 Discussion	138
4.2.6 Conclusions	140
4.3 River flow rate and the distribution of phytoplankton groups in the Gamtoos Estuary	141
4.3.1 Introduction	141
4.3.2 Materials and methods	142
4.3.3 Results	143
4.3.3.1 Salinity and nitrate	143
4.3.3.2 Phytoplankton group structure	143
4.3.3.2.1 Average flow: $1.0 \text{ m}^3 \cdot \text{s}^{-1}$	143
4.3.3.2.2 Flow: $0.3 \text{ m}^3 \cdot \text{s}^{-1}$	149
4.3.3.2.3 Flow $1.2 \text{ m}^3 \cdot \text{s}^{-1}$	150
4.3.3.2.4 Flow $2.3 \text{ m}^3 \cdot \text{s}^{-1}$	151
4.3.3.2.5 Flow $9.7 \text{ m}^3 \cdot \text{s}^{-1}$	153
4.3.3.2.6 Flow $30.5 \text{ m}^3 \cdot \text{s}^{-1}$	154
4.3.3.4 Discussion	157
4.3.3.5 Conclusions	161
4.4 Influence of the Gamtoos REI on intertidal and subtidal microphytobenthic communities	162
4.4.1 Introduction	162
4.4.2 Materials and methods	162
4.4.2.1 Experimental design	162
4.4.2.2 Sampling and analysis	163

4.4.2.3 Microphytobenthos.....	163
4.4.2.4 Sediment characteristics.....	164
4.4.2.5 Water column.....	165
4.4.3 Results.....	165
4.4.3.1 Physico-chemical.....	165
4.4.3.2 Sediment characteristics.....	172
4.4.3.3 Biological characteristics.....	173
4.4.4 Discussion.....	175
4.5 General conclusions.....	177
4.6 Acknowledgements.....	179
4.7 References.....	179
Chapter 5: Structure and dynamics of the invertebrate component within the river-estuary interface (REI) region of the Gamtoos Estuary.....	183
General summary.....	183
5.1. Faunistic attributes and determinants of the REI region - macrobenthos and zooplankton.....	186
5.1.1 Summary.....	186
5.1.2. Introduction.....	186
5.1.3. Methods.....	187
5.1.3.1. Study area.....	187
5.1.3.2. Invertebrate sampling.....	188
5.1.3.2.1. Macrobenthos.....	188
5.1.3.2.2. Zooplankton.....	189
5.1.3.3. Data analysis.....	189
5.1.4. Results.....	190
5.1.4.1. Physico-chemical characteristics - salinity regimes.....	190
5.1.4.2. Longitudinal gradients in invertebrate density and diversity.....	190
5.1.4.3. Longitudinal patterns of invertebrate community structure.....	190
5.1.5. Discussion.....	201
5.2. Integrating functional aspects and food-web properties of the REI region.....	202
5.2.1. Summary.....	202
5.2.2. Introduction.....	203
5.2.3. Methods.....	203
5.2.4. Results.....	205
5.2.4.1. Spatial trends in functional group composition.....	205
5.2.4.2. Extending the functional characterisation of the REI.....	208
5.2.5. Discussion.....	212
5.3. Variation of food web processes across estuarine salinity gradients - implications for the ecological characterisation of the REI region.....	215
5.3.1. Summary.....	215

5.3.2. Introduction	215
5.3.3. Methods	216
5.3.4. Results	218
5.3.4.1. Primary food sources - isotopic composition.....	218
5.3.4.2. Invertebrate consumers - isotopic composition.....	218
5.3.4.3. Carbon isotope variation across the estuarine salinity gradient	222
5.3.4.4. Salinity dependence of carbon isotope ratios and trophic pathways.....	222
5.3.4.5. Estuarine food web structure - the importance of salinity regimes	228
5.3.5. Discussion.....	228
5.3.5.1. Salinity as a co-variate in trophic analysis of estuarine systems	228
5.3.5.2. Carbon isotope gradients - evidence for localised food webs	230
5.4. Boundary conditions and carbon flows in low-salinity estuarine regions	233
5.4.1. Summary.....	233
5.4.2. Introduction	234
5.4.2.1. Objective 1: Boundary definitions of low-salinity estuarine regions	234
5.4.2.2. Objective 2: Carbon pathways to estuarine fish.....	235
5.4.3. Methods	235
5.4.3.1. Characterisation of boundaries of low-salinity regions	235
5.4.3.2. Food web analysis.....	237
5.4.4. Results	239
5.4.4.1. Faunal salinity ranges	239
5.4.4.2. Carbon pathways to estuarine fish.....	246
5.4.5. Discussion.....	251
5.4.5.1. Biologically defined salinity zones.....	251
5.4.5.2. Evaluation of boundaries of low-salinity regions	255
5.4.5.3. Food web structures at higher trophic levels.....	255
5.5 Acknowledgements	256
5.6 References	256
 Chapter 6: An investigation into the structure and functioning of the ichthyofauna associated with the REI zone in two Eastern Cape estuaries with differing freshwater inputs.....	260
General summary	260
General introduction	261
6.1 Distribution patterns of fishes in a freshwater rich estuary, with particular emphasis on the REI region.....	262
6.1.1 Summary.....	262
6.1.2 Introduction	262
6.1.3 Study area.....	262
6.1.4 Materials and methods.....	263
6.1.4.1 Fish sampling.....	263

6.1.4.2 Physico-chemical sampling	264
6.1.4.3 Statistical methods	264
6.1.5 Results and discussion	266
6.1.5.1 Physical parameters.....	266
6.1.5.2 Small seine net fish assemblage.....	266
6.1.5.2.1 Species analyses.....	266
6.1.5.2.2 Life history analysis	268
6.1.5.3 Large seine net fish assemblage	271
6.1.5.3.1 Species analyses.....	272
6.1.5.3.2 Life history analysis	278
6.1.5.4 Gill net fish assemblage.....	281
6.1.5.5 Salinity and fish distribution	282
6.2 Distribution patterns of fishes in a freshwater deprived estuary, with particular emphasis on the geographical headwater region	284
6.2.1 Summary.....	284
6.2.2 Introduction	284
6.2.3 Study area.....	285
6.2.4 Materials and methods.....	285
6.2.4.1 Headwater fish study.....	286
6.2.4.2 Fish distribution study	287
6.2.5 Results and discussion	287
6.2.5.1 Headwater fish study.....	287
6.2.5.1.1 Composition and abundance	287
6.2.5.1.2 Life history analysis	288
6.2.5.1.3 Seasonality	288
6.2.5.2 Fish distribution study	288
6.2.5.2.1 Physico-chemical study	288
6.2.5.2.2 Composition and abundance.....	289
6.3 Freshwater flow regime and and fish assemblages in the Great Fish REI region	296
6.3.1 Summary.....	296
6.3.2 Introduction	296
6.3.3 Study area.....	296
6.3.4 Material and methods.....	297
6.3.5 Results and discussion	299
6.3.5.1 Physical parameters.....	299
6.3.5.2 River flow volumes and fish response	299
6.4 Acknowledgements	305
6.5 References	305
Glossary	310

Chapter 1: Introduction to the study and the global hypothesis

The South African National Water Act (No. 36) of 1998 was promulgated in order to regulate the use of the country's water resources. The scope of this legislation probably makes it one of the finest pieces of environmental legislation in South Africa and possibly the most advanced water legislation in the world today. The reason for this is that the legislation was designed to recognize and protect all national water as a resource rather than as a user of water. The implication of this is that since these water sources are a declared resource, they have a very high priority with respect to the amount of water allocated to them for their continued ecological functioning.

South Africa is a signatory to a number of international agreements that have as their purpose, sustaining the environment in a condition that will ensure its sound ecological function in perpetuity. Hence, the National Water Act has the backing not only of internal legislation but also of international treaties that aim to ensure the water supplies of the country remain in as good a condition as it is possible to achieve.

The Water Act refers to rivers, lakes, groundwater, wetlands and estuaries. While there are several types of estuaries in South Africa, only permanently open systems are specifically relevant to the study reported here. This group of estuaries has received more attention because they mostly occur on the larger catchment systems that yield more water and have therefore been the focus of extensive water harvesting for human use.

Observations made in a number of estuaries (e.g. Hilmer and Bate 1990) have indicated that certain portions of estuaries appear to be more biologically active than others. Hence, river water flows into an estuary at the head and ultimately into the sea through the mouth. This produces a gradient of salinity from fresh water at the head to seawater at the mouth. Between the head and the mouth, there is a longitudinal salinity gradient. Within this gradient, which changes according to river flow and tidal regime, many researchers have observed areas that appeared to be biologically richer, more productive and diverse than other sections of the same estuary. This raised the question as to how salinity influences biological activity and whether salinity *per se* is the determining factor, or whether it is only correlated with the high activity and diversity. This resulted in the identification by members of the Consortium for Estuarine Research and Management (CERM) of an area known as the river-estuary interface (REI) region.

Estuaries receive river water as well as seawater, i.e. flow in two directions. River flow is referred to as base flow when it originates from inland springs and groundwater flow. The flow is called a freshet when rain falls inland and run-off into the river channel raises the volume of water temporarily. The flow is referred to as a flood when a considerable amount of water enters a river or estuary following very high rainfall events, e.g. such as occur only once or twice per year to only once in 50 or more years.

Water flowing out of the mouth on the ebb tide carries sediment out to sea, while seawater flowing into the mouth on the flood tide carries marine sediments into the estuary. Waves at sea stir up marine sediments near the mouth with the result that seawater flowing into the mouth carries more sediment into the estuary than the ebb flow carries out to sea. For this reason, all estuaries have a natural tendency for the mouth to block up with marine sediments unless river floods and freshets remove such accumulated material. The implication is that the rate at which river water flows into an estuary is largely responsible for retaining its specific estuarine characteristics. With water being needed inland to support human needs, managing the estuarine environment translates to ensuring that sufficient water is permitted to flow into estuaries to retain an acceptable ecological status as required by the Water Act.

An acceptable ecological condition is not easy to define. Hence, the Water Act refers to conditions of acceptability determined by interested and affected parties. The latter parties need information on which to base their decisions. At least part of the information is the relationship between natural estuarine conditions compared to present condition. In order to estimate such a relationship, research is needed to identify the cause and effect relationships between river flow and estuary condition. Two of the most important issues arising from this are how does the salinity gradient affect condition and how does mouth status affect the condition?

The purpose of the work reported here is to identify aspects of estuarine ecology that will enable sound decision making with respect to the amount of water required to support natural estuarine functions. This work only aims to represent estuaries that, in their natural condition, are normally open to the sea. The scope of this report is confined to the importance of the REI region since the amount of water required to keep the mouth open is a subsequent function of the amount required to optimally sustain the REI.

At the ecosystem level, estuarine structure and function are strongly dependent on allochthonous events, notably tidal exchange and riverine inflow. Freshwater inputs affect estuaries and adjacent oceans at all basic levels of interaction; physical, chemical and biological. This does not imply that estuarine needs for river flow are uniform, since dynamic fluctuations within the abiotic and biotic environment are a natural feature of all estuaries (Grange *et al.* 2000). Indeed, estuaries are probably the most dynamic of environments in that conditions within them vary more widely, rapidly and frequently than any other aquatic habitat.

On a landscape scale, estuaries are among the world's most striking ecotones by forming the major transition zones between fresh water biomes and the oceans. Scaling spatial analysis down from the landscape level to individual estuaries reveals the prominence of a further two transition zones. On the one hand, the river ecosystems grade into the estuary itself, while on the other hand the most seaward sections of the estuary form a dynamic link with adjacent marine systems. Dissecting patterns of physico-chemical parameters within an estuarine ecosystem even further eventually exposes more distinct habitat zones and associated interface regions. Each of these zones

may be characterised by a typical subset of species, and the identification of such distinct assemblages may be used to advance an approach where estuarine zonation schemes are based on biological grounds (Bulger *et al.* 1993).

Among the various levels of spatial patterning evident for biological communities within estuaries, the transition zone between fresh- and saltwater forms one of the most conspicuous interfaces. The salient features of this section are very steep gradients of salinity, with the amount of freshwater inflow structuring the river-estuary interface (REI) most decisively. Freshwater inflow is not only seen as the major physical force of the REI region, but constitutes a pressing research priority. Against this backdrop, the relevancy of better understanding those estuarine ecological processes where freshwater influence is most prominent becomes clear.

The region where the upper section of an estuary meets the lower riverine reaches appears clearly identifiable by rapid changes in hydrochemistry, particularly the ratios of the major ions and concentrations of the total suspended and dissolved solids (Morris *et al.* 1978; Fauzi and Mantoura 1987). The extent to which steep gradients in the physico-chemical properties of this habitat are matched by marked changes in the make-up of the biological communities inhabiting these transition zones is a topic of this report.

In order to improve confidence and accuracy of predictive capabilities in estuarine management, a major programme objective was to extend our understanding of estuarine dynamics and behaviour to specific estuarine processes. The REI was identified as a region with strong biological and physico-chemical links to both the freshwater and estuarine domain. However, key processes in this region, particularly those related to freshwater inflow and connectance with estuarine reaches downstream, were poorly known and impeded the development of more accurate predictive capabilities in estuarine management.

Although the ecotonal region bordering South African riverine ecosystems appears at first sight to be fairly clearly distinguishable on the grounds of rapid changes in salinity, the extent, structure and function of the biological communities inhabiting these transition zones are at best poorly known. In contrast to this data-poor situation, it is not unreasonable to assume that the REI region forms an essential and crucial element of the estuarine ecosystem at large, and may thus play a critical role for vital processes affecting the estuary as a whole (e.g. fish recruitment, nutrient supply and microalgal primary productivity). Moreover, estuarine conservation efforts often primarily target the biotic components of the system and the lack of adequate biological information would thus not bode well for the development of sound conservation strategies.

A number of schemes are currently in use to classify segments of estuaries into zones with different regimes of tidal energy and salinity (Bulger *et al.* 1993; McLusky 1993). The terminology of 'REI' used here is consistent with the findings of Morris *et al.* (1978) and highlights the need to examine biotic structures and processes in different parts of the estuary without assuming that these regions function in an identical manner.

Early South African estuarine management practises largely considered freshwater requirements in terms of the scouring of accumulated sediments from the mouth region or the evaporative requirements of an estuary. To overcome these narrow perceptions it is necessary to understand the structure and functioning of all reaches in an estuary, particularly the upper reaches which have historically been neglected from a research perspective. The current study was directed at examining the role of the REI region in Eastern Cape estuaries, the associated relationship with freshwater inflow, and the importance of this interface to the character and functioning of the estuary.

Riverine input has long been recognized as a critical component to the functioning of southern African estuaries (e.g. Day *et al.* 1954). The semi-arid climate and low rainfall in South Africa, coupled with high evaporation, result in the region having one of the lowest conversions of rainfall to run-off in the world (Davies and Day 1998). In addition, the rapid growth in South Africa's human population has resulted in a marked increase in the demand for freshwater, and available riverine resources are expected to be fully exploited within the next two decades (Davies and Day, 1986). The future ecological viability of each of South Africa's estuaries depends upon the recognition of the biological consequences of reduced freshwater inflow on the various estuarine sections and therefore the system as a whole.

Initially the interface between the river and estuary was considered to be that part of the ebb and flow region where the vertically averaged salinity is usually < 10 ‰. This definition provided a working guideline for the boundaries of the REI. Perhaps more important than defined salinity boundaries were the processes associated with a typical REI region. The area is usually characterized by strong water movements at certain times, with freshwater often exerting a major influence on the physical and chemical structure of the water column. Water current structures and mixing processes in the REI region determine the stratification characteristics, while sediment types may also differ from the rest of the estuary. This leads to the hypothesis that reduced amounts of freshwater input will alter the physical, chemical and biotic dynamics of the upper estuary, which will in turn affect the structure and functioning of the whole estuary.

The following study provides new information on the importance of the REI region to the estuary and its degree of dependence on freshwater for essential processes. Knowledge gained will lead to a better understanding of the consequences of freshwater attenuation in estuaries.

This investigation had the following primary objectives :

- (i) To establish the importance of the REI region to the functioning of permanently open Eastern Cape estuaries.
- (ii) To assess the role of freshwater, particularly base flows, in maintaining the ebb and flow region of permanently open estuaries.

- (iii) To assess the role of freshwater and its associated nutrients in structuring the plant and animal communities in the REI region of these estuaries.
- (iv) To improve predictive capabilities and decision support for the management of estuaries, particularly in relation to the REI region.

To address the above objectives the following key questions were addressed :

- (i) What are the spatial and temporal ranges/patterns of the interface region in relation to varying amounts of freshwater inflow?
- (ii) What is the nutrient status and composition of the REI community in relation to other parts of the estuary?
- (iii) How does the biotic community and nutrient status of the REI region respond to freshwater pulses of varying magnitude?
- (iv) Does the nutrient status and community of the REI region contribute significantly to vital estuarine processes within the system as a whole?

The above key questions were answered in the context of the following 'global' hypothesis:

"The structure and functioning of the river-estuary interface region is governed primarily by the quality, quantity and supply pattern of freshwater received. Furthermore, the interface region has strong influences on the physico-chemical as well as biological structure and functioning of the entire estuary."

References

- Bulger, A.J., Hayden, B.P., Monaco, M.E., Nelson, D.M. and McCormick-Ray, M.G. 1993. Biologically-based estuarine salinity zones derived from multivariate analysis. *Estuaries* 16: 311-322.
- Davies, B.R. and Day, J.A. 1986. The Biology and Conservation of South Africa's Vanishing Waters. Centre for Extra-Mural Studies, University of Cape Town: 186 pp.
- Davies, B.R. and Day, J.A. 1998. Vanishing Waters. University of Cape Town Press, Cape Town: 487 pp.
- Day, J.H., Millard, N.A.H. and Broekhuysen, G.J. 1954. The ecology of South African estuaries. Part 4: The St Lucia system. *Transactions of the Royal Society of South Africa* 34: 129-156.
- Fauzi, R. and Mantoura, C. 1987. Organic films at the halocline. *Nature* 328: 579-580.
- Grange, N., Whitfield, A.K., De Villiers, C.J. and Allanson, B.R. 2000. The response of two South African estuaries to altered river flow regimes. *Aquatic Conservation: Marine and Freshwater Ecosystems* 10: 155-177.

- Hilmer, T. and Bate.C. 1990. Covariance analysis of chlorophyll distribution in the Sundays River estuary, Eastern Cape. *South African Journal of Aquatic Science* 16(1/2): 37-59.
- McLusky, D.S. 1993. Marine and estuarine gradients - an overview. *Netherlands Journal of Aquatic Ecology* 27: 489-493.
- Morris, A.W., Mantoura, R.F.C., Bale, A.J. and Howland, R.J.M. 1978. Very low salinity regions of estuaries: important sites for chemical and biological reactions. *Nature* 274: 678-680.

Chapter 2: The effects of different run-off conditions on salinity distributions in the Gamtoos Estuary

L. van Niekerk and P. Huizinga

Division of Water, Environment and Forestry Technology, CSIR

Summary

Using the Mike 11 modelling system as a 'tool', the effects of selected run-off conditions on salinity distributions within the Gamtoos Estuary were investigated. This specific task was part of a more comprehensive research programme on the estuary, with the results being used to inform the ecological investigations that were undertaken as part of the river-estuary interface (REI) study.

A large number of model simulations were undertaken to determine the salinity distributions in the Gamtoos Estuary for different river flows. The results are included as graphs for high and low neap and spring tides. These simulations were also undertaken for pre-1996 flood and for post-1996 flood conditions, with significant differences in salinity distributions being found as a result of the scouring effect of these floods. The model results were also used to determine the volumes of water in the estuary below (and between) selected salinity levels of 5, 10, 15, 20, 25, 30 and 35 ‰ for different river flows under pre- and post flood conditions. In this manner the volume of available habitat (based on salinity) for aquatic organisms could be calculated.

2.1 Introduction

This project was undertaken by the Consortium for Estuarine Research and Management (CERM) for the Water Research Commission, to improve the understanding of the interaction between river flow, salinity distribution and the ecology of estuaries. In a joint subproject between the CSIR, the University of Port Elizabeth and the JLB Smith Institute for Ichthyology, a study of the Gamtoos Estuary was undertaken to identify some of the main behavioural responses and parameters related to this interaction.

In this chapter, the work undertaken by the CSIR on the effects of selected run-off conditions on the salinity distributions in the Gamtoos estuary, utilising the MIKE 11 modelling system, is described. The contributions by the CSIR to this study include:

1. Collection of field data for the model calibration.
2. Setting-up of the model of the Gamtoos Estuary on the Mike 11 modelling system.
3. Initial calibration of the hydrodynamics and salinity distributions in the estuary.
4. Undertaking of a limited number of simulations of selected run-off conditions for the estuary. to determine the effects of variation in the flow condition on the salinity distributions.
5. Model simulation comparisons based on the estuary topography before and after the flood, which occurred in November 1996.

6. Undertaking model simulations to estimate retention times for different river flows in the Gamtoos Estuary.
7. Estimating the correlation between river flow and volume of water for selected salinity concentrations within the estuary.
8. Preparation of tables of the distances from the mouth at which selected salinity concentrations occur under different river flows.

2.2 Model design and calibration

2.2.1 The Mike 11 modelling system

The Mike 11 modelling system, which was developed in the 1970s at the Danish Hydraulics Institute. It is an advanced and user-friendly one-dimensional dynamic modelling system for rivers and estuaries (see Slinger *et al.* 1998). The hydrodynamic module is the basic module in Mike 11; additional modules can be used for simulations of transport dispersion, water quality and sediment transport, amongst others. The hydrodynamic and transport-dispersion modules were used in the investigations described in this report.

2.2.1.1 The hydrodynamic module

The hydrodynamic module is based on the solution of the Saint Venant equations of continuity and momentum for one-dimensional flow by implicit finite difference techniques. The model results can be presented in tables or graphs for different locations as time-series of water levels, velocities or flows. They can also be presented as longitudinal variations of water levels or velocities from the mouth to the upstream end of the estuary. Such a model is usually driven by the tidal variation of water levels at the downstream open boundary at the mouth and by river flow at the upstream open boundary condition. The calibration of the hydrodynamic model is done by adjusting the bottom roughness until a satisfactory agreement is reached between simulated and measured water level variations.

2.2.1.2 Survey data collected after the flood of November 1996

New cross-section data were collected in the Gamtoos Estuary in February 1997 after the November 1996 flood as part of the ongoing monitoring programme undertaken by the CSIR for the Department of Environmental Affairs and Tourism (DEAT). A direct comparison between the new data and the data collected in 1985 for the depth and the width along the estuary is made in Table 2.1. The new data were used to modify the Mike 11 Model in preparation for model simulations that would compare water level variations and salinity distributions for pre- and post-flood conditions.

A comparison of the 1985 and 1997 surveys (Figures 2.3 A - D) reveals that the upper reaches of the estuary, more than 10 km from the mouth, experienced extensive scouring during the November 1996 floods. At most upstream cross-sections, the depths of the channel increased by more than one metre. It is difficult to quantify the effects in the lower reaches, because the 1985 and 1997 cross-sections were not all measured at the same positions. Differences in the lower estuary could also be due to the movement of sandbanks within this region.

2.2.3 Additional field data

Additional field data were collected in the form of water level recordings and salinity measurements between 3 November 1996 and 11 November 1996. This collection was done as part of a comprehensive field data collection exercise undertaken by CERM members for the project. The purpose was to obtain information on the hydrodynamics and salinity distributions within the estuary.

2.2.3.1 Tidal variations

Water level recordings were collected at a number of positions in the estuary by visual observation from temporarily installed gauging poles. These recordings were undertaken during a neap tidal cycle on 3 November 1996 (Figure 2.4 A) and during a spring tidal cycle on 11 November 1996 (Figure 2.4 B).

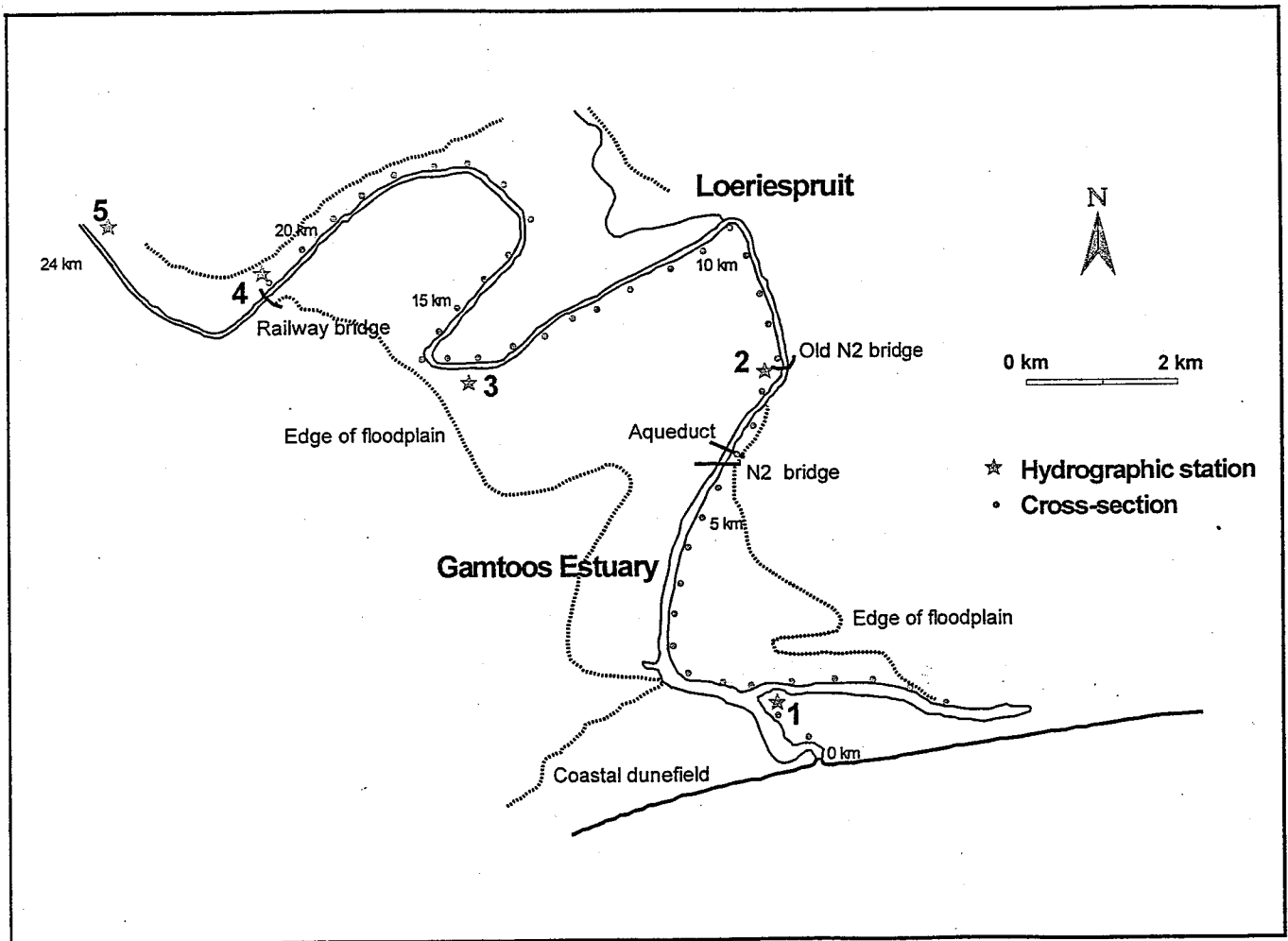


Figure 2.1: Locality map of the Gamtoos Estuary, with positions of cross-sections and hydrographic stations.



May 1978



December 1981

Figure 2.2: Aerial photographs of the estuary mouth – May 1978 and December 1981.

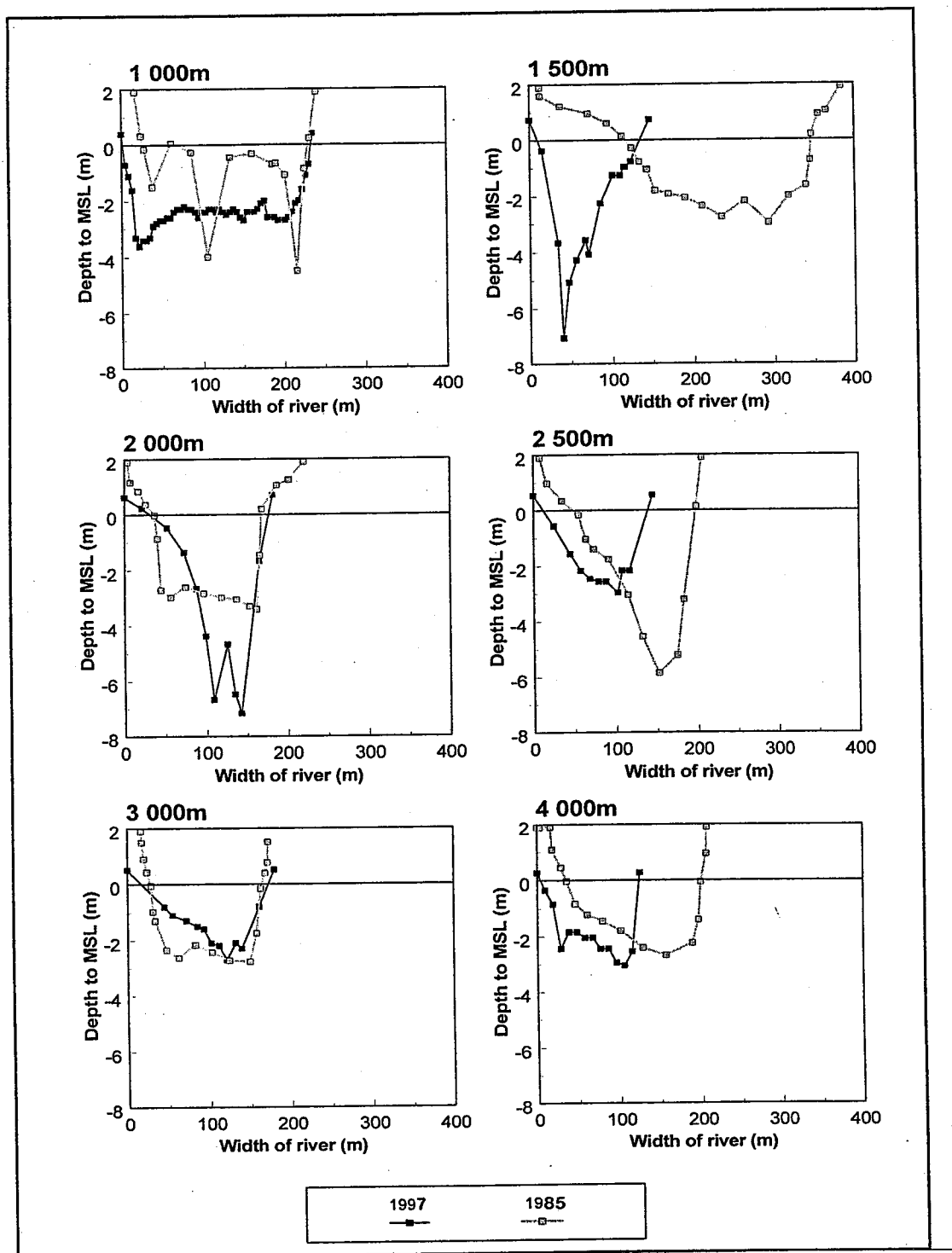


Figure 2.3 A: Comparison of cross-section data between 1985 and 1997 (after November floods).

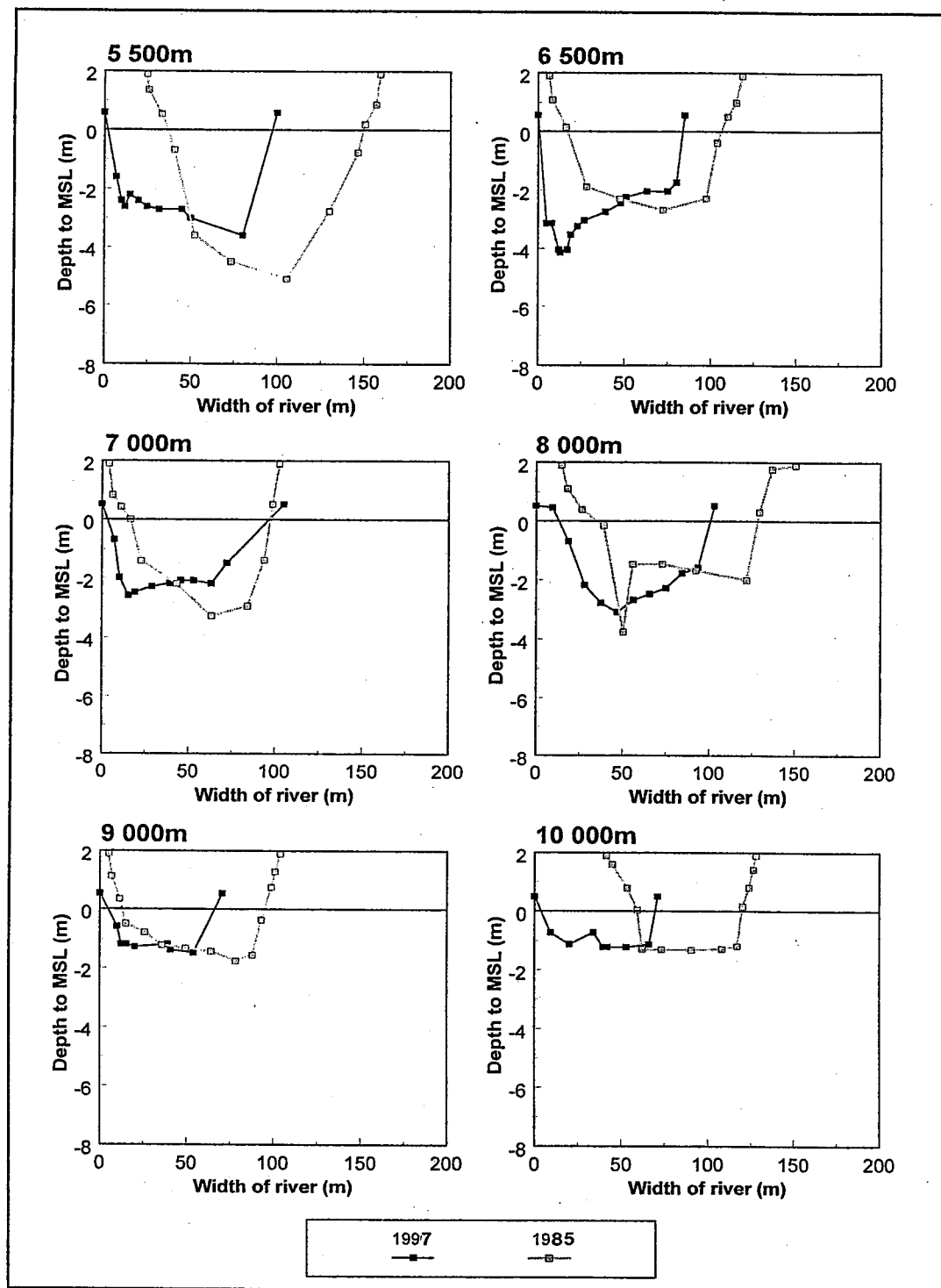


Figure 2.3 B: Comparison of cross-section data between 1985 and 1997 (after November floods).

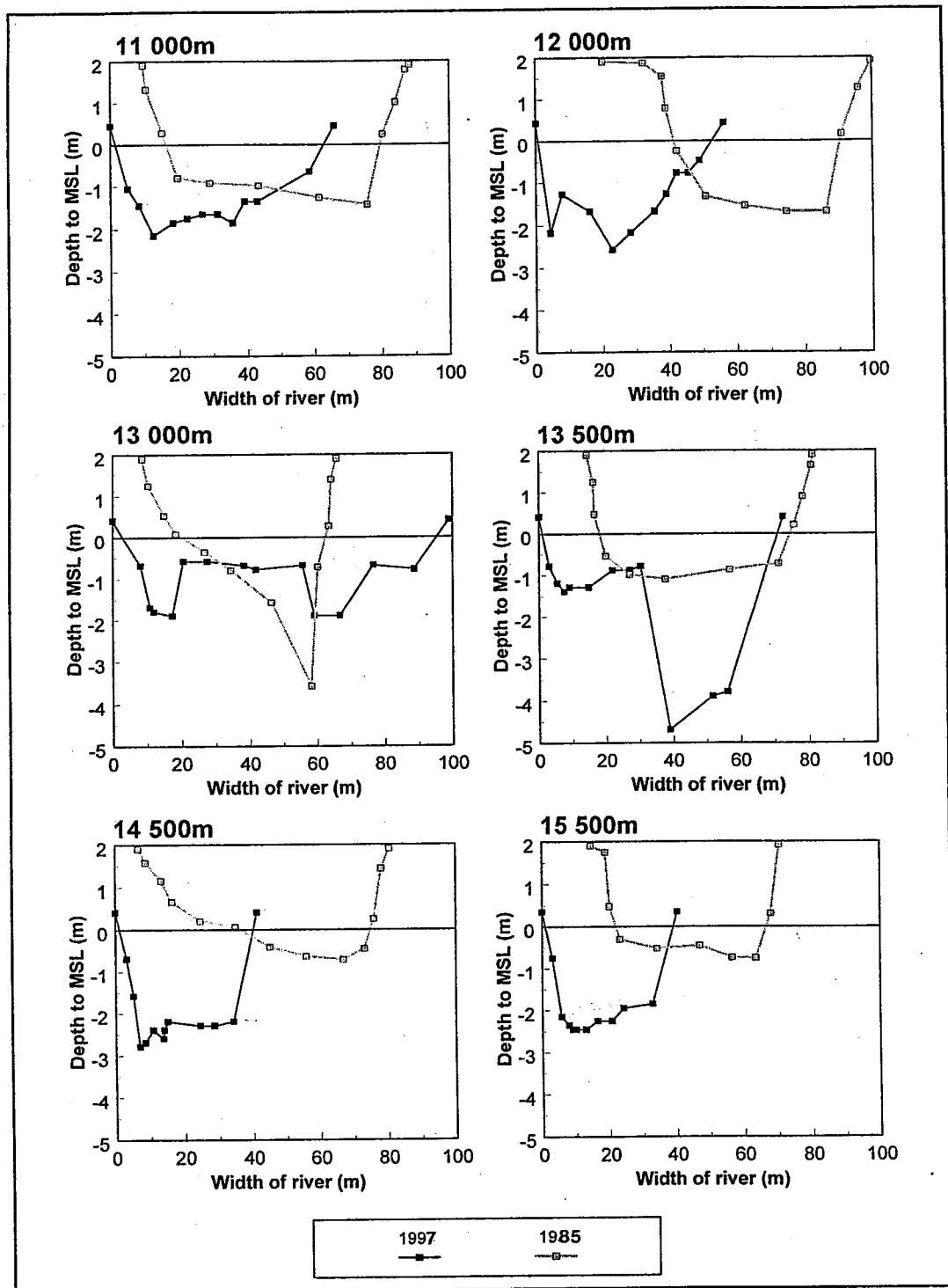


Figure 2.3 C: Comparison of cross-section data between 1985 and 1997 (after November floods).

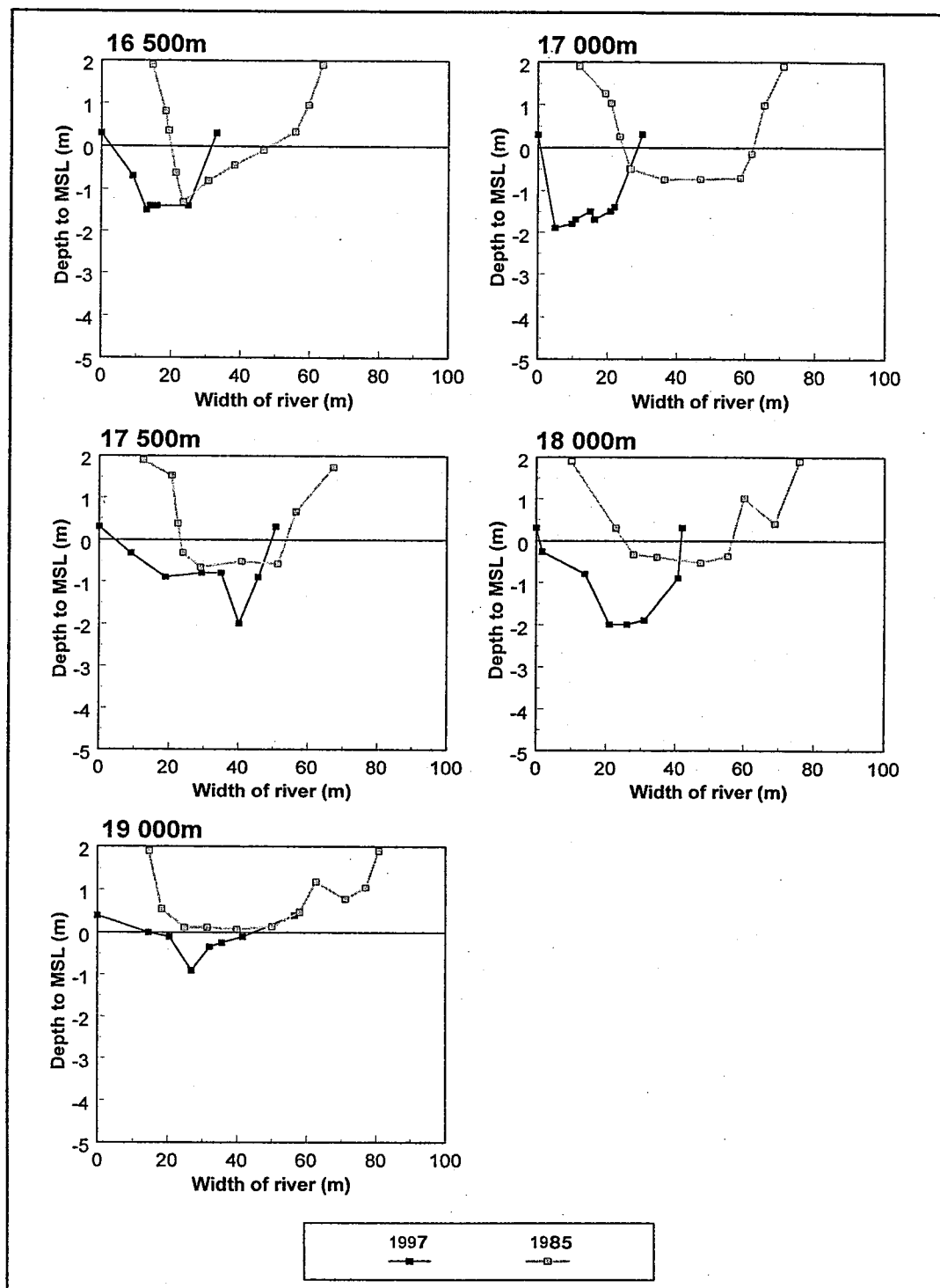


Figure 2.3 D: Comparison of cross-section data between 1985 and 1997 (after November floods).

Table 2.1: Comparison between the cross-sections of the 1997 and 1985 surveys (MSL = mean sea level).

Survey 1997			Survey 1985		
Distance from Mouth (m)	Depth (m)	Width at MSL (m)	Distance from Mouth (m)	Depth (m)	Width at MSL (m)
1 028	-3.61	232	1 000	-4.5	204
1 712	-7.07	125	1 500	-2.97	231
2 014	-7.17	146	2 000	-3.41	132
2 479	-2.96	128	2 500	-5.85	150
3 233	-2.7	156	3 000	-2.77	137
3 941	-3.04	120	4 000	-2.68	164
5 401	-3.61	95	5 500	-5.12	133
6 318	-4.14	83	6 500	-2.68	90
7 238	-2.58	93	7 000	-3.27	81
8 207	-3.08	88	8 000	-3.76	93
8 956	-1.48	61	9 000	-1.77	83
9 989	-1.22	65	10 000	-1.32	61
11 013	-2.15	61	11 000	-1.43	64
11 746	-2.57	53	12 000	-1.68	50
12 892	-1.89	92	13 000	-3.58	42
13 707	-4.69	70	13 500	-1.1	57
14 694	-2.79	40	14 500	-0.72	39
15 652	-2.45	38	15 500	-0.75	45
16 721	-1.49	30	16 500	-1.3	28
17 217	-1.89	28	17 000	-0.74	38
17 835	-1.99	43	17 500	-0.65	30
18 153	-1.99	41	18 000	-0.52	31
19 070	-0.9	30	19 000	0.07	0

2.2.3.1.1 Water level recordings during neap tide

The recordings were compared with the predicted sea tidal variation at Port Elizabeth and relative vertical corrections were applied. Normally, accuracies of < 0.02 m can be obtained for neap tidal variations. Figure 2.4 A shows that between the sea and Station 1 (inside the mouth) the tidal variation was reduced by approximately 30.0 %, with only limited further reduction occurring between Stations 1 to 4. Due to the fact that there were still surprisingly large tidal fluctuations upstream at Station 4 (20.5 km) on 3 November 1996 at neap tide, it was decided to add an extra station (Station 5 - see Figure 2.1) 24 km upstream for the recordings at spring tide on 11 November 1996. Considerable rainfall occurred in the catchment between 3 and 11 November 1996 and while river flow on 3 November 1996 was approximately $1.0 \text{ m}^3 \text{ s}^{-2}$ it was estimated that flow was considerably higher ($8.0 \text{ m}^3 \text{ s}^{-1}$) on 11 November 1996. Information obtained from DWAF indicated that considerably higher flows occurred between 3 and 11 November 1996.

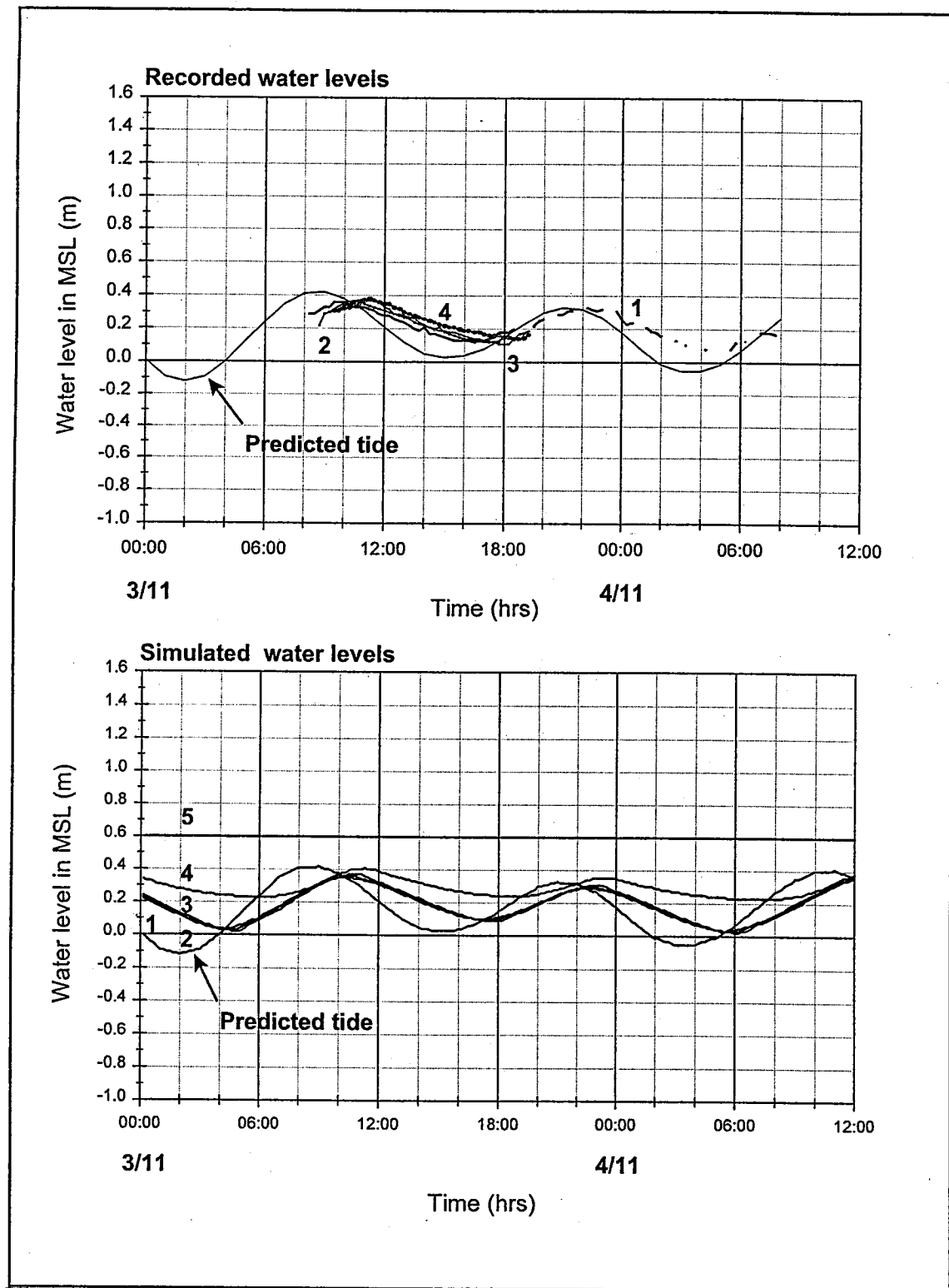


Figure 2.4 A: Recorded and simulated water levels in the estuary at Stations 1 to 5 for neap tide, 3 November 1996, at a flow of $1 \text{ m}^3 \text{ s}^{-1}$.

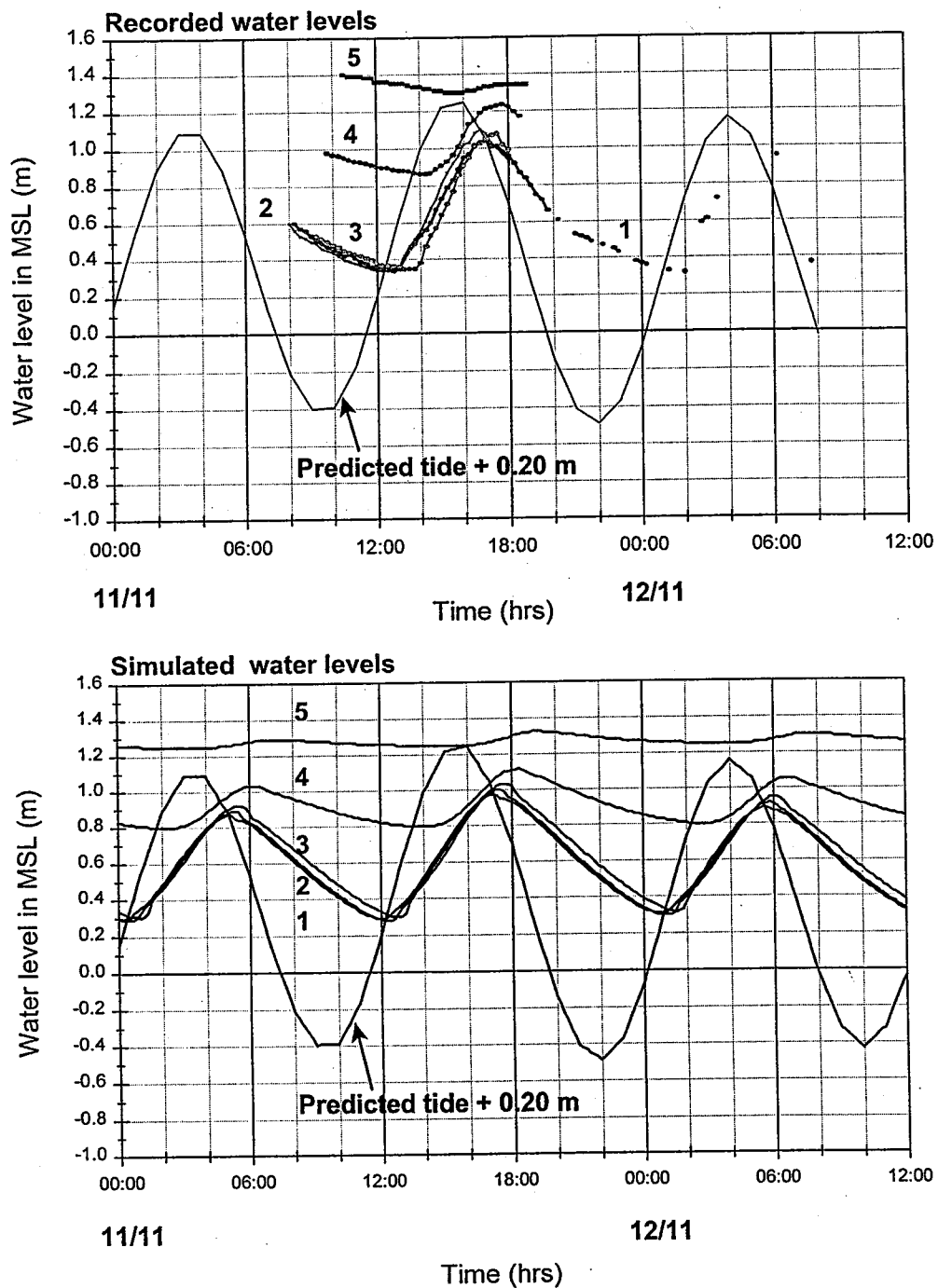


Figure 2.4 B: Recorded and simulated water levels in the estuary at Stations 1 to 5 for spring tide, 11 November 1996, at a flow of $8 \text{ m}^3 \text{ s}^{-1}$.

2.2.3.1.2 Water level recordings during spring tide

The recorded water levels during spring tide are shown in Figure 2.4 B, and are compared with the predicted tidal variation for Port Elizabeth for the same period. The same gauging poles were used at the same positions and at the same heights as during neap tide and therefore the corrections obtained for neap tide could be applied to the data for the spring tide. A comparison of the predicted tide and observed tide at Port Elizabeth Harbour is shown in Figure 2.5.

Again, little difference in water level variation was observed between Stations 1 to 4, while what variations there were, were approximately 50.0 % of the predicted tide for Port Elizabeth. A further reduction was observed for Station 4, but the average height of the water had increased by approximately 0.4 m resulting from the increase in river flow which had been approximately $8.0 \text{ m}^3 \text{ s}^{-1}$ during the day. A small tidal variation of approximately 0.08 m was observed at Station 5 and comparison with model simulation results indicated that the average water level was approximately 0.35 m higher than at Station 4 (Figure 2.4 B).

2.2.3.2 River inflow

Based on the water level recordings, and through the model calibration process, the river flow during the field measurements was estimated at $1.0 \text{ m}^3 \text{ s}^{-1}$ on 3 November 1996 (neap tide) and $8.0 \text{ m}^3 \text{ s}^{-1}$ on 11 November 1996 (spring tide). The difference in river flow between the neap and spring tide is noteworthy because a flow of $1.0 \text{ m}^3 \text{ s}^{-1}$ would not affect the tidal variations, whereas a river flow of $8.0 \text{ m}^3 \text{ s}^{-1}$ would influence the tidal variation significantly. River flow data for the period November 1996 to January 1998 were also obtained from the Department of Water Affairs and Forestry.

2.2.3.2.1 Simulated Monthly Flow Data for the period 1975 to 1989

Monthly river flow data for the estuary were provided by Ninham Shand Consulting Engineers. These data were generated by combining observed monthly flow data at Groot River Poort with simulated run-off data for the catchment downstream of the Groot/Kouga confluence. The combined results for the period 1975 to 1989 (hydrological years, i.e. from 1 October to 30 September) are listed in Table 2.2.

2.2.3.3 Salinity concentrations at neap tide

Longitudinal and vertical distributions of salinities measured in the Gamtoos estuary are presented in Figures 2.5 A to D. These measurements were undertaken at neap tide on 3 November using a Valeport series 600 MK II CTDS meter (accuracy 0.2 ‰). Both the flood and ebb tides were monitored (Figures 2.5 A and B). The upper graph shows the readings taken at the various positions along the estuary and the lower graphs are contour graphs based on the measurements. Only minor differences were observed between the salinity concentrations of the flood tide and the ebb tide. The intrusion of saline water extended until approximately 14.0 km upstream of the mouth. The corresponding river flow was estimated at $1.0 \text{ m}^3 \text{ s}^{-1}$ based on model simulation results. Fairly strong stratification was observed between the mouth and approximately 15.0 km upstream. For example, 8.0 km upstream, salinity values $> 30.0 \text{ ‰}$ were still present in the bottom layer while salinities $< 5.0 \text{ ‰}$ were measured at the surface.

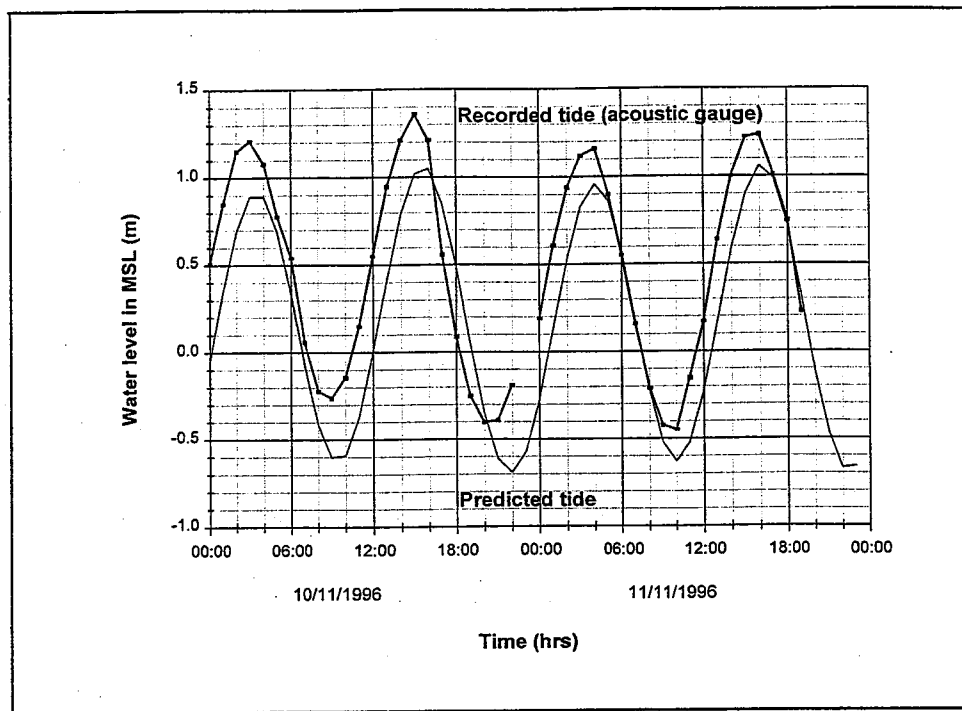


Figure 2.5: Recorded tide versus predicted tide for Port Elizabeth Harbour for spring tide (10 and 11 November 1996).

Table 2.2: Observed flow at Groot River Poort + simulated natural flow downstream of Groot/Kouga confluence ($\text{m}^3 \text{s}^{-1}$).

Year	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Total
1975	5.65	4.22	4.95	2.69	1.4	5.66	2.2	1.27	0.51	3.09	2.45	2.45	36.49
1976	5.82	6.25	1.44	0.94	15	7.2	8.1	36.8	8.41	2.49	2.84	2.79	97.67
1977	3.14	3.36	3.19	1.93	0.3	2.99	4.3	3.81	3.03	2.39	2.55	3.45	34.47
1978	3.11	5.88	3.03	5.39	3.1	1.12	0.8	1.95	2.17	18.54	88.9	23.4	157.4
1979	4.14	1.38	0.54	7.38	2.3	0.69	0.5	0.34	2.35	1.57	1.04	1.28	23.5
1980	5.89	3.09	2.65	6.94	4.5	113	27	22.7	20.6	3.38	14.6	13.1	237.5
1981	20.19	4.83	3.18	1.79	0.3	1.47	11	6.46	1.2	1.69	2.23	1.87	56.59
1982	2.11	0.56	0.71	0.74	0.4	0.82	0.1	0.26	0.73	101	20.5	1.87	129.8
1983	3.14	5.74	2.03	1.1	0.4	3.03	1.3	0.27	0.39	2.06	1.49	1.03	21.96
1984	0.52	0.66	0.29	2.2	5.8	2.57	1.5	0.88	1.24	1.69	1.15	0.54	19.03
1985	3.83	8.64	16.2	2.12	0.6	0.54	0.4	0.2	0.31	0.31	5.77	4.8	43.72
1986	11.3	4.56	0.87	0.35	1.1	0.83	0.9	0.53	1.13	0.86	0.78	2.09	25.3
1987	1.64	0.57	0.3	0.39	0.8	50.9	17	2.62	1.37	0.71	1.53	0.69	78.55
1988	0.81	0.42	0.92	0.62	1.2	0.34	5	2.86	0.85	1.11	1.81	0.78	16.76
1989	16.39	43.4	8.41	0.8	4.2	2.01	0.9	0.7	1.57	1.24	0.56	0.37	80.49
Mean	5.85	6.23	3.25	2.36	2.7	12.9	5.4	5.44	3.06	9.47	9.88	4.03	70.61
Median	3.83	4.22	2.03	1.79	1.2	2.01	1.5	1.27	1.24	1.69	2.23	1.87	24.83

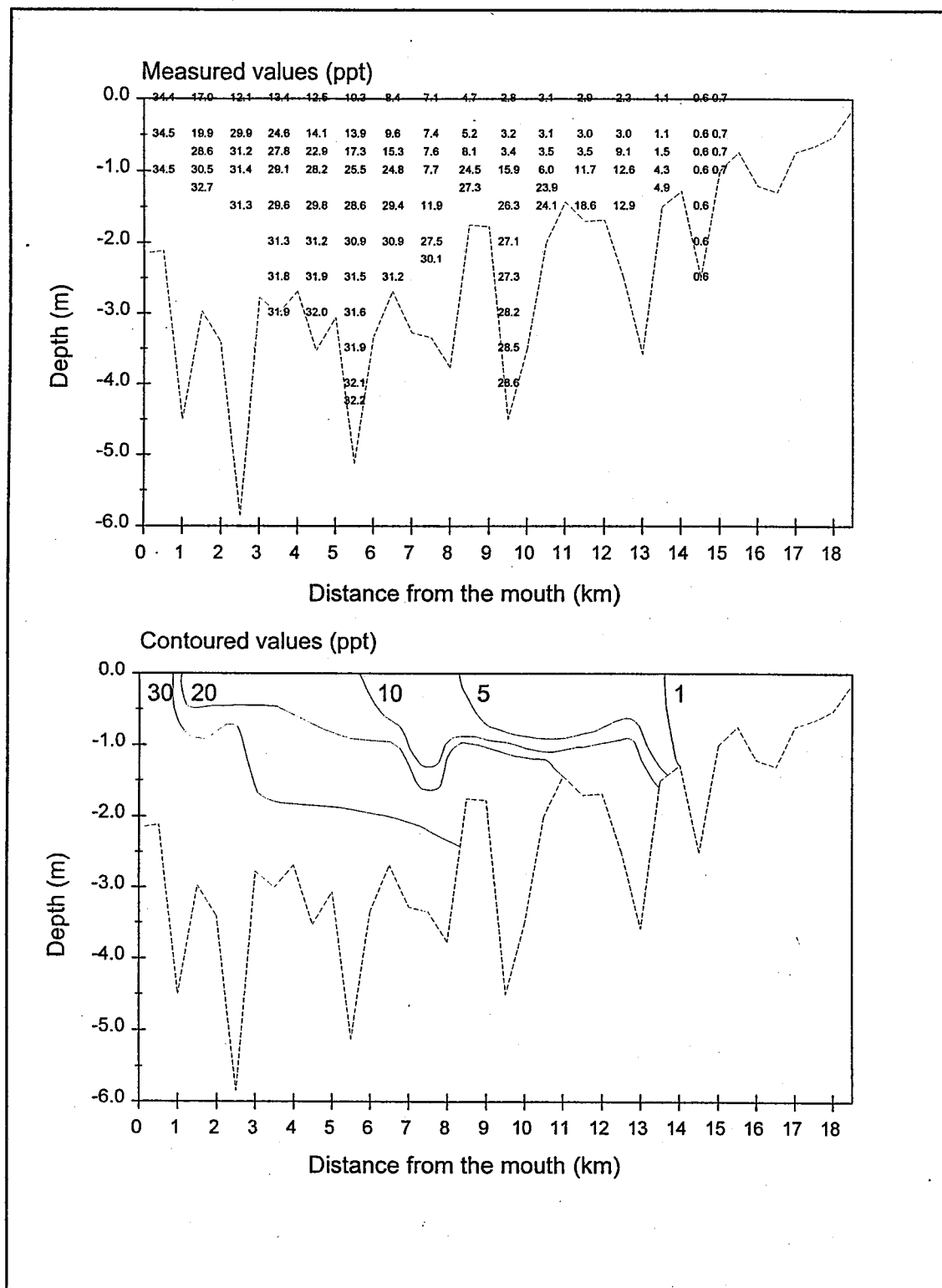


Figure 2.5 A: Longitudinal salinity profile in the Gamtoos Estuary during the neap flood tide, 3 November 1996.

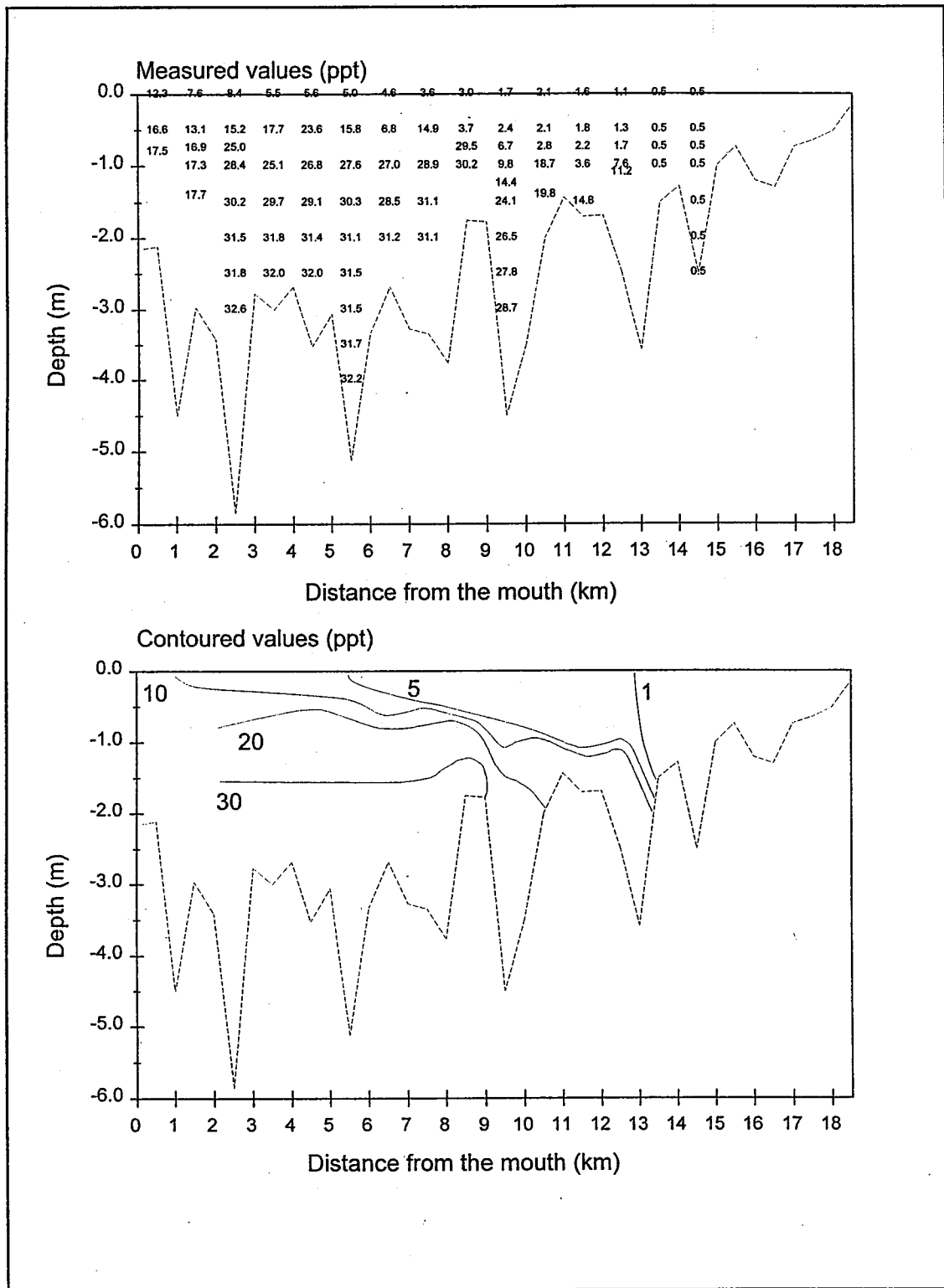


Figure 2.5 B: Longitudinal salinity profile in the Gamtoos Estuary during the neap ebb tide, 3 November 1996.

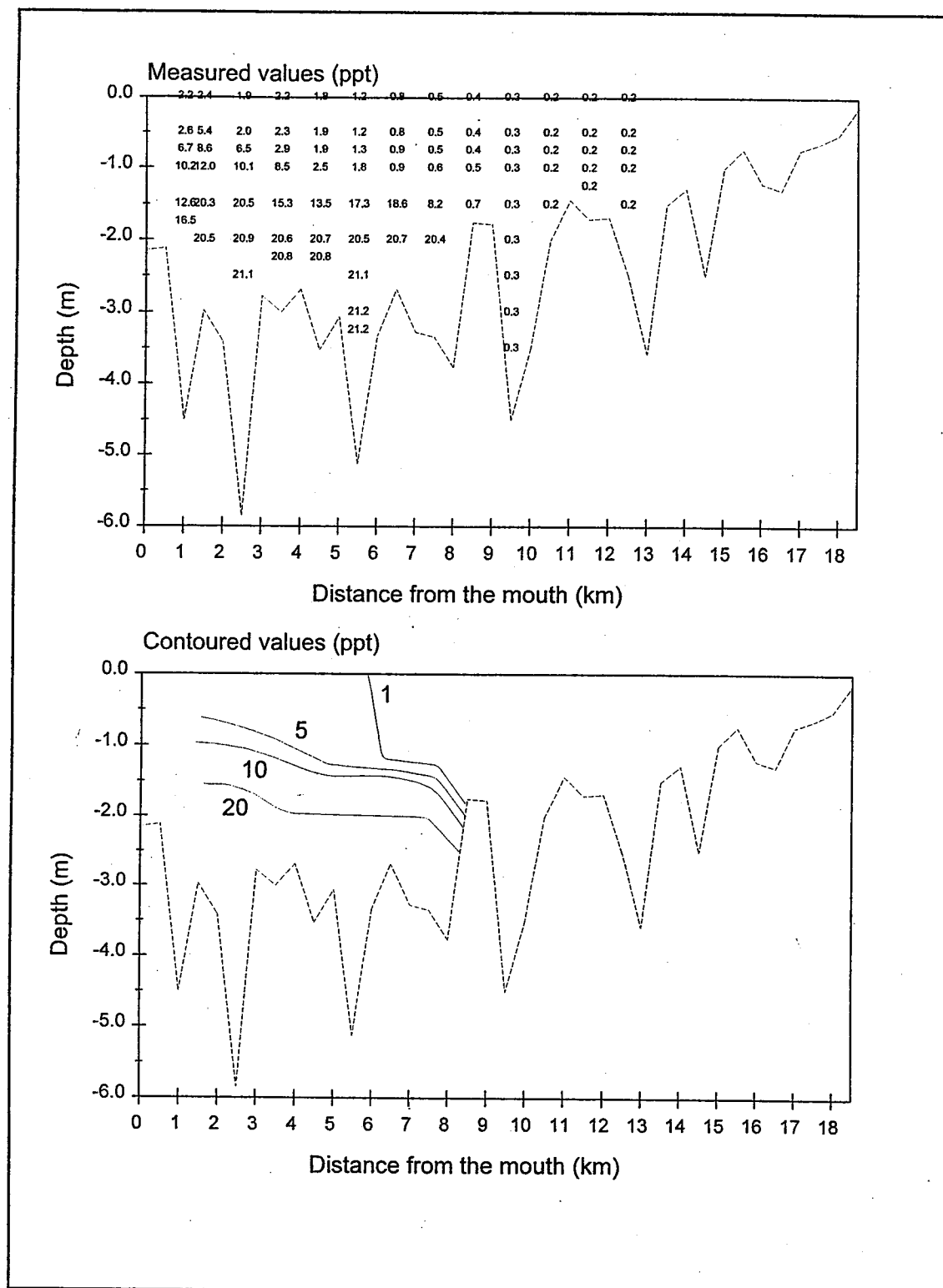


Figure 2.5 C: Longitudinal salinity profile in the Gamtoos Estuary during the spring ebb tide, 11 November 1996.

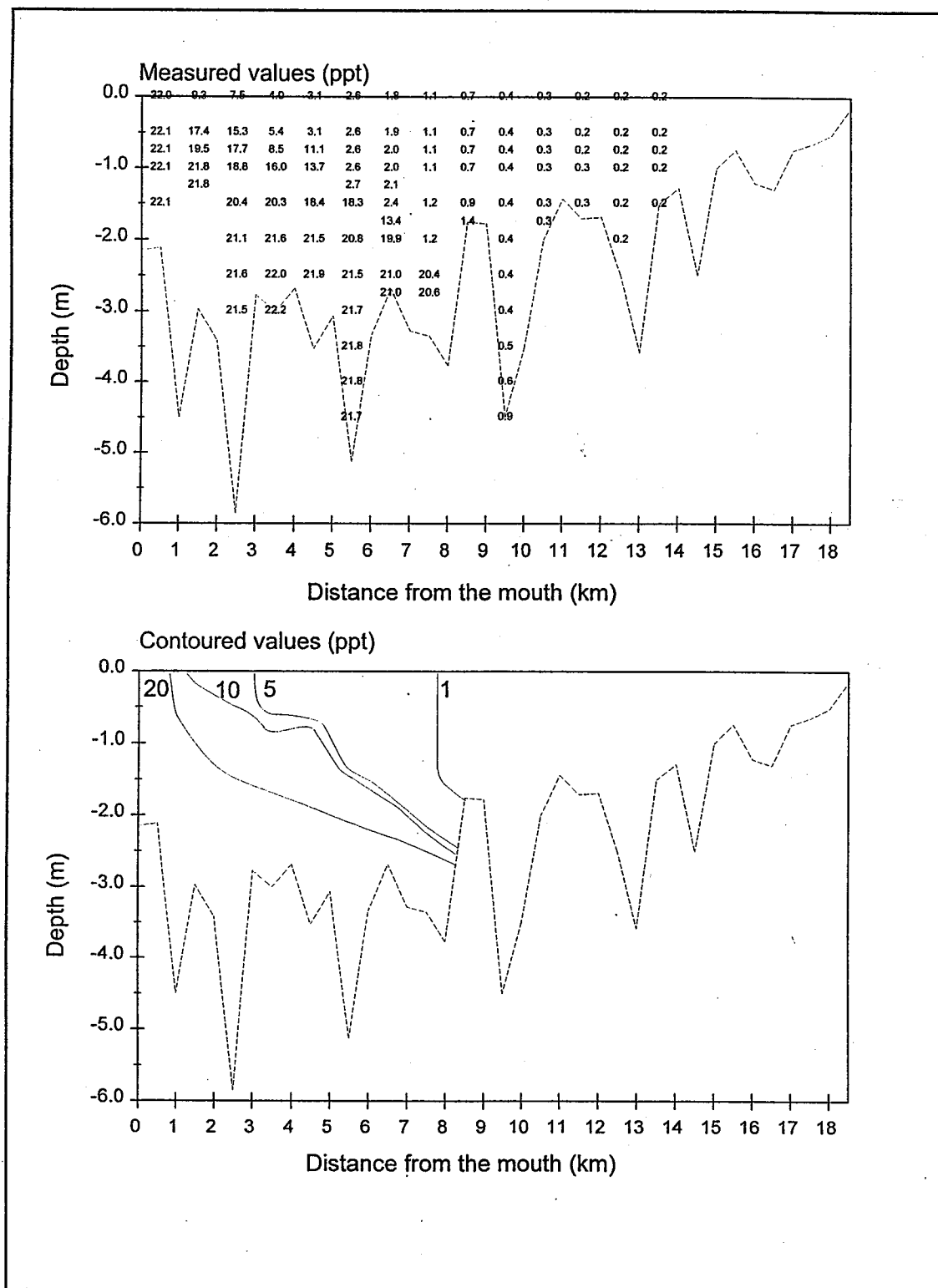


Figure 2.5 D: Longitudinal salinity profile in the estuary during the spring flood tide, 11 November 1996.

2.2.3.4 Salinity distributions at spring tide

The Valeport series 600 MK II CTDS meter used at neap tide was not available for the measurements during spring tide. The salinity measurements at spring tide on 11 November 1996 were therefore undertaken with a WTW model LFG 191 conductivity/temperature meter from the University of Port Elizabeth.

Strong river flow occurred in the catchment between 3 and 11 November 1996. The effects can be observed on the salinity distributions measured on 11 November 1996. The results of the measurements on the flood tide show saline intrusion until approximately 8.0 km upstream of the mouth (Figures 2.5 C and D). These intrusions are reduced on the ebb tide. Increased salinities were observed approximately 8 km upstream on the bottom but only about 5.0 km upstream at the surface. Strong stratification is observed for both profiles, e.g. on the ebb tide 2.5 km from the mouth, salinities of approximately 2.0 ‰ were measured on the surface and salinities of more than 20.0 ‰ were measured at a depth of 2.0 m at the same position. These salinities were measured to obtain data for the calibration of the model. Therefore, a period of fairly constant flow is required between the measurements. As mentioned, a considerable increase in the flow occurred between measurements, making the data less suitable for model calibration (see also Section 2.3.2). However, the data are interesting and are therefore included in the report.

2.2.4 Model calibration

2.2.4.1 Tidal variations

The model was calibrated in terms of water levels using the field data collected on 3 and 11 November 1996. The comparisons are shown in Figures 2.4 A and B. In general, a very acceptable agreement between measured and simulated water level variations was obtained. A minor exception is the reduced water level variation at neap tide for Station 4, which is upstream in the estuary. This variation is probably related to relatively small changes in estuarine topography and was judged to be irrelevant for the calibration of the model. It was also considered that the water level variations for the top part of the estuary would not influence the results of the flows through the mouth or the phases of the tides further downstream.

2.2.4.2 Comparison of water level variations for pre- and post flood conditions

A comparison between simulated water level variations for Stations 1 to 5 before and after the flood is presented in Figure 2.6 A for a neap tide and in Figure 2. 6 B for a spring tide. This comparison shows the effect of scouring of the estuary channel during the flood.

A river flow of $1.0 \text{ m}^3 \text{ s}^{-1}$ was again used for the simulations at neap tide (Figure 2.6 A) for the calibrated period of 3 November 1996. The results show that the simulated water level variations were substantially lower, especially at low tide, at Stations 4 and 5 for the post-flood situation because of the

deepening of the channel upstream in the estuary. A river flow of $8.0 \text{ m}^3 \text{ s}^{-1}$ was again used for the simulations at spring tide. The results show an increase in water level variations (Figure 2.6 B) when compared to those at neap tide, with Station 4 and 5 water levels at low spring tide being substantially lower than those recorded at the same stations during the neap tide.

2.2.4.3 Salinity distribution

To calibrate the model in terms of salinity distribution in the estuary, a relatively constant flow over the period during which the data were collected is required. This was not the case during this period due to the increase in flow rate between 3 and 11 November 1996. A proper calibration using the salinity data was therefore impossible. However, based on expertise obtained during previous model applications, assumptions could be made on the dispersion coefficient in the estuary.

2.3 Model simulation results

2.3.1 Flow through the mouth

Model simulations were undertaken to determine the flows through the mouth for 3 and 4 November 1996 at neap tide and for 11 and 12 November 1996 at spring tide. These results were combined with measurements, undertaken by the University of Port Elizabeth, of fish larvae concentrations in the estuary mouth during the same periods. The results were then used to quantify tidal influences on fish larval movements through the mouth during these periods. Simulated flows through the mouth at 10 minute intervals for neap tide and spring tides were provided to the University of Port Elizabeth. The same results are presented as graphs in Figures 2.7 A and B. The graphs of the tidal variations in the sea and the simulated water levels in the mouth of the estuary have also been added for comparison.

These results show the following maximum tidal flows:

Neap tide	
Maximum flood tidal flow	$26.5 - 36.6 \text{ m}^3 \text{ s}^{-1}$
Maximum ebb tidal flow	$27.2 - 31.7 \text{ m}^3 \text{ s}^{-1}$
Spring tide	
Maximum flood tidal flow	$88.4 - 103.3 \text{ m}^3 \text{ s}^{-1}$
Maximum ebb tidal flow	$69.9 - 76.9 \text{ m}^3 \text{ s}^{-1}$

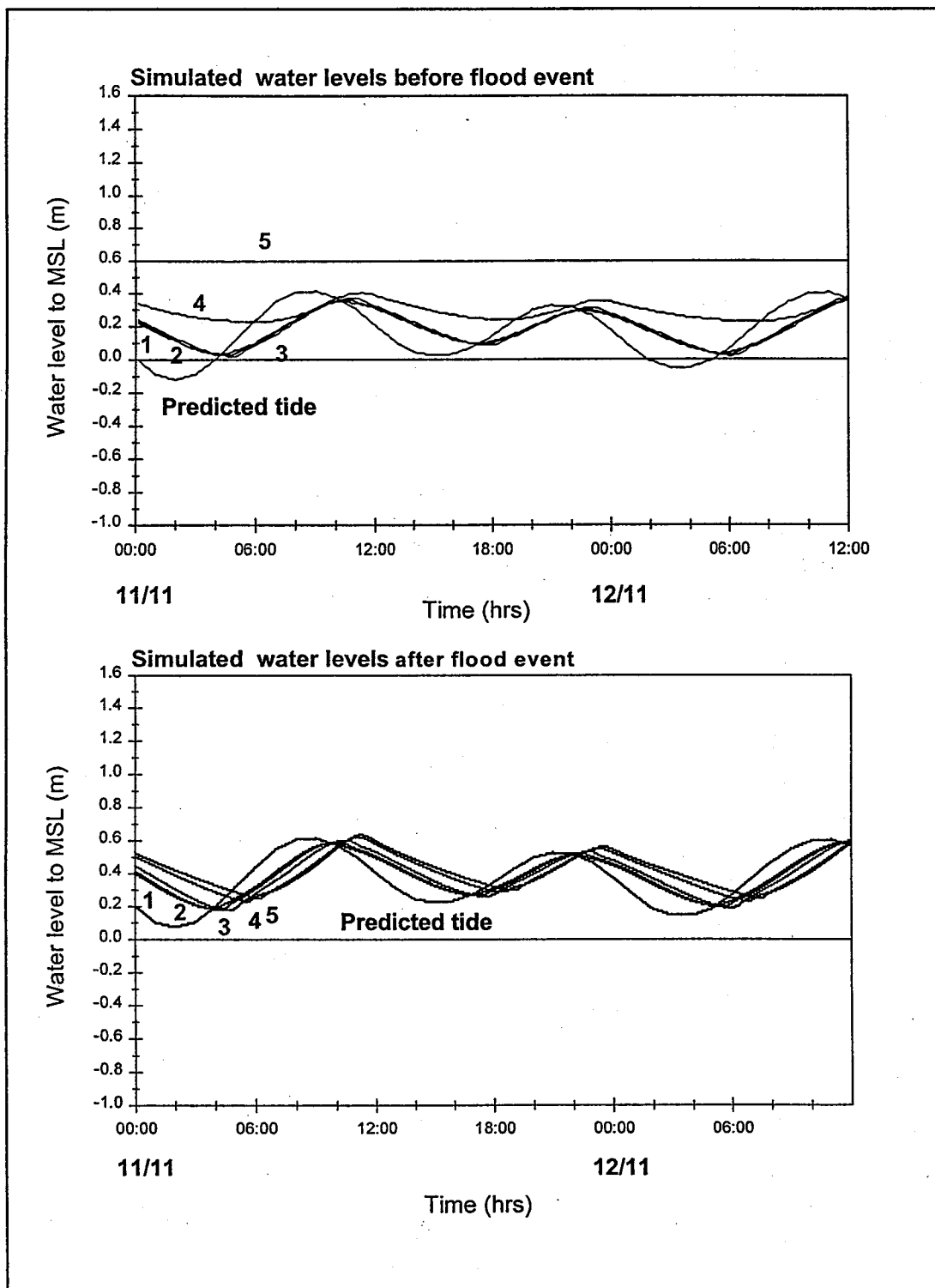


Figure 2.6 A: Simulated neap tide water levels (3 November 1996) at Stations 1 to 5 before and after a flood event (river flow = $1 \text{ m}^3 \text{ s}^{-1}$).

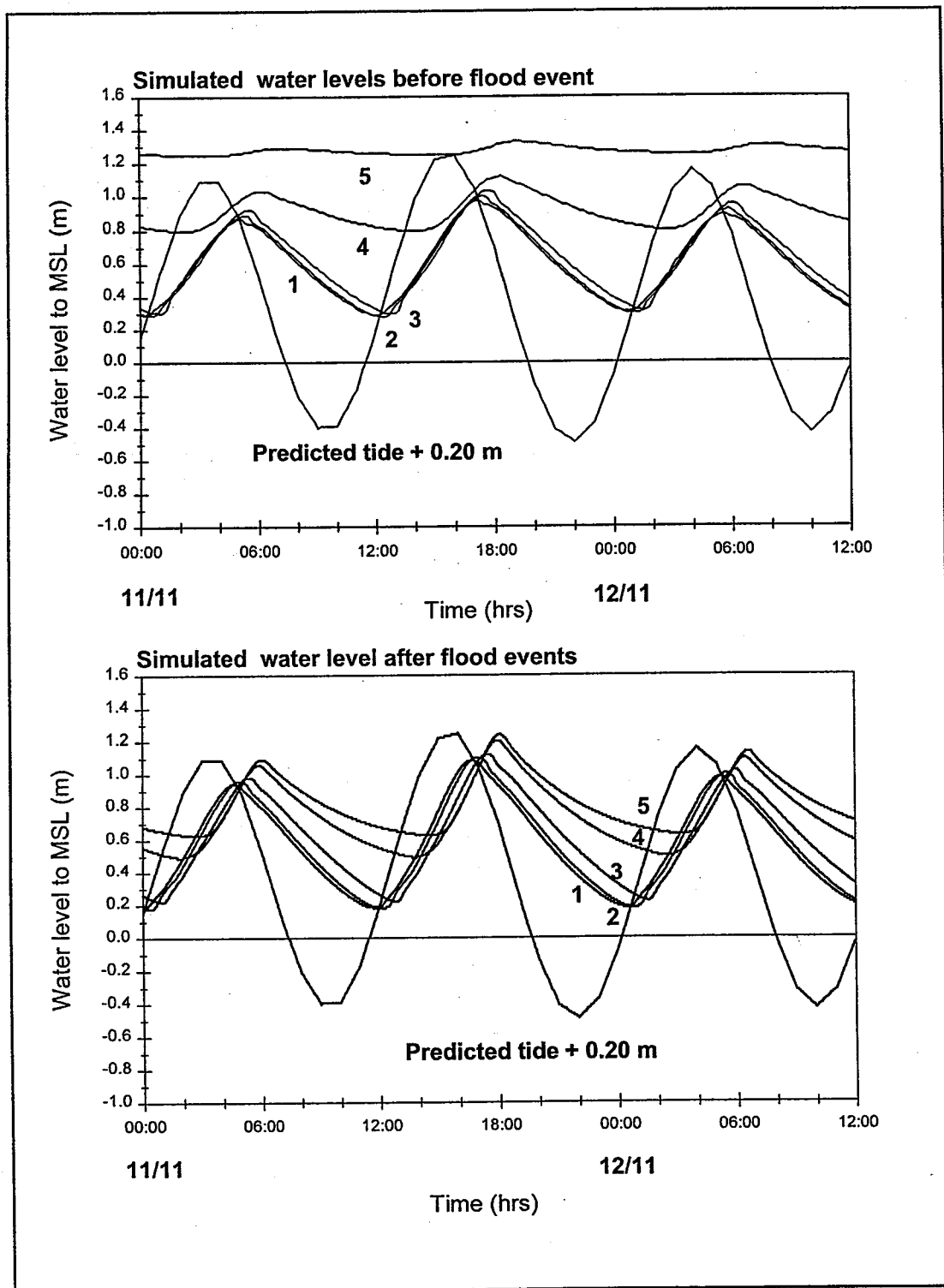


Figure 2.6 B: Simulated neap tide water levels (11 November 1996) at Stations 1 to 5 before and after a flood event (river flow = $8 \text{ m}^3 \text{ s}^{-1}$).

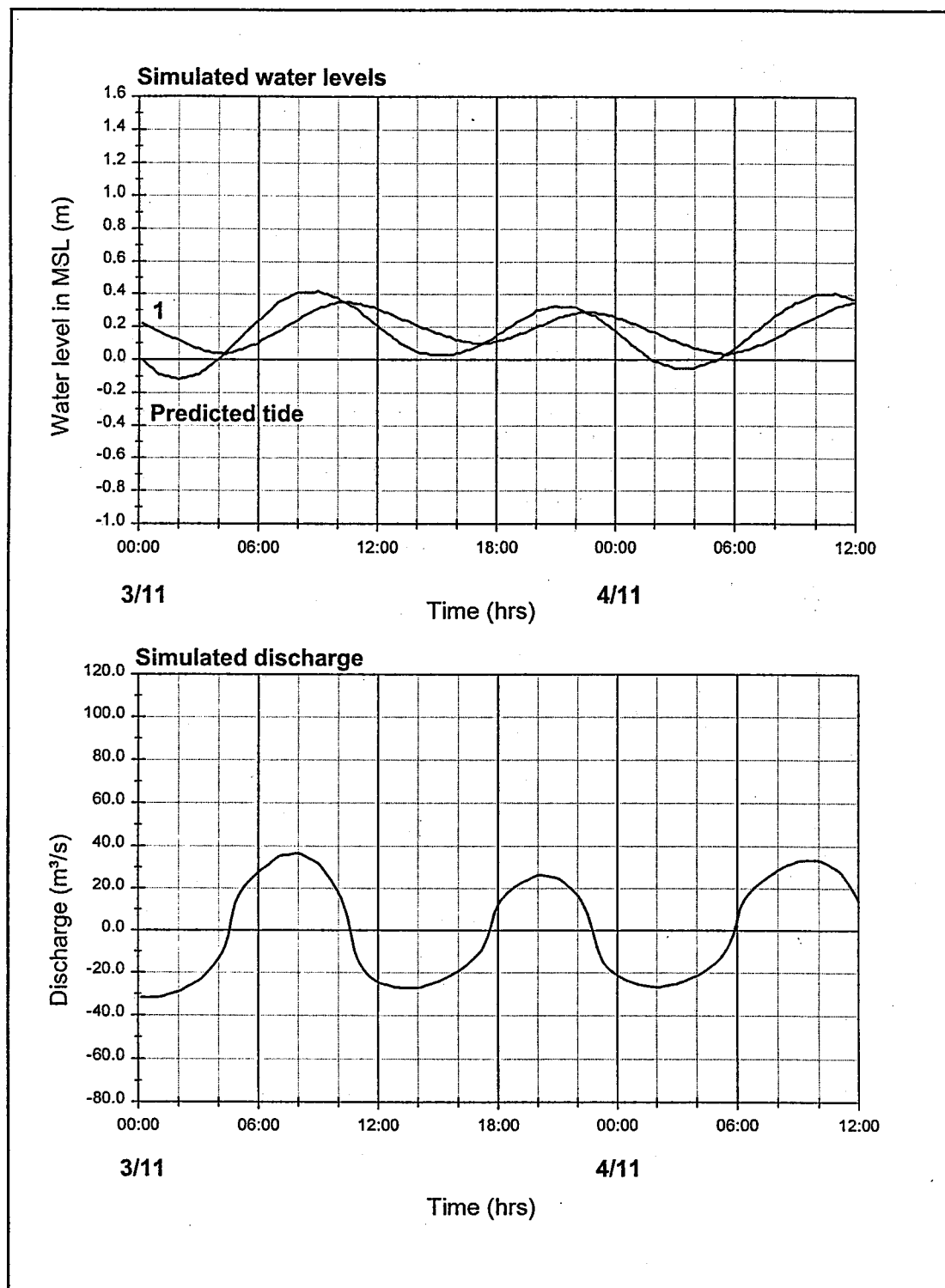


Figure 2.7 A: Simulated neap tide water levels and discharge at Station 1 for 3 November 1996 (river flow = $1 \text{ m}^3 \text{ s}^{-1}$).

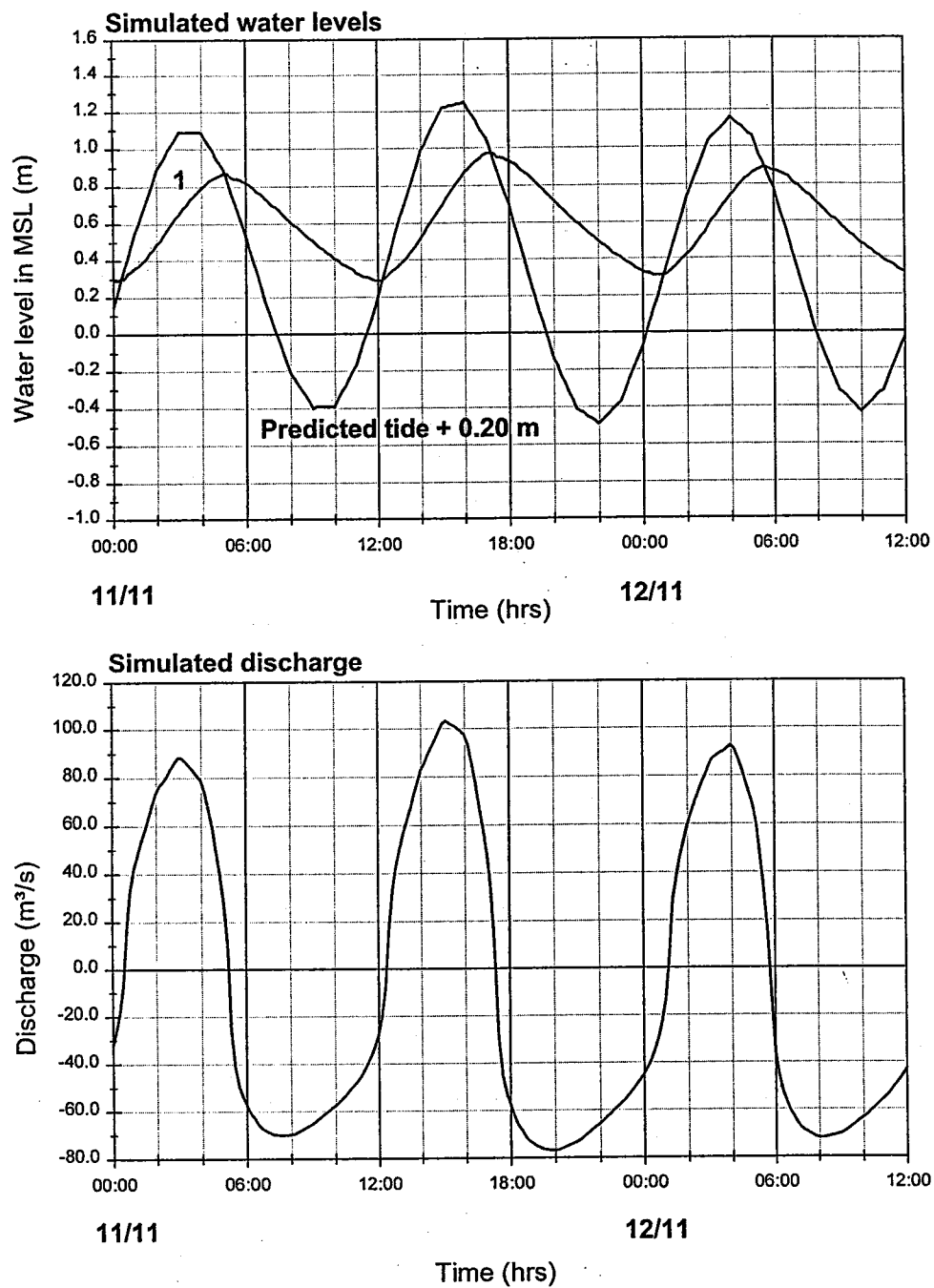


Figure 2.7 B: Simulated spring tide water levels (+0.20 m) and discharge for 11 November 1996 at Station 1 (river flow = $8 \text{ m}^3 \text{ s}^{-1}$).

2.3.2 Salinity distributions at constant low flow conditions (pre-flood)

The effects of a number of low flow conditions on salinity distributions were simulated for the Gamtoos Estuary.

2.3.2.1 Longitudinal profiles

Figures 2.8 A to C present the simulated salinity distributions along the estuary at high water and at low water for a neap tide and for a spring tide under constant flow conditions of 0.1, 0.25, 0.5, 1.0, 5.0, 10.0 and 20.0 m³ s⁻¹.

2.3.2.2 Salinity concentration variations with time (pre-flood conditions)

Figures 2.9 A to G show the variations in salinities with time for the same flow conditions at chainages (distances from the mouth) of 1.0, 7.5, 13.5, 20.5 and 24.0 km. An interesting observation is that even for a river flow of 0.1 m³ s⁻¹, zero salinities are simulated at the upstream end of the estuary and that a full gradient from zero salinity upstream to seawater salinities downstream exists for all the simulated flow conditions. This gradient results from the gradual bed slope upstream in the estuary, where the estuary also becomes very narrow.

Comparing the results of constant flow conditions of 0.1, 0.25, 0.5, 1.0, 5.0, 10.0 and 20.0 m³ s⁻¹, in terms of the longitudinal salinity profiles in Figure 2.8, shows that under all these conditions a salinity gradient will exist in the estuary, but that the salinity profiles are shifted downstream when the flow is increased. All the graphs show the strong influence of even low river flows on the salinity distributions. The results for flows of 5.0 m³ s⁻¹ and higher show that most of the estuary becomes fresh upstream and that sea water intrusion takes place only in the lower 10.0 km of the estuary. This effect is even stronger at higher river flows. During a neap tide, with a flow of 5.0 m³ s⁻¹, salinities are only present in the lower 7.0 km of the estuary and salinity gradients are very steep. At a high river flow of 20.0 m³ s⁻¹ the salinity gradient will exist in only the lower 3.0 km of the estuary during a neap tide. This again indicates that at a flow of 20.0 m³ s⁻¹, the estuary is nearly fresh at low tide. The same effect can also be observed in the time histories of the salinities in Figures 2.9 A to G.

2.3.3 Comparison of salinity distributions for pre- and post-flood conditions

2.3.3.1 Salinity distributions at different constant low river flows

The model simulations described in Section 2.3.2 for different low river flow conditions were repeated for the post-flood topography. The results are presented in Figures 2.10 A to G and these can be compared with those for pre-flood conditions shown in Figures 2.9 A to G. A direct comparison between longitudinal profiles for pre- and post-flood conditions at different constant river flows is presented in Figures 2.11 A to C. These profiles are shown for spring and neap tide high water and low water. All these graphs show increased intrusion of saline water following flood scouring of the estuary. This is most marked in the upper reaches of the estuary and at low flows (0.1 to 0.5 m³ s⁻¹). These graphs can be used directly to hindcast or predict salinity distributions in the estuary for both pre- and post-flood topography conditions.

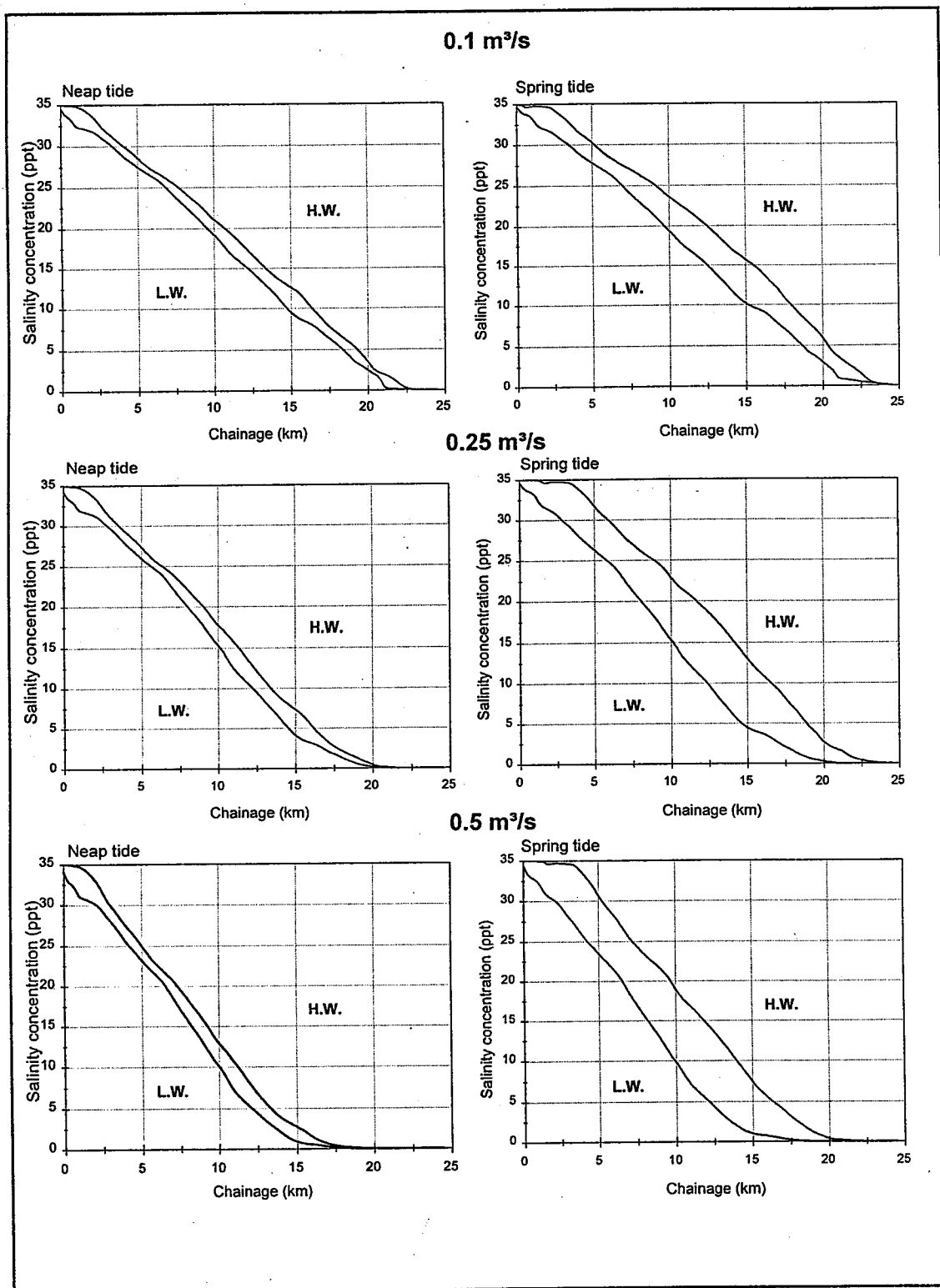


Figure 2.8 A: Simulated salinity distributions for indicated flow conditions.

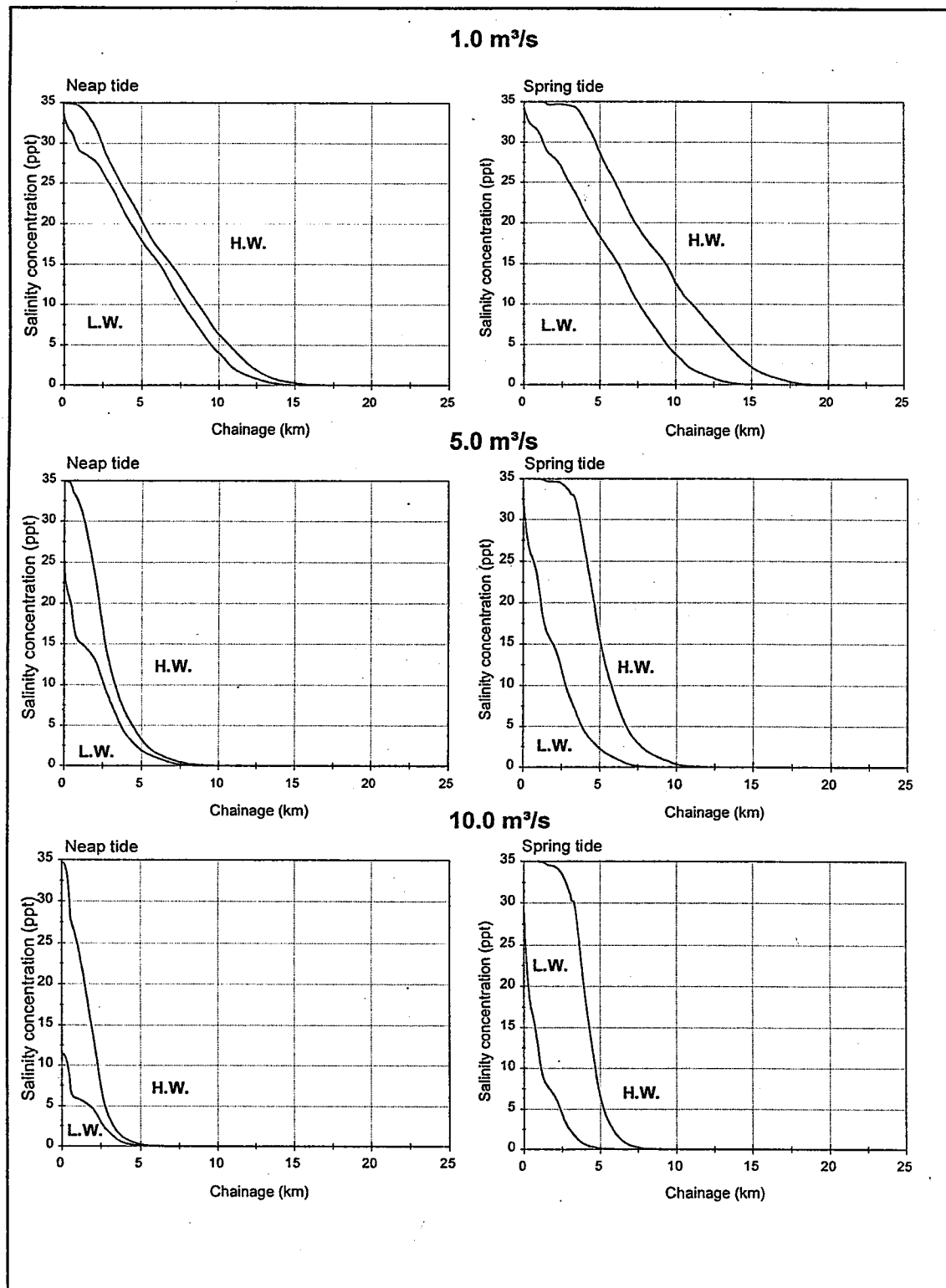


Figure 2.8 B: Simulated salinity distributions for indicated flow conditions.

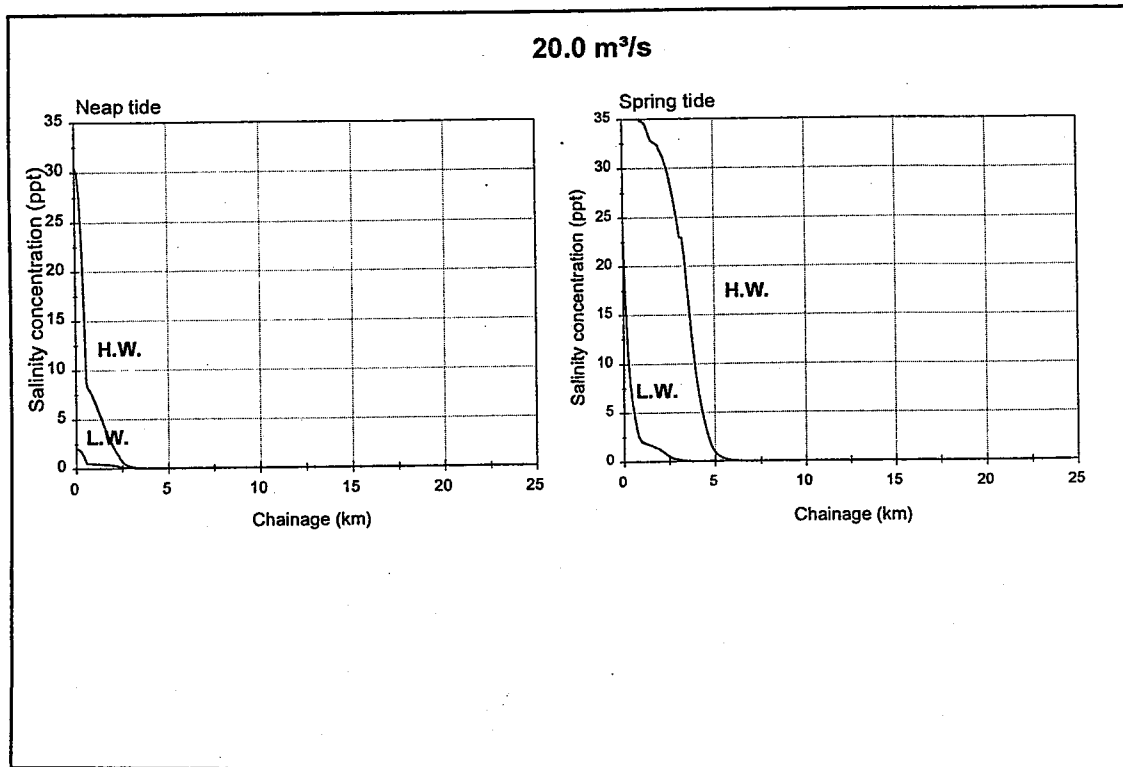


Figure 2.8 C: Simulated salinity distributions for indicated flow conditions.

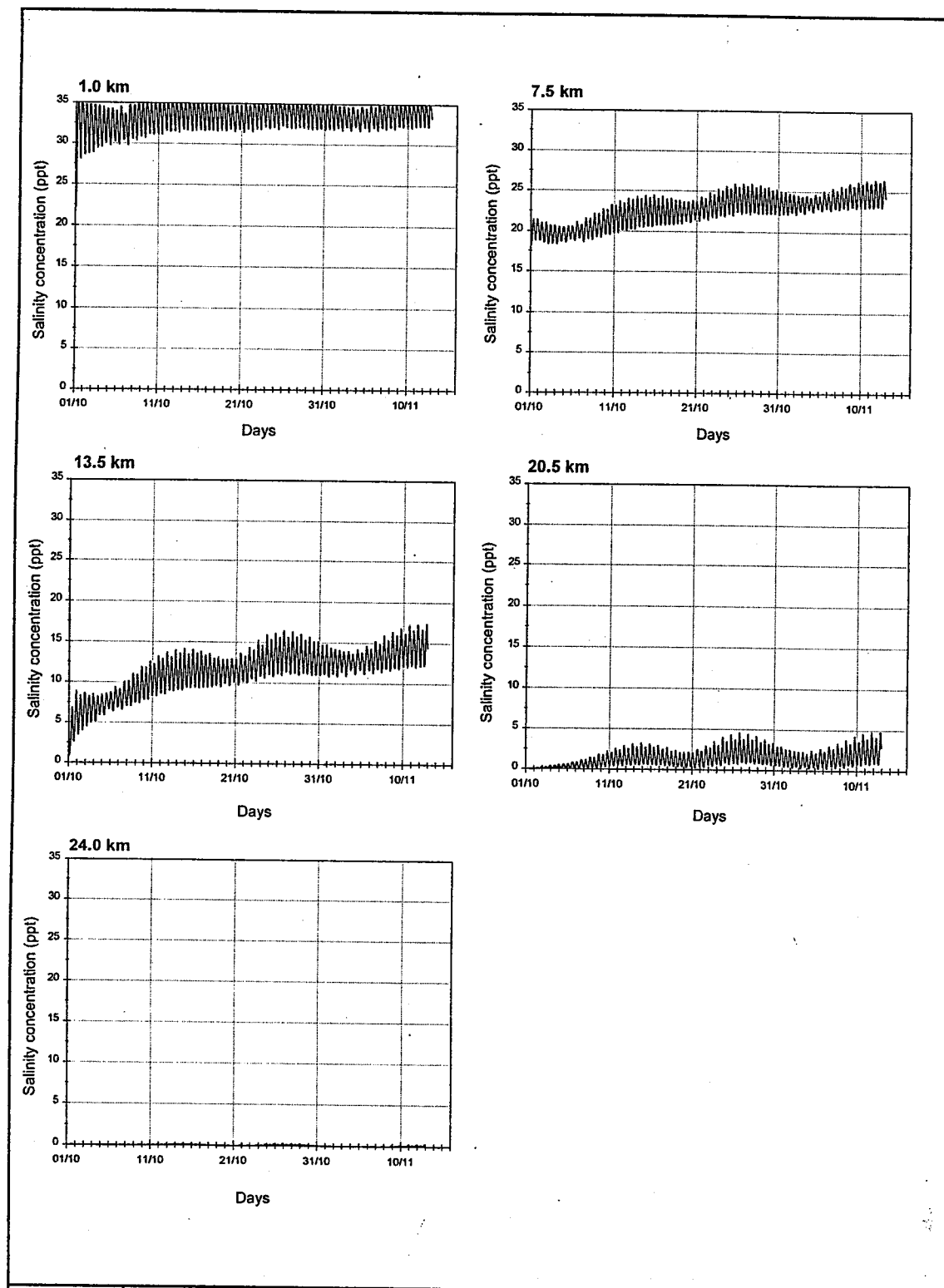


Figure 2.9 A: Simulated salinity concentrations at a river flow of $0.1 \text{ m}^3 \text{ s}^{-1}$ (before 1996 flood event).

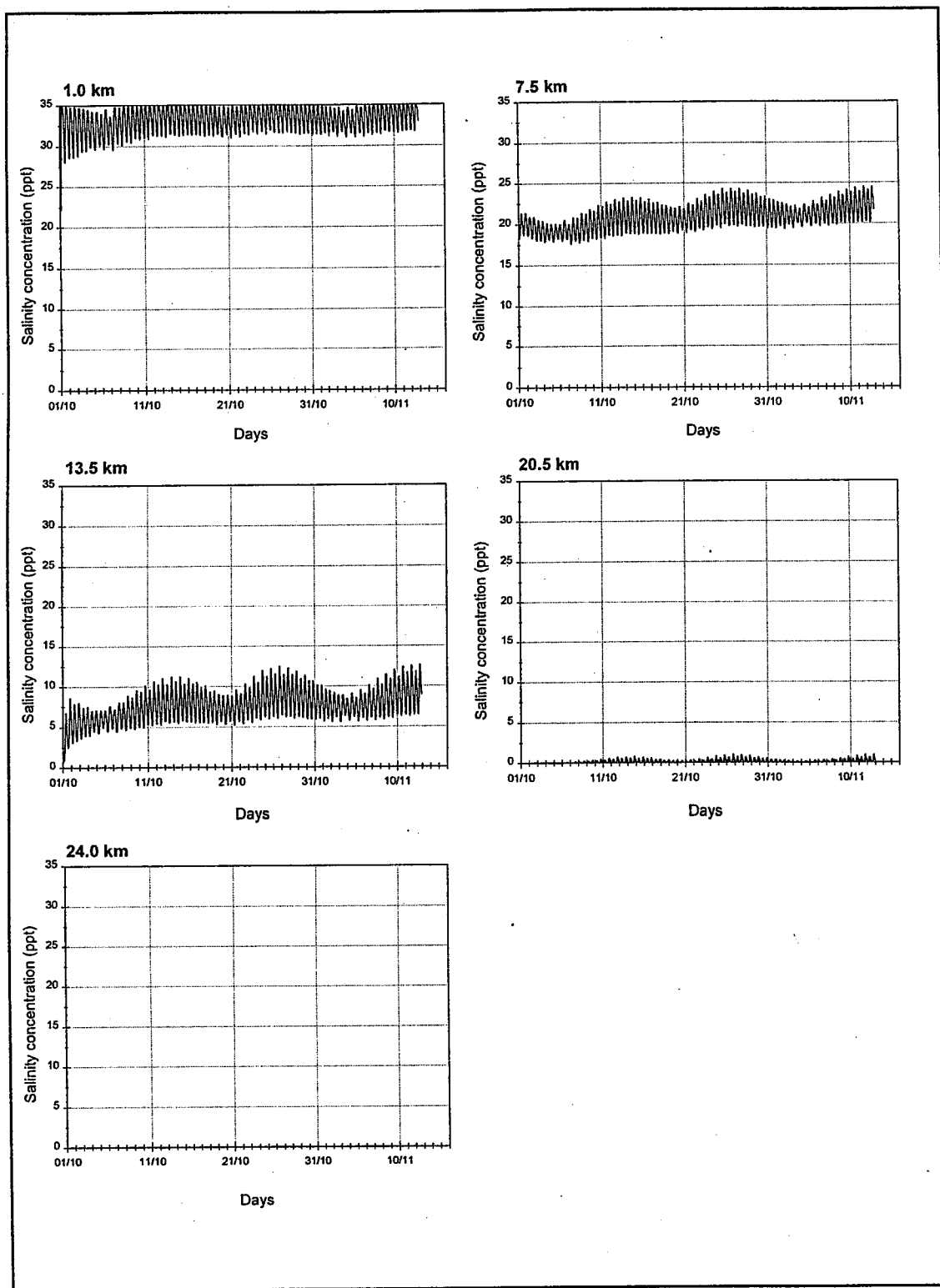


Figure 2.9 B: Simulated salinity concentrations at a river flow of $0.25 \text{ m}^3 \text{ s}^{-1}$ (before 1996 flood event).

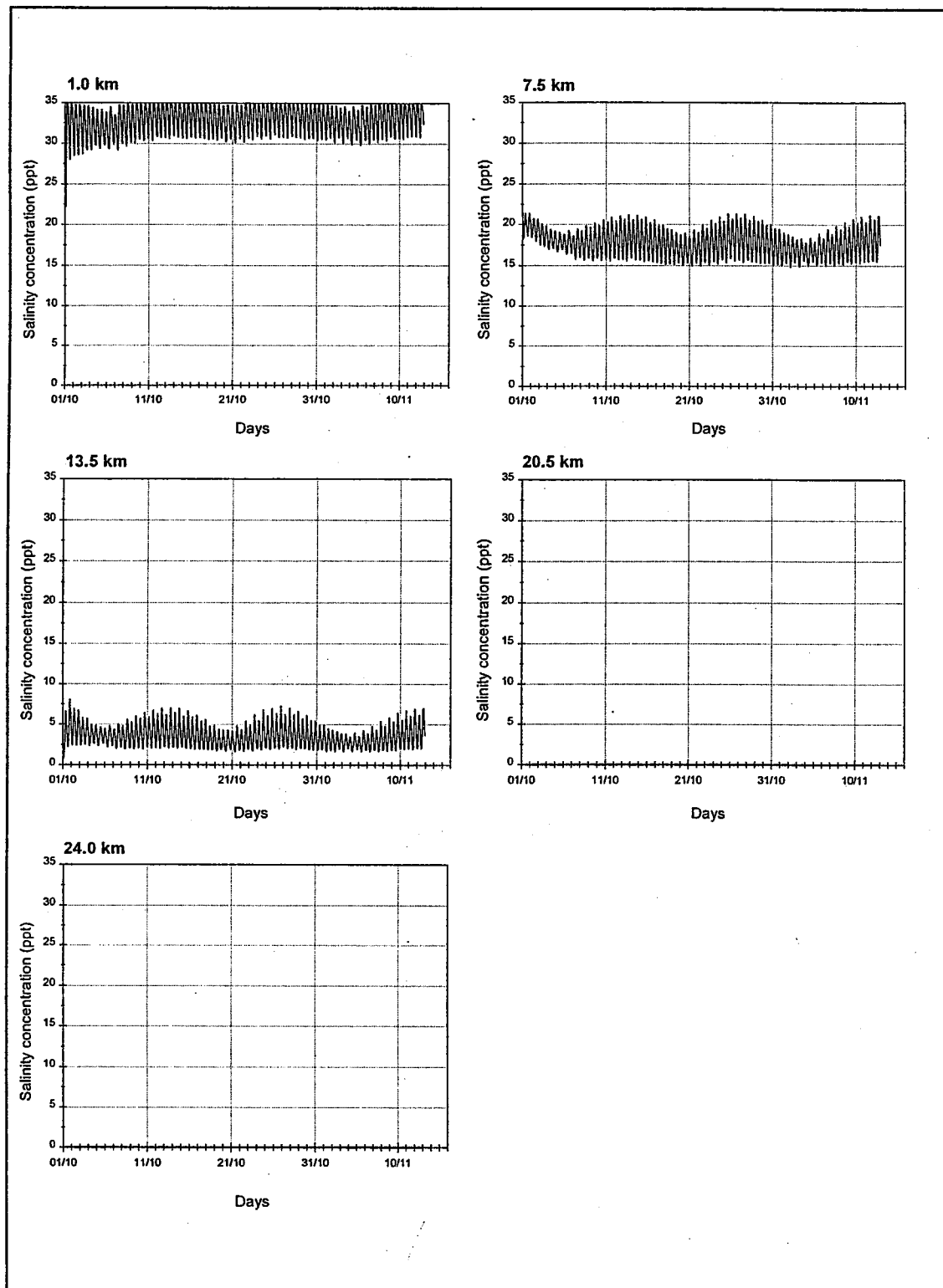


Figure 2.9 C: Simulated salinity concentrations at a river flow of $0.5 \text{ m}^3 \text{ s}^{-1}$ (before 1996 flood event).

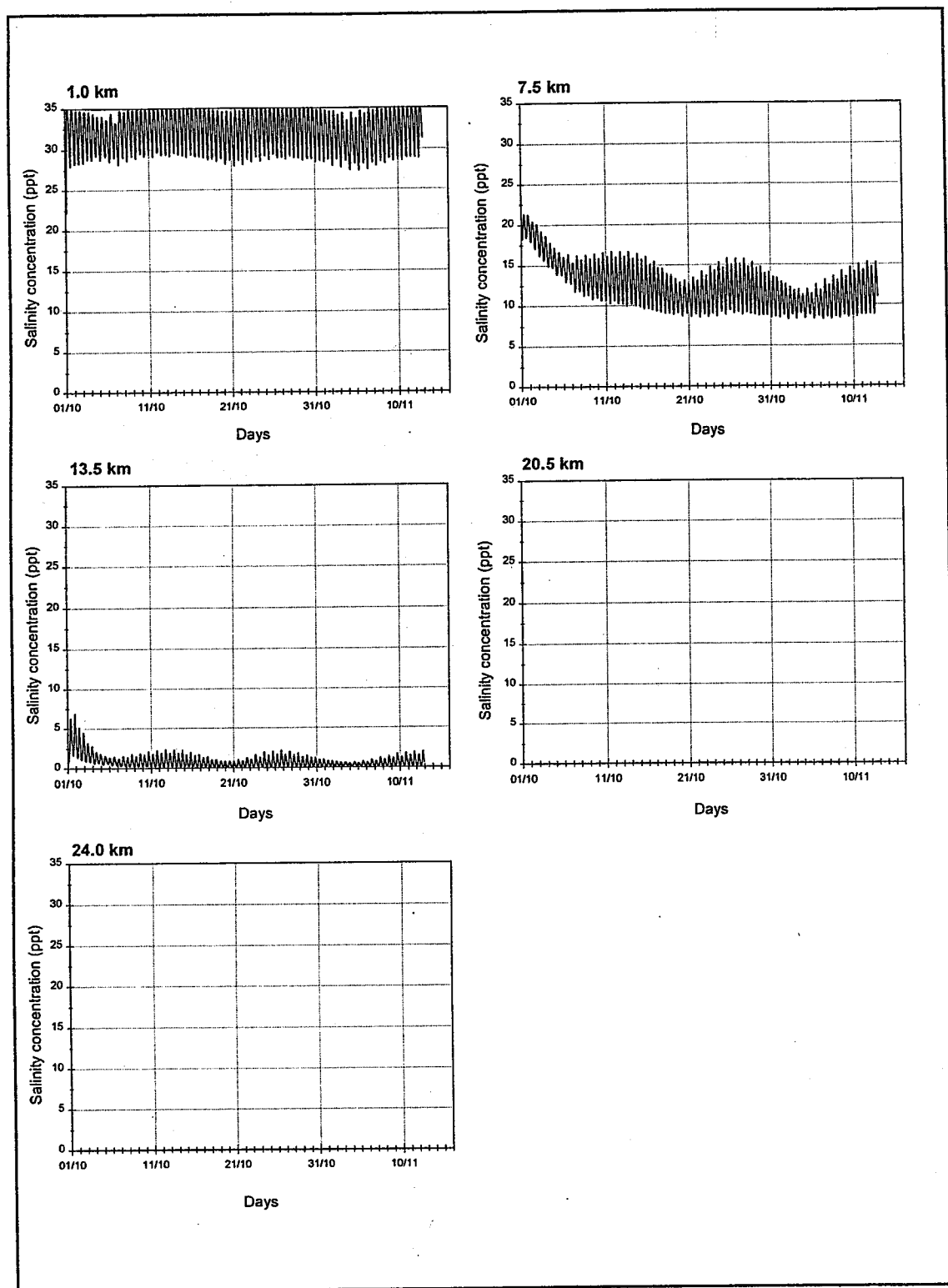


Figure 2.9 D: Simulated salinity concentrations at a river flow of $1.0 \text{ m}^3 \text{ s}^{-1}$ (before 1996 flood event).

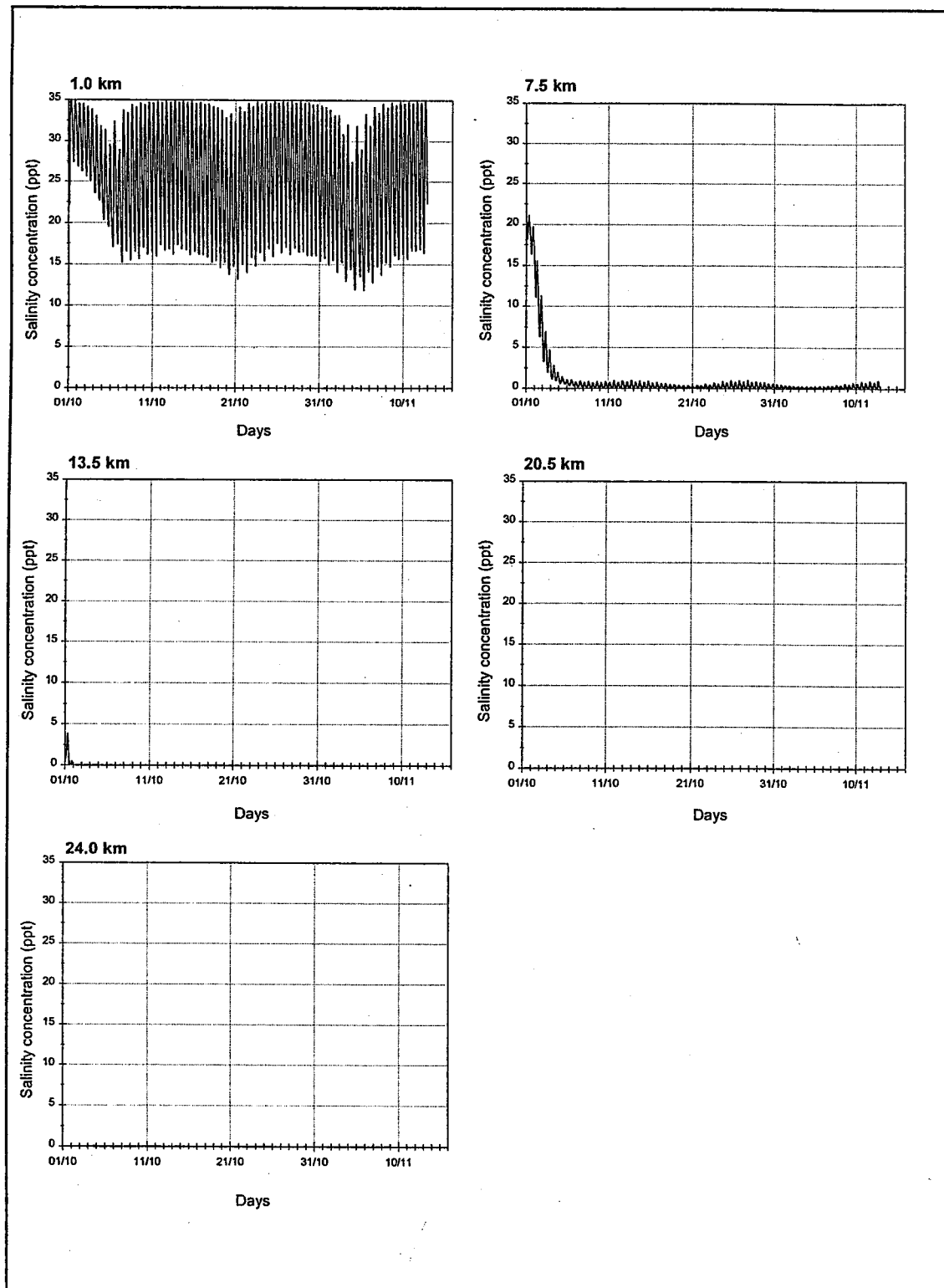


Figure 2.9 E: Simulated salinity concentrations at a river flow of $5.0 \text{ m}^3 \text{ s}^{-1}$ (before 1996 flood event).

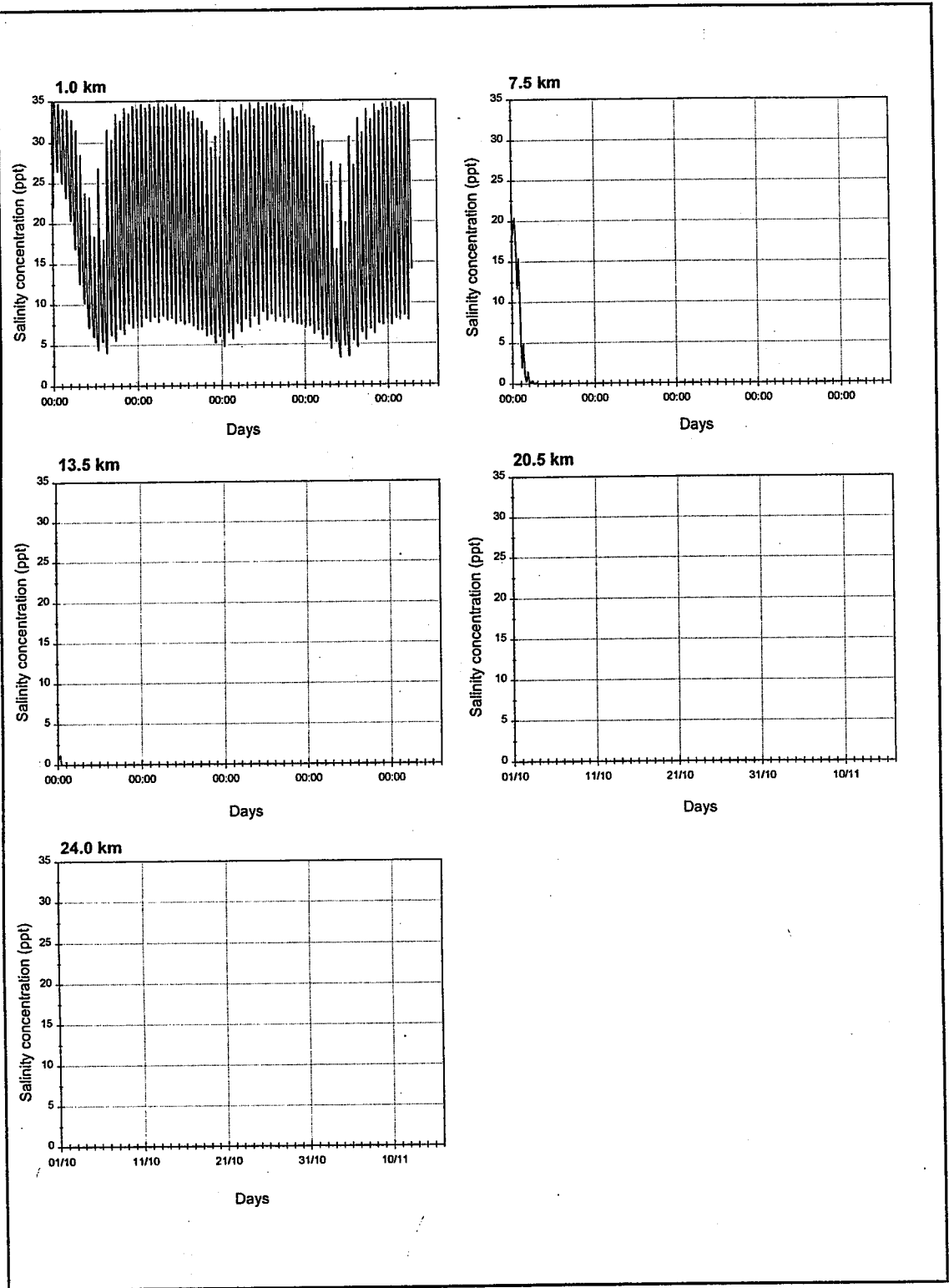


Figure 2.9 F: Simulated salinity concentrations at a river flow of $10.0 \text{ m}^3 \text{ s}^{-1}$ (before 1996 flood event).

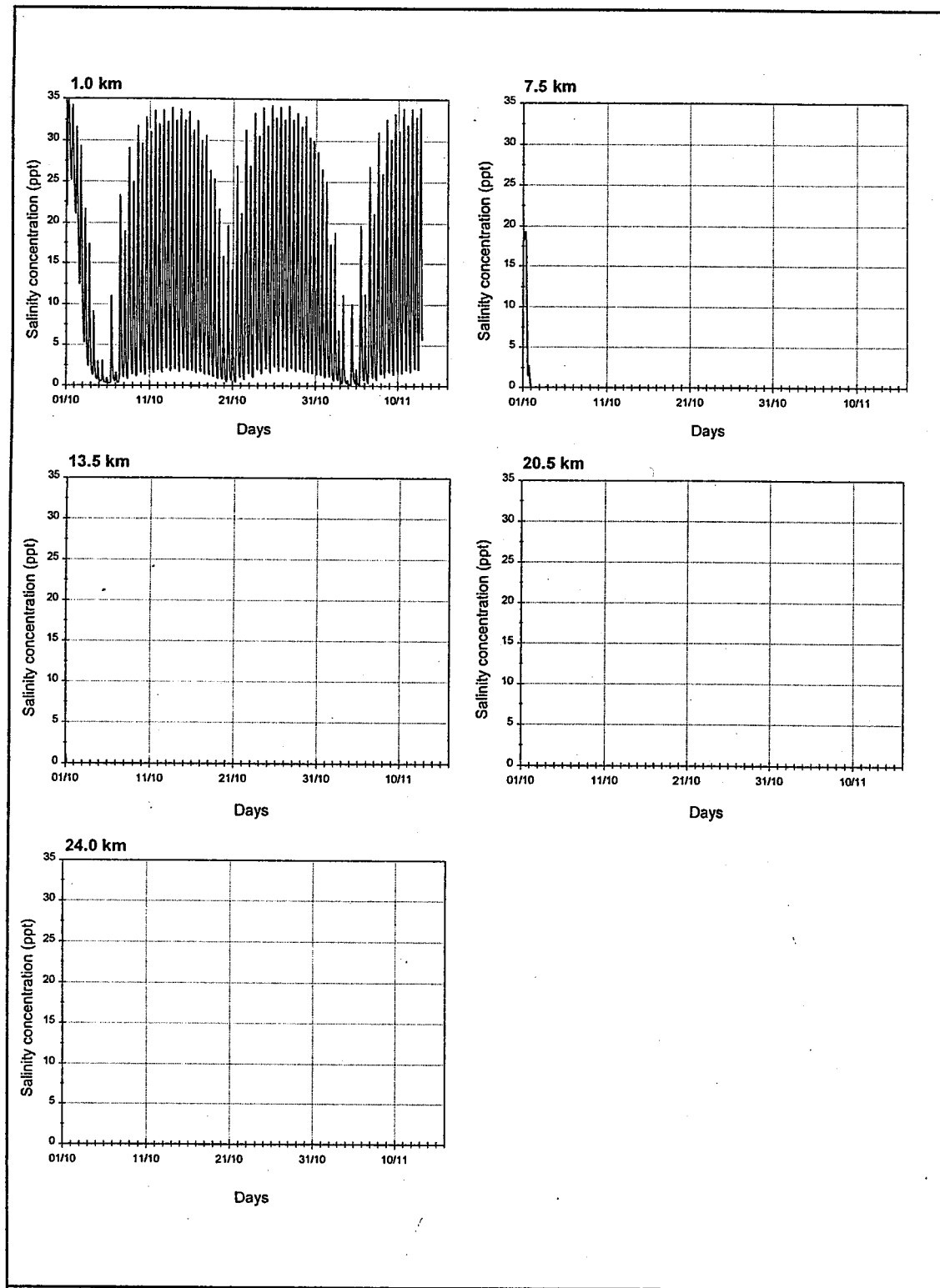


Figure 2.9 G: Simulated salinity concentrations at a river flow of $20.0 \text{ m}^3 \text{ s}^{-1}$ (before 1996 flood event).

2.3.3.2 Dry year simulations for pre- and post-flood conditions

Data for a low run-off period of the 1985/1986 hydrological year was selected for the model simulations to compare the effects of the deepening by the river flood. These data were:

Month	Flow (m ³ s ⁻¹)
Oct	3.83
Nov	8.64
Dec	16.17
Jan	2.12
Feb	0.59
Mar	0.54
Apr	0.44
May	0.20
Jun	0.31
Jul	0.31
Aug	5.77
Sep	4.80

The results of the model simulations of salinity concentrations in the estuary for the selected (dry) year (using the pre- and post-1996 flood scour topography as reference conditions) are presented in Figures 2.12 A to C.

Minimal differences in salinity concentrations were observed for the high run-off period from July to December. Greater differences in salinity concentrations were observed for the low runoff period from January to June. These are briefly discussed below for each location:

1. Station 1 (1.0 km from mouth) - the differences are insignificant for Station 1. The simulated salinity concentrations vary between 27.0 and 35.0 ‰ for pre-flood conditions and between 28.0 and 35.0 ‰ for post-flood conditions.
2. Station 2 (7.5 km from mouth) - in general, slightly higher salinity concentrations were simulated for pre-flood conditions than for post-flood conditions, indicating a slightly stronger saline intrusion. The variations in salinity concentrations were also slightly more for post-flood conditions.
3. Station 3 (13.5 km from mouth) - the average level of salinity concentrations were similar for pre- and post-flood conditions, but for pre-flood conditions a broader variation in salinity concentrations was observed.
4. Station 4 (20.5 km from mouth) - only a small penetration of saline water (maximum concentrations of 2.0 ‰) was observed from April to June for pre-flood conditions. For post-flood conditions an increase in penetration of saline water (maximum concentrations of 6.0 ‰) was observed, which also occurred for a longer period from February to June.

5. Station 5 (24.0 km from mouth) - no saline penetration was observed for pre-flood conditions, but a slight penetration (maximum concentrations of 2.0 ‰) was observed for post-flood conditions.

2.3.4 River flow and salinity concentration correlations

The work on salinity distributions described in this report was undertaken in parallel with research on the importance of salinity distributions to the ecology of the estuary. This work concentrated on the REI region, especially the position of the 10.0 ‰ isohaline. Not only is the determination of this position important, but the volume of water upstream of this interface is also important, as this represents the size of the habitat available for certain biotic components. Further investigations were therefore undertaken using the model to quantify the volumes of water present in the estuary below selected salinity concentrations under the different (low) flow conditions. These volumes were determined between the following salinity concentrations:

Volumes	Salinity concentrations
Volume 1	< 5 ‰
Volume 2	< 10 ‰
Volume 3	< 15 ‰
Volume 4	< 20 ‰
Volume 5	< 25 ‰
Volume 6	< 30 ‰
Volume 7	< 35 ‰

These volumes were determined for the constant river flows of 0.1, 0.25, 0.5, 1.0, 5.0, 10.0 and 20.0 m³ s⁻¹ and for pre- and post-flood topography conditions. The results are summarised in Tables 2.3 A and B for pre-flood and in Tables 2.4 A and B for post-flood conditions, both for neap tide and for spring tide and also in each case for high tide and for low tide.

Tables 2.4 A and B show the volumes of water in the Gamtoos Estuary with lower salinity concentrations than indicated for spring tide and for neap tide, both for high tide and low tide, for pre-1997 flood conditions in 10⁶ m³ s⁻¹. Tables 2.4 A and B show the volumes of water in the estuary with lower salinity concentrations than indicated for spring tide and for neap tide, both for high tide and for low tide, for post-1997 flood conditions in 10⁶ m³ s⁻¹. The results provide an input to the ecological research discussed elsewhere in this report.

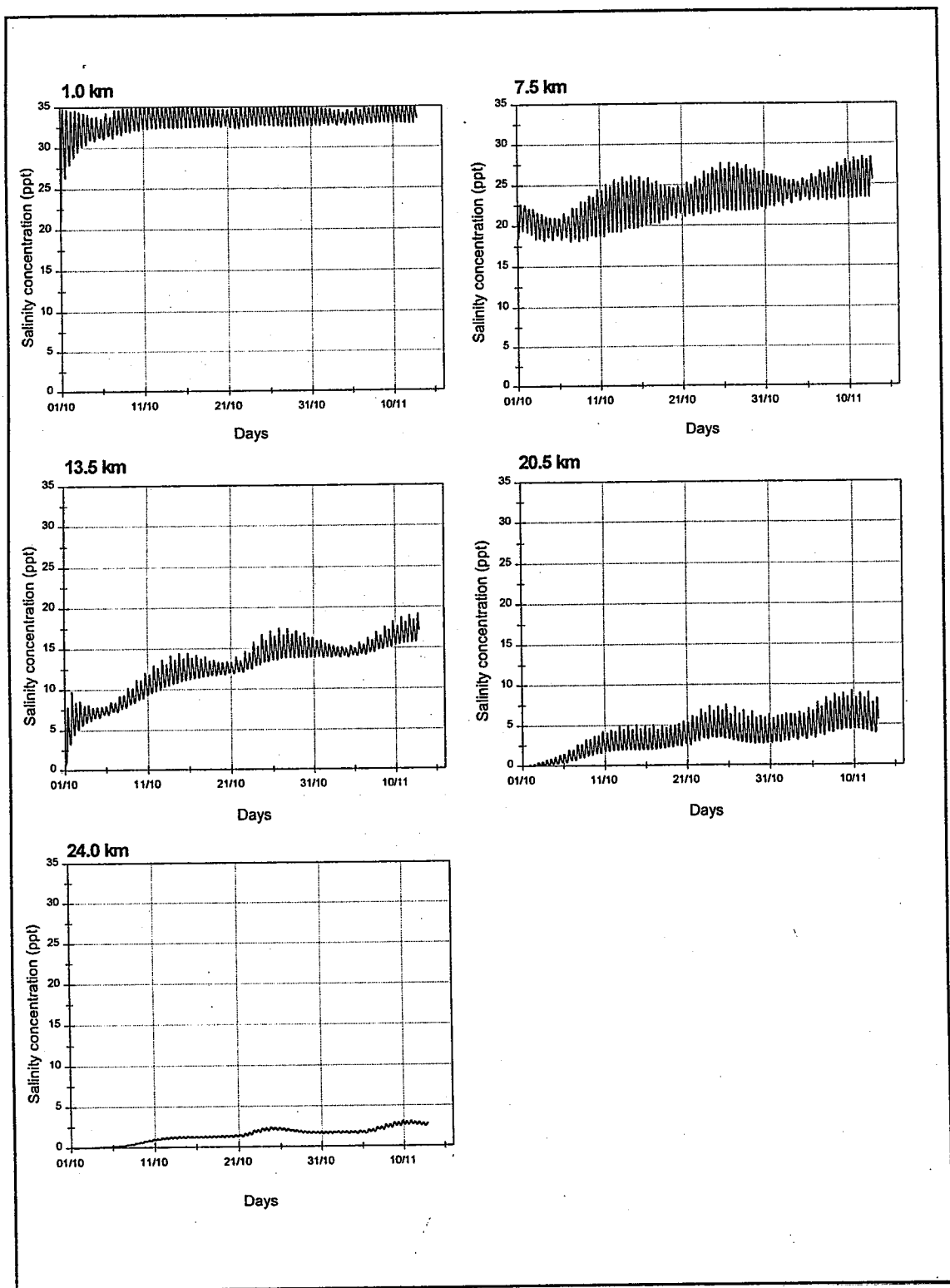


Figure 2.10 A: Simulated salinity concentrations at various cross-sections at a river flow of $0.1 \text{ m}^3 \text{ s}^{-1}$ (after a 1996 flood event).

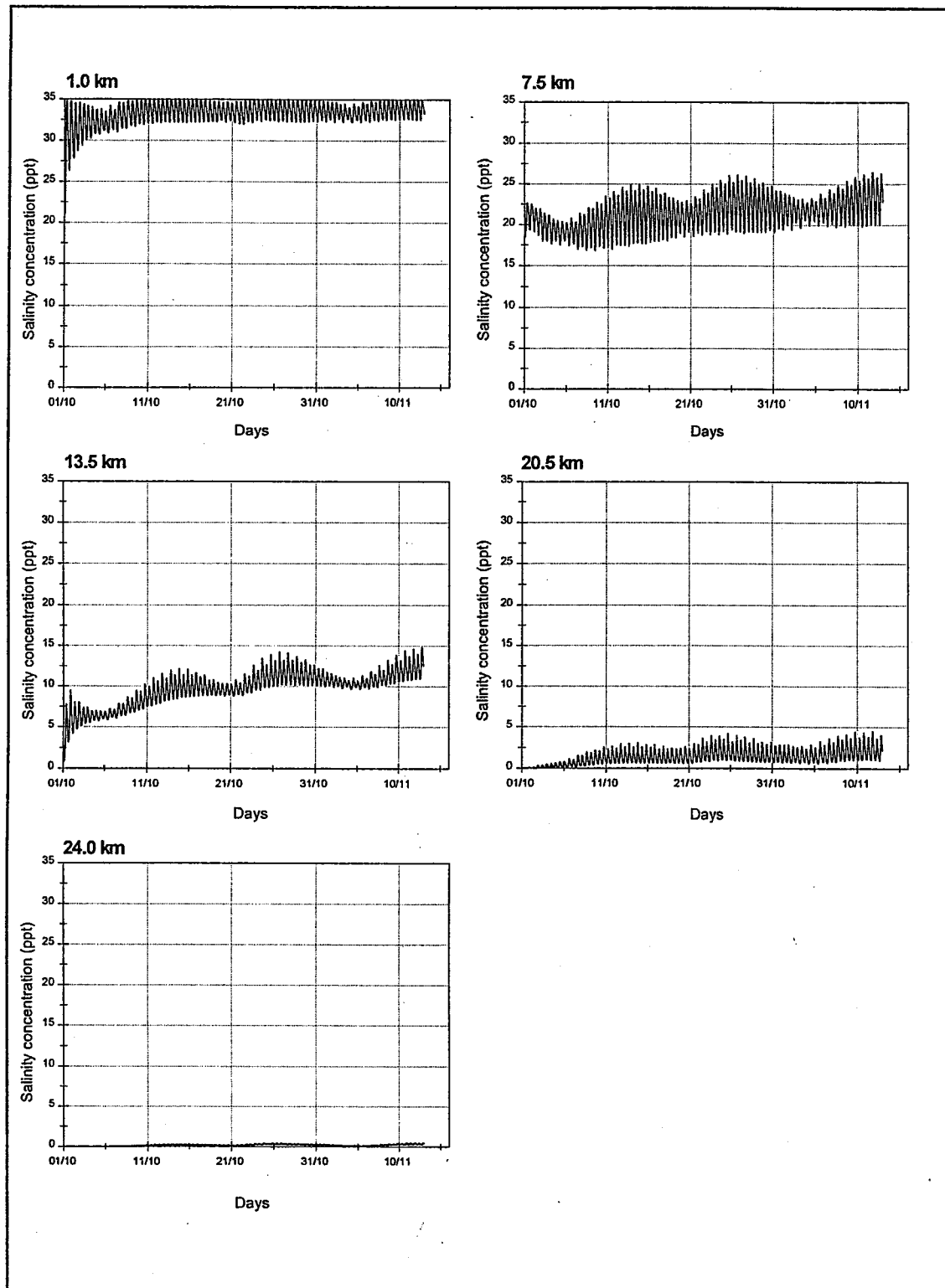


Figure 2.10 B: Simulated salinity concentrations at various cross-sections at a river flow of $0.25 \text{ m}^3 \text{ s}^{-1}$ (after a 1996 flood event).

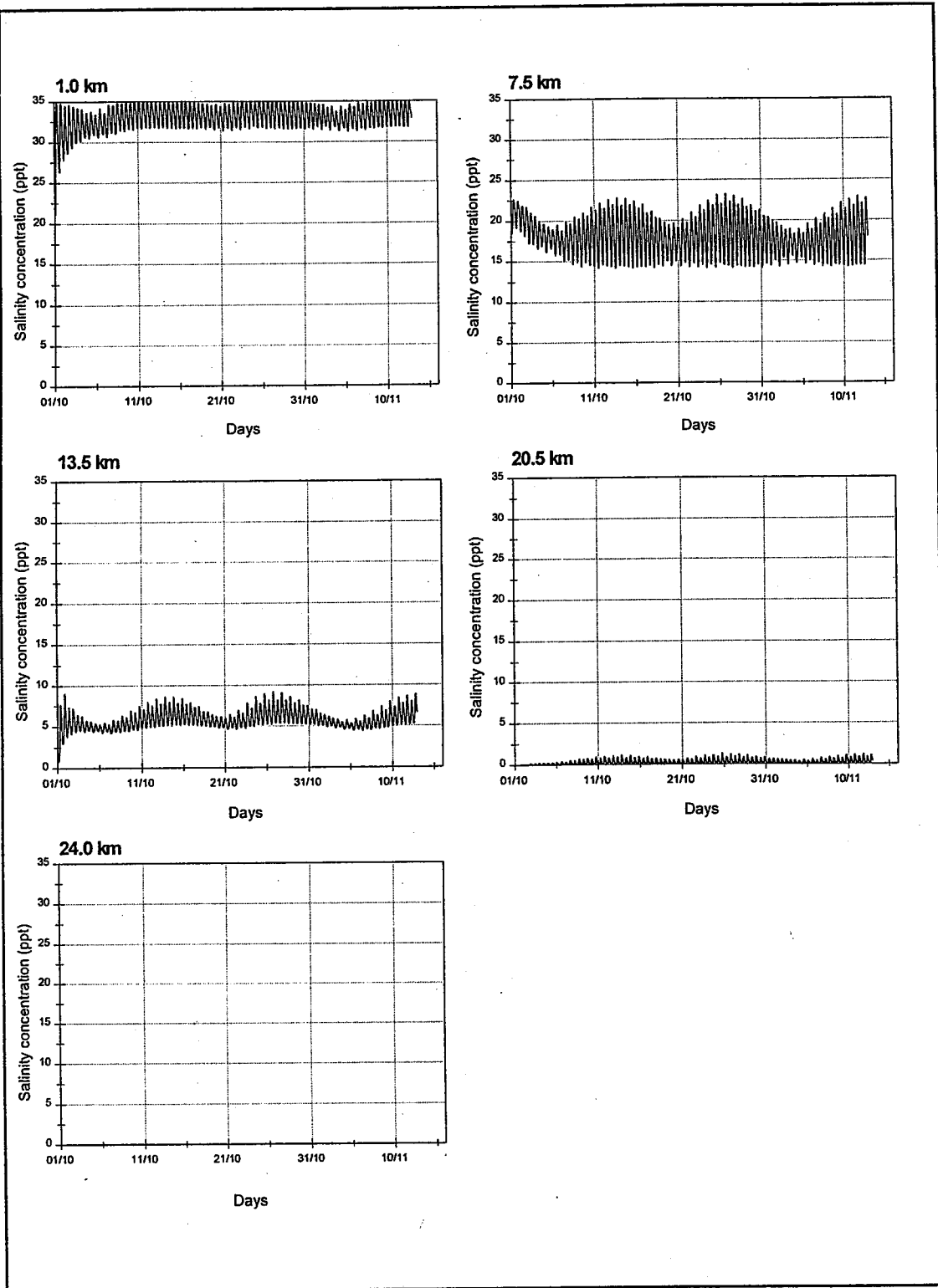


Figure 2.10 C: Simulated salinity concentrations at various cross-sections at a river flow of $0.5 \text{ m}^3 \text{ s}^{-1}$ (after a 1996 flood event).

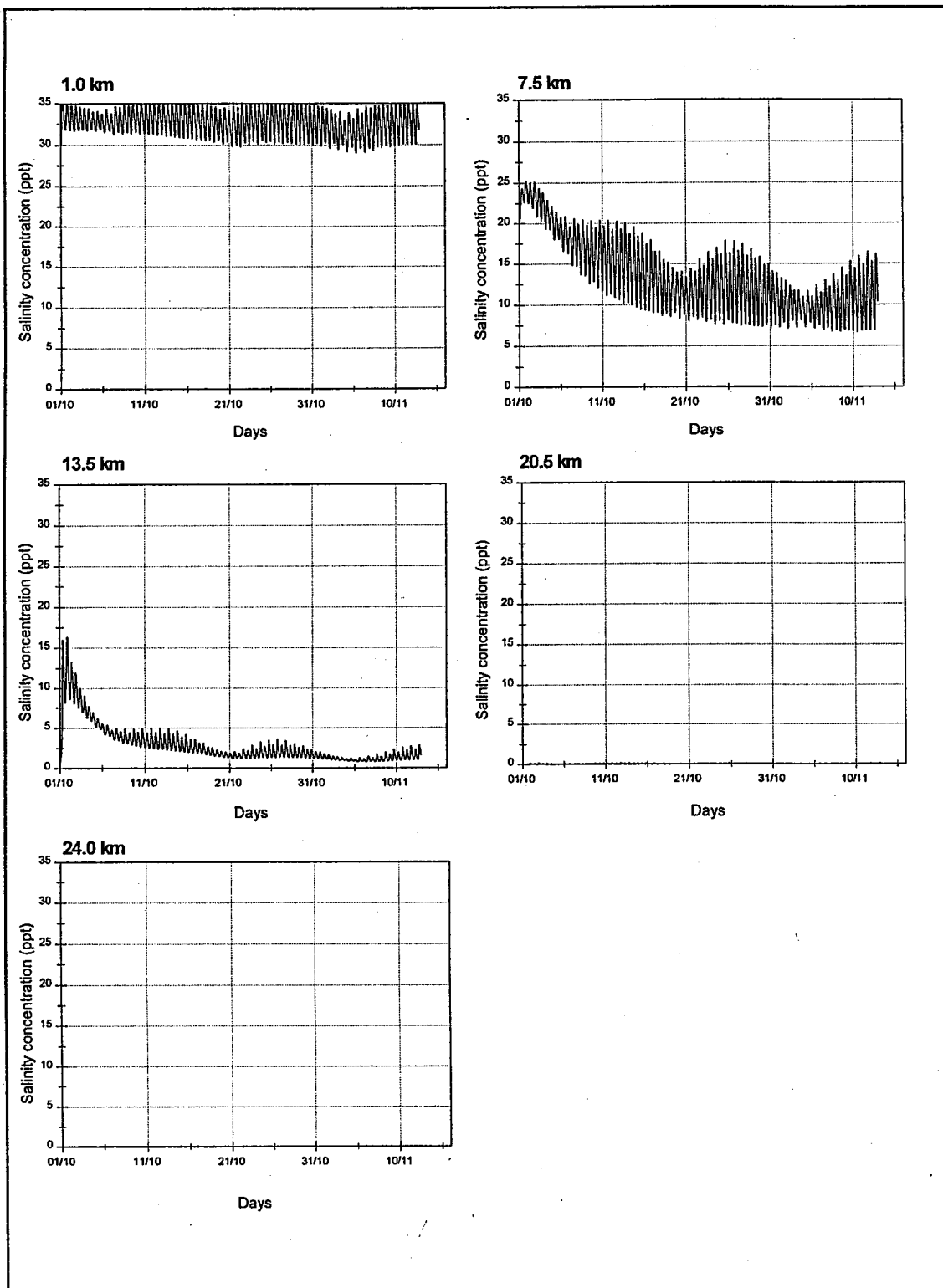


Figure 2.10 D: Simulated salinity concentrations at various cross-sections at a river flow of $1.0 \text{ m}^3 \text{ s}^{-1}$ (after a 1996 flood event).

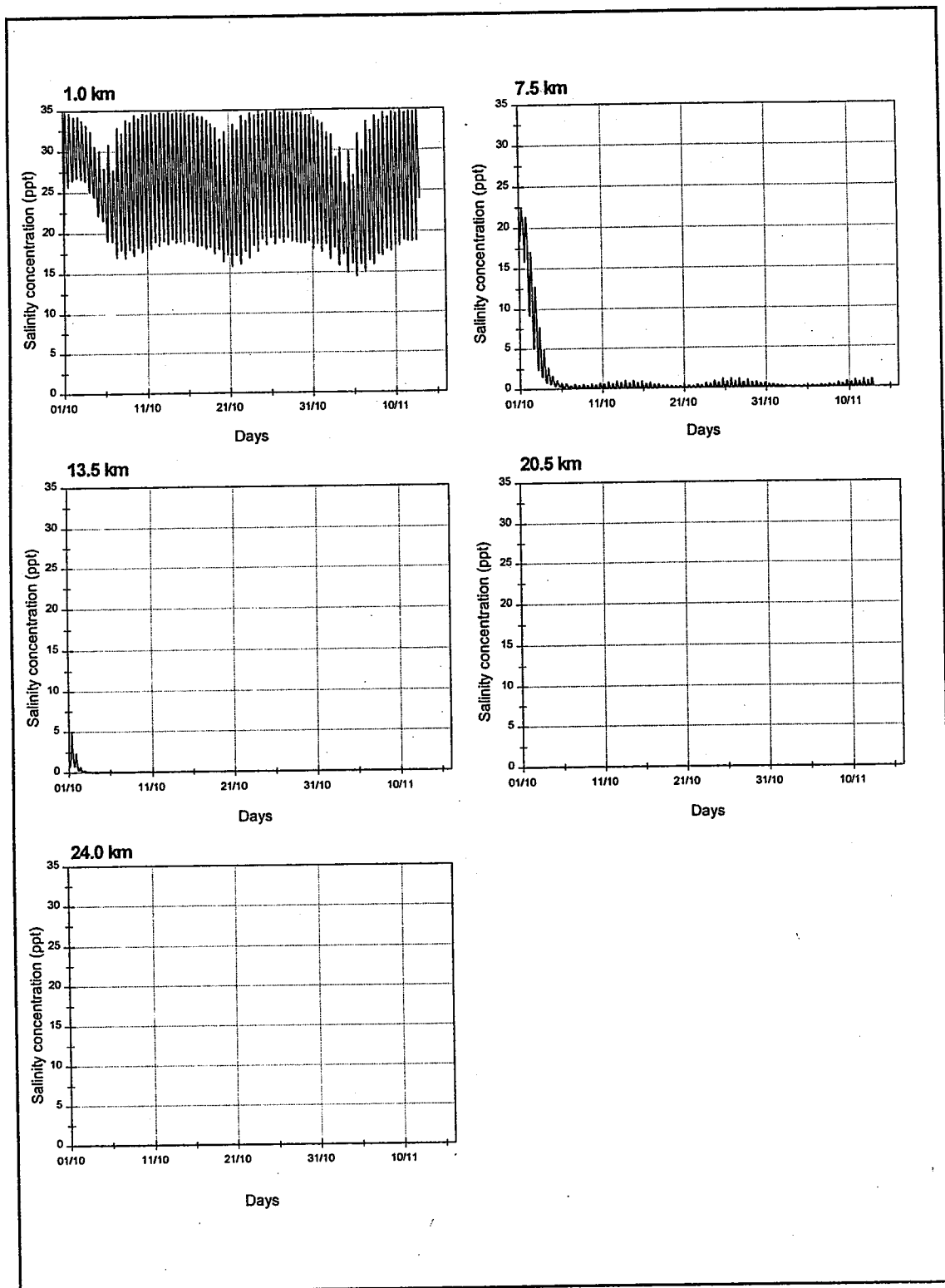


Figure 2.10 E: Simulated salinity concentrations at various cross-sections at a river flow of $5.0 \text{ m}^3 \text{ s}^{-1}$ (after a 1996 flood event).

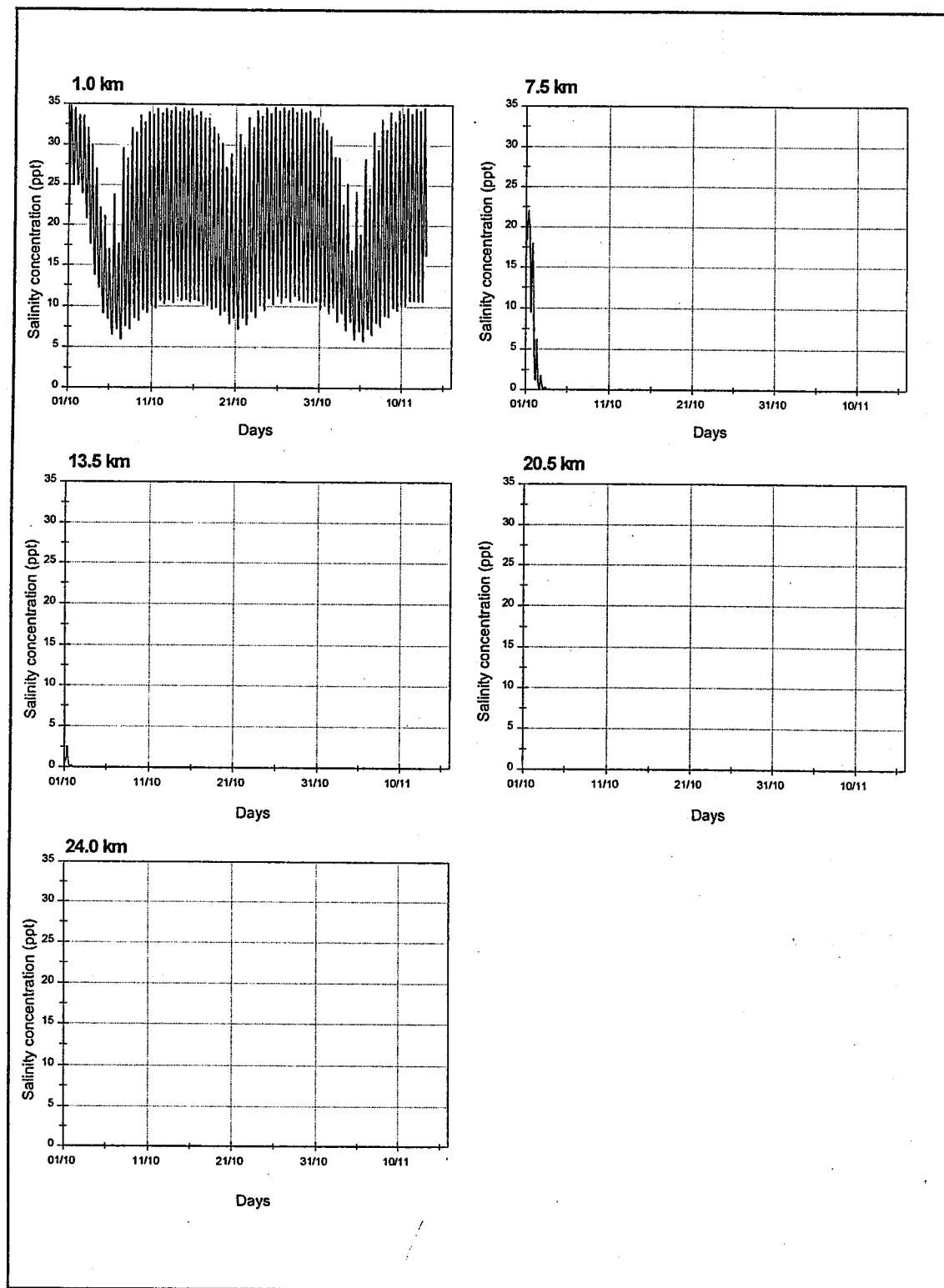


Figure 2.10 F: Simulated salinity concentrations at various cross-sections at a river flow of $10.0 \text{ m}^3 \text{ s}^{-1}$ (after a 1996 flood event).

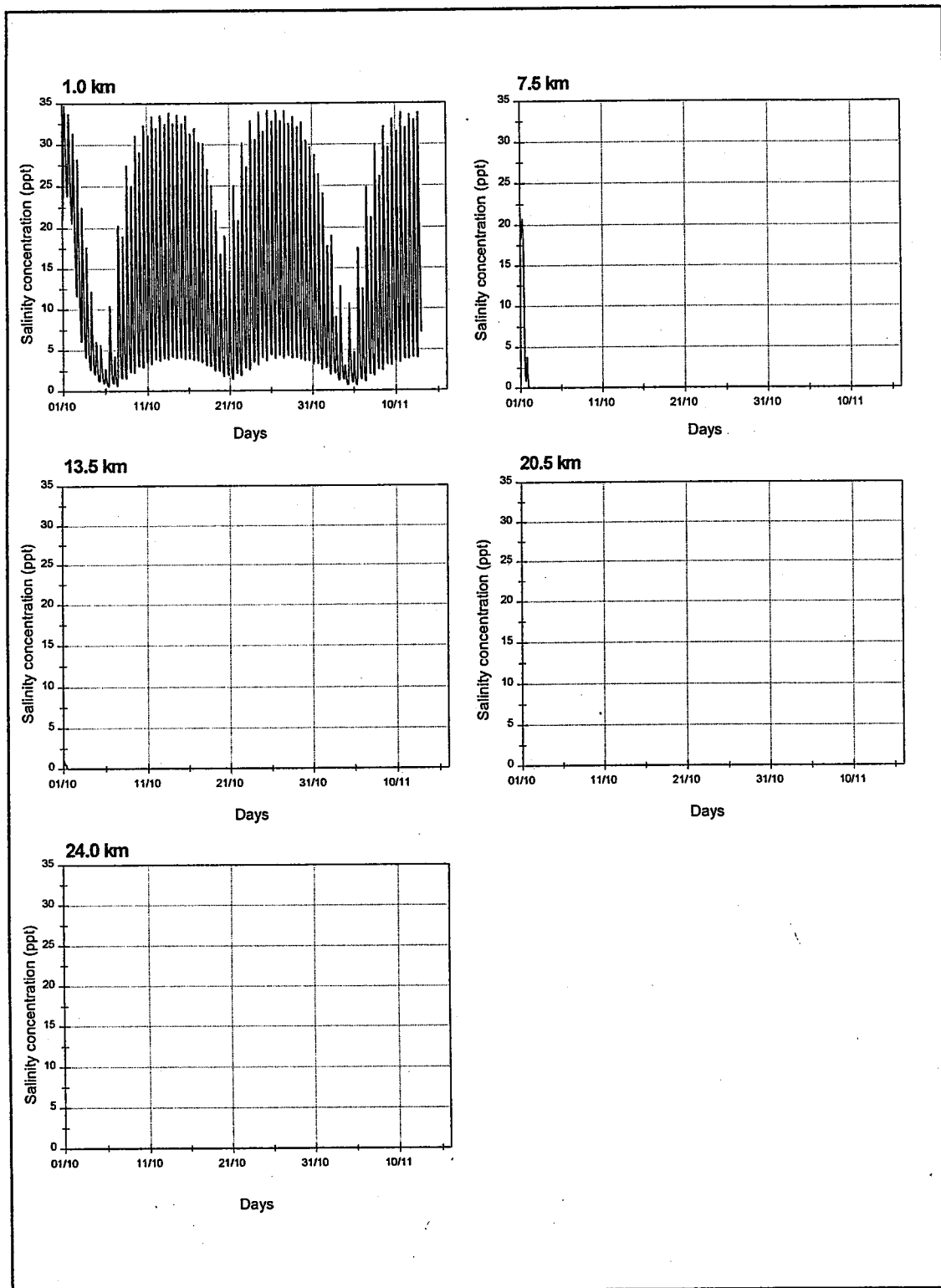


Figure 2.10 G: Simulated salinity concentrations at various cross-sections at a river flow of $20.0 \text{ m}^3 \text{ s}^{-1}$ (after a 1996 flood event).

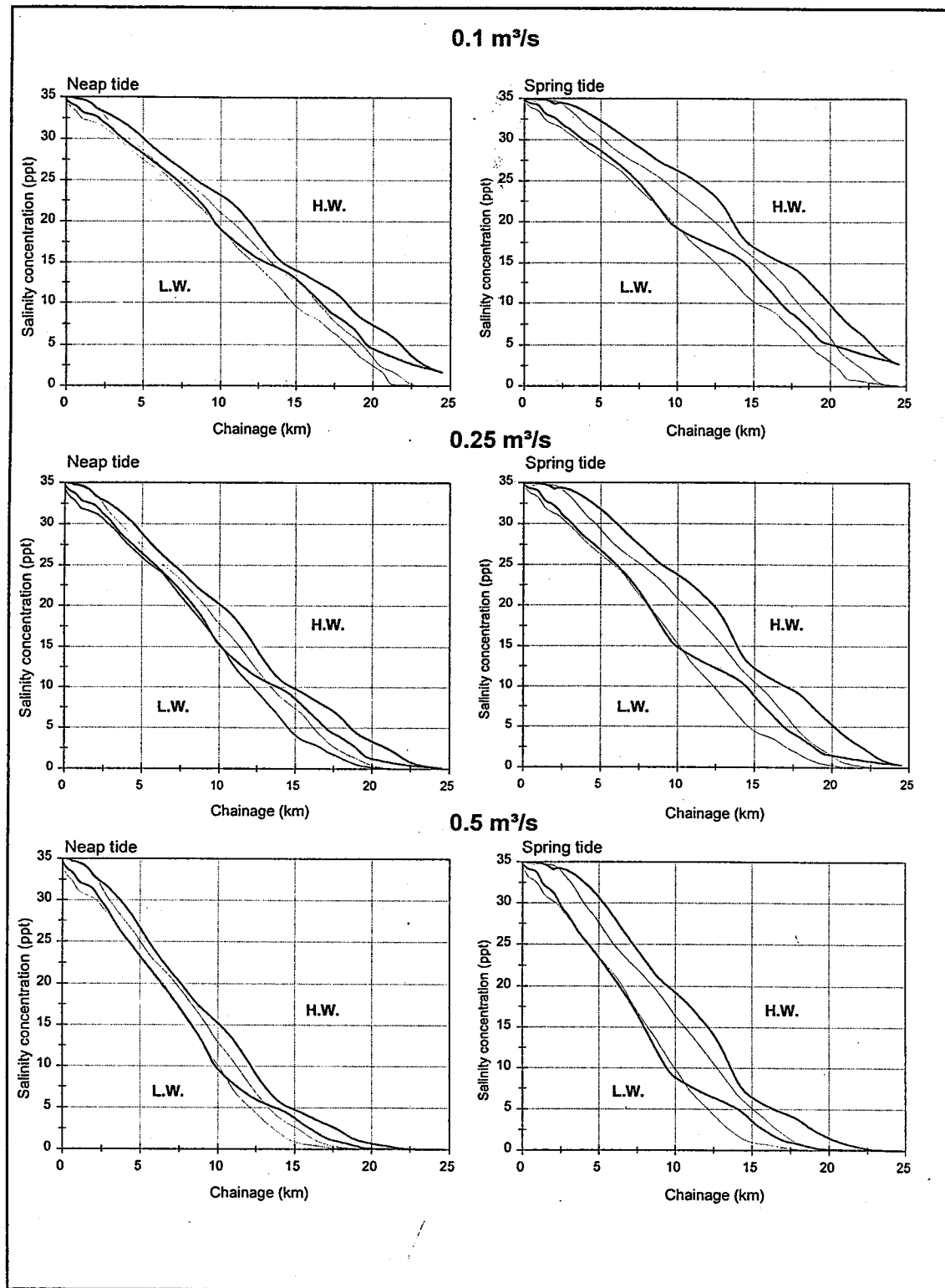


Figure 2.11 A: Simulated salinity distributions illustrating changes in phases for indicated flow conditions before a 1996 flood event (dark lines) and after a 1996 flood event (light lines).

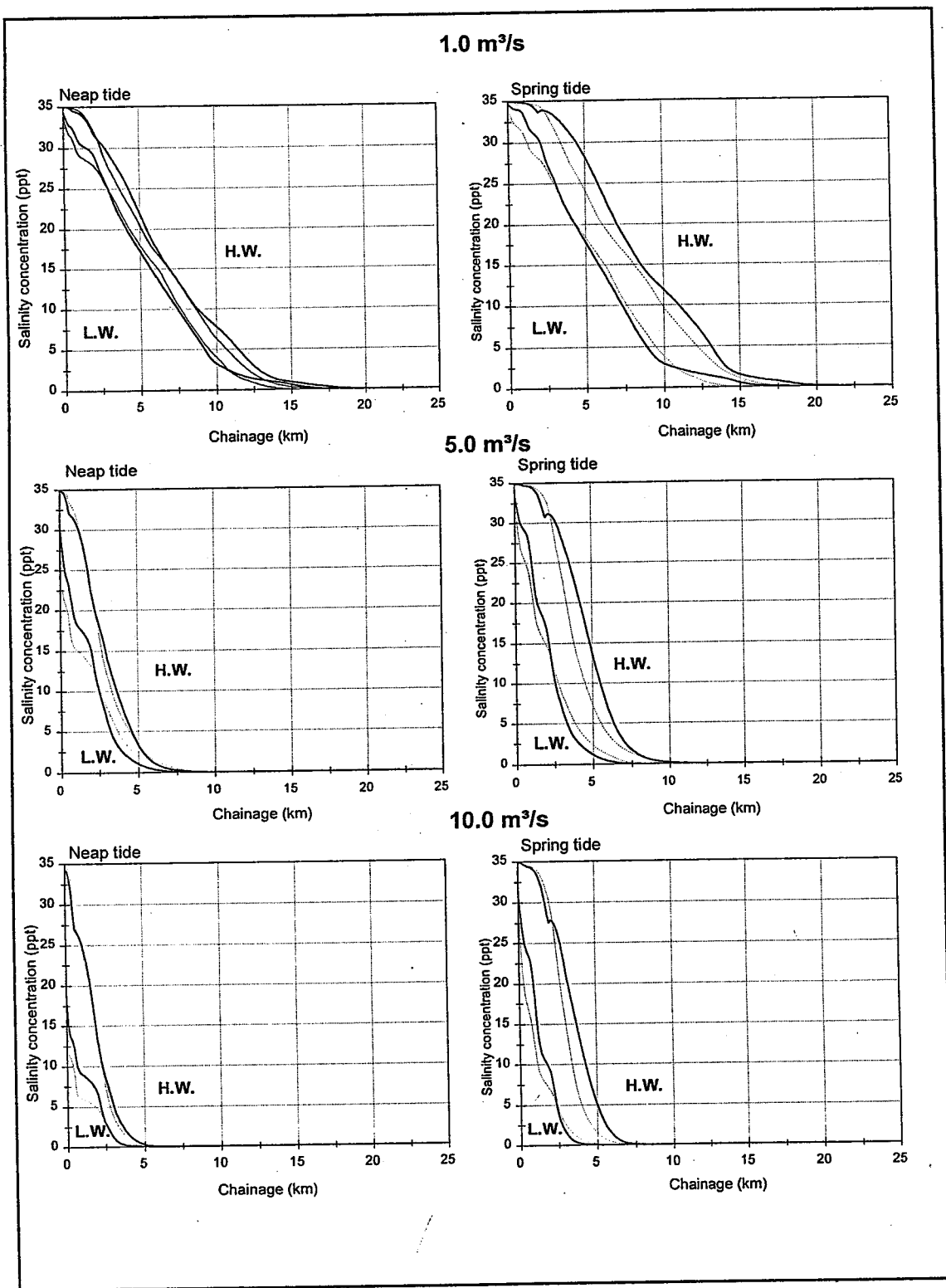


Figure 2.11 B: Simulated salinity distributions illustrating changes in phases for indicated flow conditions before a 1996 flood event (dark lines) and after a 1996 flood event (light lines).

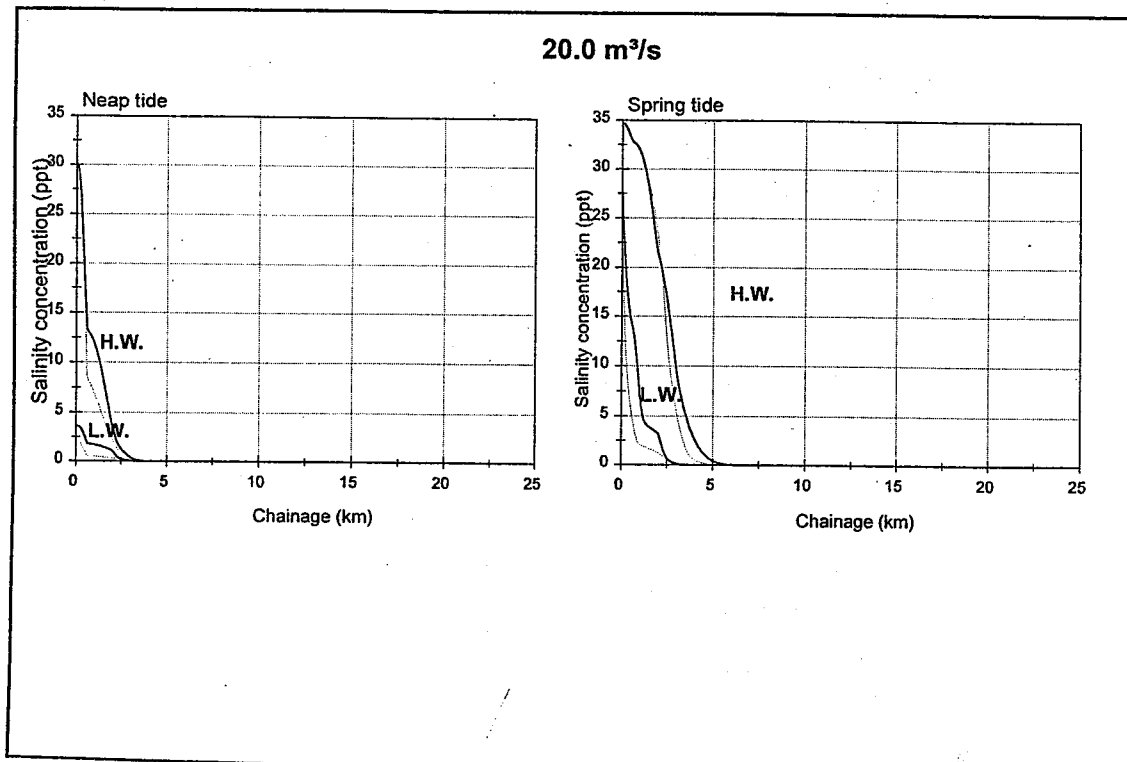


Figure 2.11 C: Simulated salinity distributions illustrating changes in phases for indicated flow conditions before a 1996 flood event (dark lines) and after a 1996 flood event (light lines).

2.3.5 Monthly averaged salinity distributions

Tables 2.5 A and B, and Tables 2.6 A and B can be used to estimate the averaged monthly salinity distributions from 1990 to 1998 in the Gamtoos Estuary. In Table 2.5 A, the distances from the mouth at which indicated salinity concentrations occur under different river flows at neap tide and spring tide, and averaged for high tide and low tide (for conditions before the flood of November 1996), are presented. The same information is provided in Table 2.5 B for the conditions after the flood in November 1996.

The salinity concentrations are obtained from simulations with the Mike 11 model, which was not properly calibrated because of lack of data. The Mike 11 model is also one-dimensional and generates vertically averaged salinity distributions. The accuracy of the salinity concentrations is therefore limited and this should be considered when the data are used.

Estimated monthly river flow data for the period 1990 to 1998 were compiled by Ninham Shand Consulting Engineers. The results are included in Tables 2.6 A to C. In Table 2.6A, the estimated monthly river flow, including the flow from the Loerie River (see Figure 2.1) into the Gamtoos Estuary in $10^6 \text{ m}^3 \text{ s}^{-1}$, is presented. This data is based on measured river flow and additional hydrological modeling. This table can be used for the estimation of salinity concentrations below the confluence with the Loerie River, which is at 9.5 km from the mouth of the estuary. Table 2.6 B contains similar data to Table 2.6 A, with the exception that it excludes the river flow from the Loerie River. Table 2.6 B should be used for the

estimation of salinity concentrations in the Gamtoos Estuary upstream of the confluence with the Loerie River. The accuracy of the data in Tables 2.6 A and B are limited because the tables had to be compiled from partly measured flow data and data partly generated by uncalibrated hydrological models. Although the data is useful, the shortcomings in accuracy should be considered when it is used. The procedure for using the tables is as follows:

1. For the stretch below the confluence with the Loerie River use Table 2.6 A and above the confluence Table 2.6 B.
2. Select the month for which the salinity concentrations are needed. For example June 1994 had a flow of $0.40 \text{ m}^3 \text{ s}^{-1}$ below the Loerie confluence (Table 2.6 A) and of $0.36 \text{ m}^3 \text{ s}^{-1}$ above the confluence (Table 2.6B).
3. June 1994 was before the November 1996 flood and therefore the salinity data of Table 2.5 A should be used.
4. By interpolation between the salinity data for the flows of 0.25 and $0.5 \text{ m}^3 \text{ s}^{-1}$, the salinity concentrations along the estuary can be determined.

2.3.6 Retention times

The term 'retention time' is often used in ecological studies on estuaries. This term normally refers to the period in which the volume of water in an estuary, or in a part of the estuary, is being replaced by river and/or tidal flows. For example, the retention time for a whole estuary with a volume of one million m^3 and a river flow of $1.0 \text{ m}^3 \text{ s}^{-1}$, could be 1 000 000 seconds or 278 hours or 11.6 days. In this example, the retention time literally indicates that all the water originally in the estuary is replaced after 11.6 days. This would also imply that a tracer originally present would have completely disappeared after 11.6 days.

Advection and dispersion play major roles in estuaries and are complex processes. They determine the periods during which tracer concentrations in an estuary or part of it are being reduced. Model simulations were undertaken for river flows of 0.1 , 1.0 and $10.0 \text{ m}^3 \text{ s}^{-1}$ to illustrate this reduction for the Gamtoos Estuary. In these simulations it was assumed that a tracer with a constant concentration of 100 % was initially present in the estuary. It was also assumed that the tracer concentration was zero in seawater and in the river flow entering the estuary.

The results are presented as graphs of longitudinal profiles of tracer concentrations along the estuary at indicated time intervals (Figure 2.13) and as time histories of these concentrations at distances of 1.0 , 7.5 , 13.5 , 20.5 and 24.0 km from the mouth (Figures 2.14 A to C).

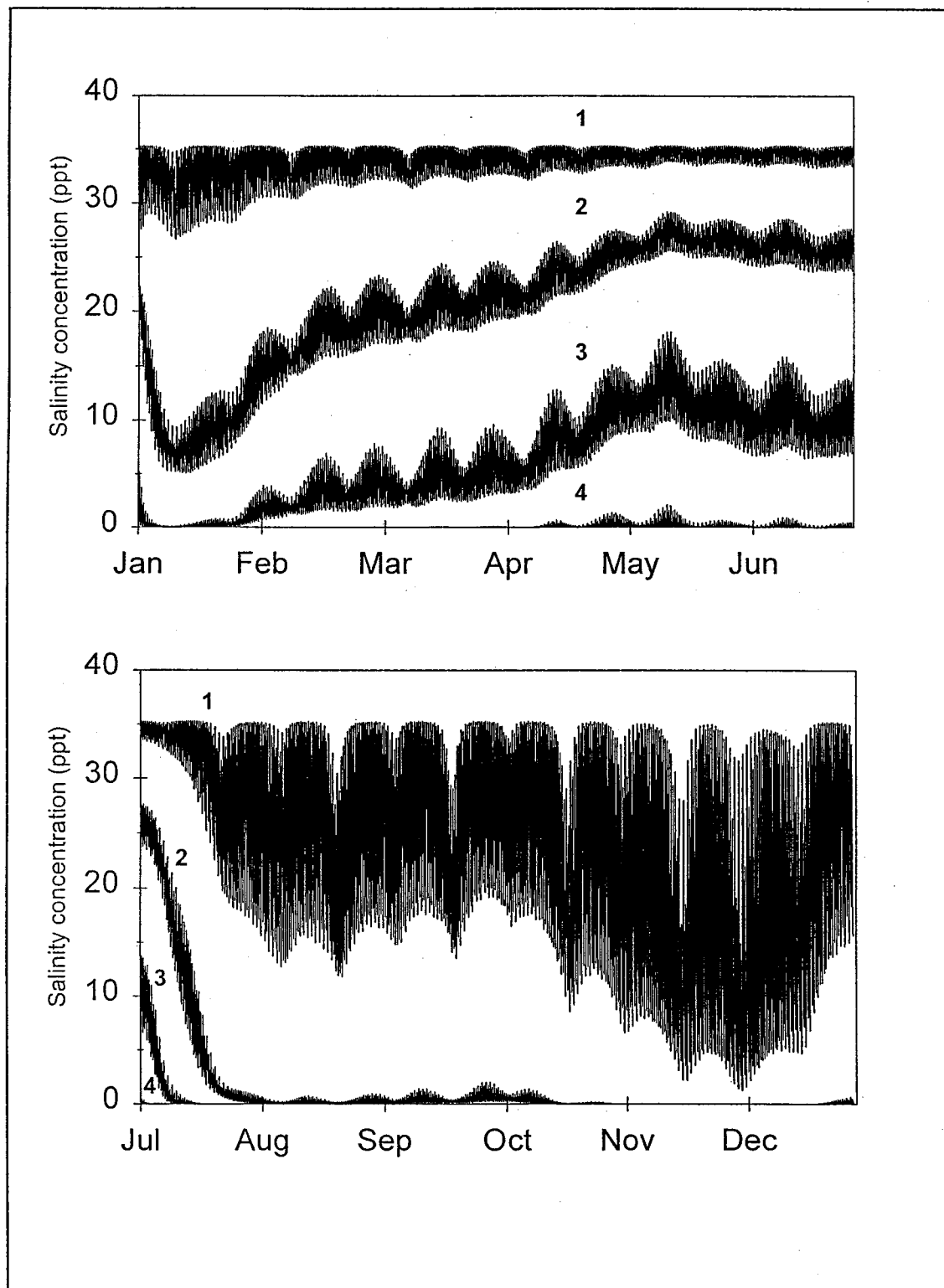


Figure 2.12 A: Simulated annual salinity concentrations (‰) before a flood event in the estuary at Stations 1 to 5.

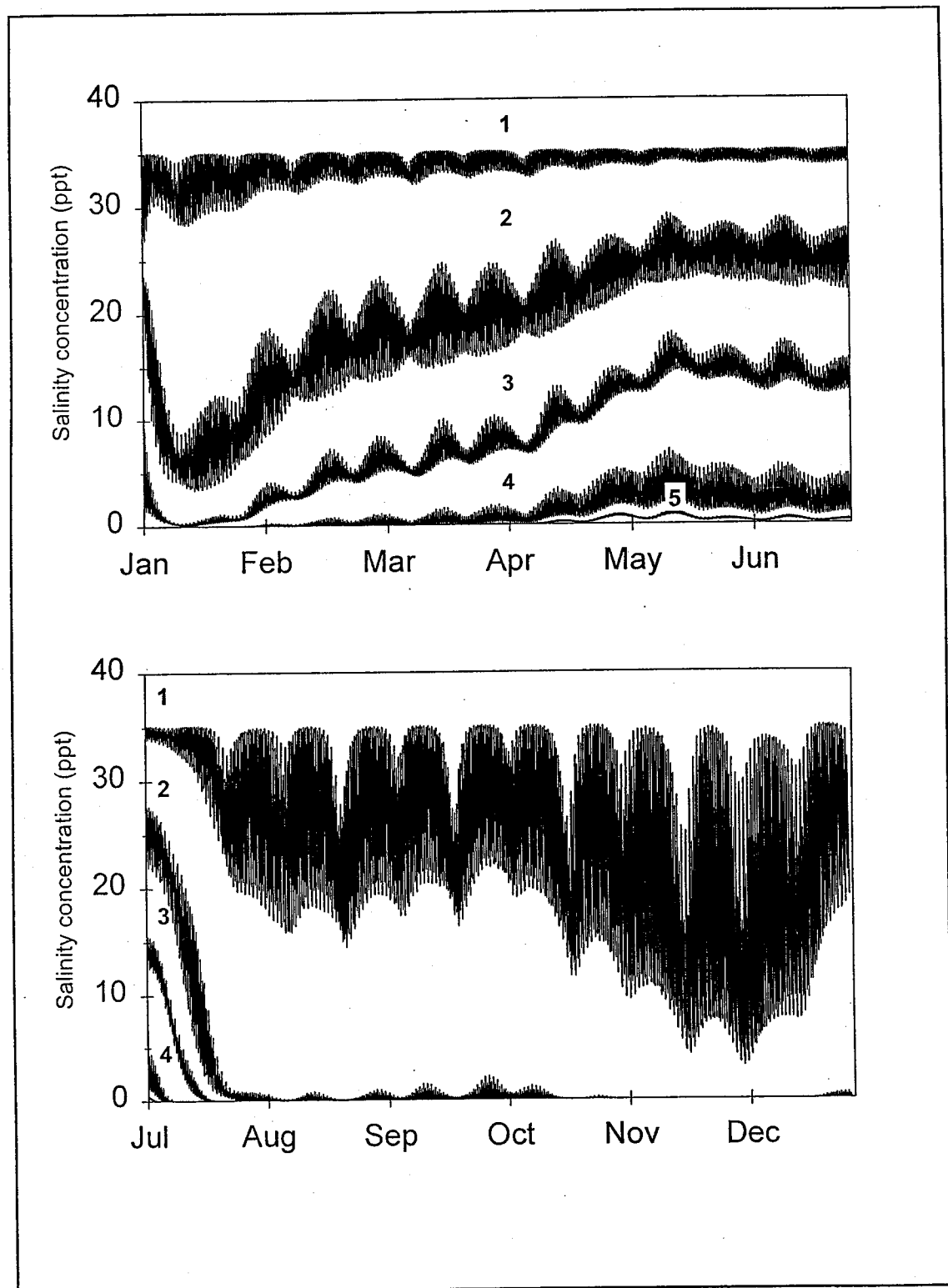


Figure 2.12 B: Simulated annual salinity concentrations (‰) after a flood event in the estuary at Stations 1 to 5.

Table 2.3 A: Estuary salinity concentration volumes (million m³) at different river flows: Spring tide for pre-1996 flood.

High tide	Salinity level							Total
	<5	<10	<15	<20	<25	<30	<35	
0.1	0.02	0.04	0.11	0.24	0.46	1.04	4.17	4.17
0.25	0.08	0.16	0.28	0.46	0.79	1.59	4.17	4.17
0.5	0.21	0.38	0.55	0.79	1.22	2.14	4.18	4.18
1	0.46	0.72	0.95	1.51	1.96	2.7	4.2	4.2
5	1.97	2.36	2.68	3	3.3	3.46	4.36	4.36
10	2.69	3.11	3.45	3.6	3.62	3.71	4.53	4.53
20	3.6	3.9	3.92	4.01	4.01	4.15	4.86	4.86
Low tide	Salinity level							Total
	<5	<10	<15	<20	<25	<30	<35	
0.1	0.02	0.04	0.11	0.22	0.48	1.1	3.64	3.64
0.25	0.08	0.17	0.29	0.47	0.76	1.59	3.65	3.65
0.5	0.21	0.37	0.52	0.77	1.23	2.11	3.66	3.66
1	0.48	0.7	1.01	1.33	2.03	2.65	3.68	3.68
5	1.94	2.4	2.64	2.99	3.5	3.81	3.81	3.81
10	2.64	2.99	3.42	3.77	3.97	3.97	3.97	3.97
20	3.4	4.07	4.26	4.26	4.26	4.26	4.26	4.26

Table 2.3 B: Estuary salinity concentration volumes (million m³) at different river flows: Neap tide for pre-1996 flood.

High tide	Salinity level							Total
	<5	<10	<15	<20	<25	<30	<35	
0.1	0.01	0.04	0.1	0.17	0.39	0.87	3.47	3.66
0.25	0.08	0.15	0.29	0.45	0.74	1.59	3.66	3.66
0.5	0.22	0.36	0.53	0.79	1.24	2.11	3.67	3.67
1	0.47	0.71	1	1.53	2.03	2.83	3.68	3.68
5	2.02	2.4	2.9	3.16	3.27	3.34	3.78	3.78
10	2.84	3.27	3.38	3.45	3.51	3.51	3.9	3.9
20	3.71	3.73	3.74	3.75	3.75	3.9	4.09	4.09
Low tide	Salinity level							Total
	<5	<10	<15	<20	<25	<30	<35	
0.1	0.01	0.03	0.1	0.17	0.38	0.9	3.34	3.34
0.25	0.08	0.14	0.28	0.41	0.72	1.5	3.28	3.28
0.5	0.22	0.37	0.54	0.77	1.29	2.09	3.29	3.29
1	0.47	0.7	0.98	1.52	2	2.85	3.3	3.3
5	2.02	2.43	2.95	3.41	3.41	3.41	3.41	3.41
10	3.04	3.27	3.54	3.54	3.54	3.54	3.54	3.54
20	3.87	3.87	3.87	3.87	3.87	3.87	3.87	3.87

Table 2.4 A: Estuary salinity concentration volumes (million m³) at different river flows: Spring tide for post-1996 flood.

High tide	Salinity level							Total
	<5	<10	<15	<20	<25	<30	<35	
0.1	0.06	0.17	0.32	0.75	1.08	1.92	4.75	4.75
0.25	0.16	0.3	0.68	0.92	1.35	2.15	4.75	4.75
0.5	0.32	0.73	0.97	1.25	1.76	2.38	4.76	4.76
1	0.86	1.16	1.47	1.85	2.22	2.76	4.78	4.78
5	2.17	2.39	2.61	2.9	3.64	3.84	4.92	4.92
10	2.59	2.85	3.43	3.78	3.93	4.07	5.07	5.07
20	3.92	4.08	4.08	4.12	4.14	4.14	4.47	4.47
Low tide	Salinity level							Total
	<5	<10	<15	<20	<25	<30	<35	
0.1	0.08	0.17	0.32	0.75	1.09	1.93	3.92	3.92
0.25	0.16	0.3	0.66	0.97	1.36	2.13	3.93	3.93
0.5	0.35	0.74	0.99	1.28	1.75	2.62	3.94	3.94
1	0.9	1.15	1.47	1.82	2.26	2.77	3.95	3.95
5	2.22	2.49	2.68	2.96	3.54	4.07	4.07	4.07
10	2.69	3	3.19	3.93	4.19	4.19	4.19	4.19
20	3.18	3.92	4.4	4.4	4.4	4.4	4.4	4.4

Table 2.4 B: Estuary salinity concentration volumes (million m³) at different river flows: Neap tide for post-1996 flood.

High tide	Salinity level							Total
	<5	<10	<15	<20	<25	<30	<35	
0.1	0.05	0.15	0.31	0.74	1.01	1.88	4.13	4.13
0.25	0.15	0.31	0.72	0.89	1.34	2.08	4.14	4.14
0.5	0.39	0.77	0.93	1.3	1.82	2.32	4.14	4.14
1	0.86	1.17	1.57	1.96	2.22	2.82	4.15	4.15
5	2.23	2.45	2.74	3.43	3.56	3.78	4.24	4.24
10	2.73	3.45	3.64	3.78	3.87	3.94	4.33	4.33
20	3.92	4.08	4.08	4.12	4.12	4.14	4.14	4.47
Low tide	Salinity level							Total
	<5	<10	<15	<20	<25	<30	<35	
0.1	0.05	0.14	0.32	0.74	1.02	1.86	3.63	3.63
0.25	0.14	0.32	0.71	0.89	1.33	2.1	3.63	3.63
0.5	0.38	0.77	0.94	1.29	1.81	2.29	3.63	3.63
1	0.87	1.15	1.58	1.93	2.21	2.94	3.65	3.65
5	2.23	2.46	2.68	3.46	3.73	3.73	3.73	3.73
10	2.68	3.51	3.84	3.84	3.84	3.84	3.84	3.84
20	4.07	4.07	4.07	4.07	4.07	4.07	4.07	4.07

Table 2.5A: Distances (km) from the mouth at which indicated salinity concentrations occur under different river flows at neap tide (N) and at spring tide (S) and averaged for high tide and low tide, for conditions before the flood of November 1996.

Flow ($\text{m}^3 \text{ s}^{-1}$)		Salinity (‰)							
		35	30	25	20	15	10	5	0
0.1	N	0	3.5	7	10	12.5	15.5	18	22
	S	0	4	8	12	13.5	17	19	23
0.25	N	0	3	6	8	11	12.5	15	21
	S	0	3.5	7.5	10	12.5	14	17	22
0.5	N	0	2.5	4.5	7	8	10	12	16
	S	0	3	5.5	8	10	12	14	19
1	N	0	2	3	4.5	7	8	10	15
	S	0	3	4	6	8	9	11	17
5	N	0	1	1	1.5	2	3	4	8
	S	0	2	2.5	3	3	4	5	9
10	N	0	0.5	0.5	1	1	1.5	2	5
	S	0	2	2	2.5	2.5	2.5	3	6
20	N	0	0	0	0.5	0.5	1	1.5	3
	S	0	1	2	2	2	2.4	2.5	5

Table 2.5 B: Distances (km) from the mouth at which indicated salinity concentrations occur under different river flows at neap tide (N) and at spring tide (S) and averaged for high tide and low tide, for conditions after the flood of November 1996.

Flow ($\text{m}^3 \text{ s}^{-1}$)		Salinity (‰)							
		35	30	25	20	15	10	5	0
0.1	N	0	4	7.5	11	13	17.5	21	>25
	S	0	5.5	9	12	16	18.5	22	>25
0.25	N	0	3.5	6.5	9	11	14	17.5	24
	S	0	4	7	9.5	12	15.5	18.5	24.5
0.5	N	0	3	5	7	9	11	14	23
	S	0	3.5	6	8	10	12	15	24
1	N	0	2.5	4	5	6.5	8	10.5	17
	S	0	3	4.5	6	7.5	9	11	18
5	N	0	1	1.5	2	2.5	3	4	7
	S	0	2	2.5	3	3.5	4.5	5	9
10	N	0	0.5	0.5	1	1	2	2	5
	S	0	2	2	2.5	2.5	3	3.5	6
20	N	0	0	0	0.2	0.3	1	1.5	3
	S	0	1	1	1.5	2	2	2.5	5

Table 2.6 A: Estimated monthly river flow, including the flow from the Loerie River, into the Gamtoos Estuary in $\text{m}^3 \text{s}^{-1}$, based on measured river flow and additional hydrological modeling.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1990	0.78	2.92	1.49	0.92	0.71	1.48	1.14	0.87	1.1	1.47	0.75	2.26
1991	0.96	0.22	0.13	0.73	0.1	0.16	0.27	0.42	0.49	2.56	1.48	6.96
1992	3.17	7.15	3.15	0.47	0.35	0.41	0.7	3.49	1.68	13.7	12.5	2.23
1993	0.87	4.91	0.18	0.49	0.47	12.9	4.87	1.88	70.2	22.5	2.11	9.7
1994	8.98	1.54	6.36	1.31	0.72	0.4	0.79	9.8	5.32	6.3	0.77	8.61
1995	14.3	3.47	5.11	11.8	11.2	6.48	0.65	0.64	0.9	0.83	2.26	8.41
1996	3.54	2.09	0.44	0.93	0.24	0.19	0.31	0.57	0.61	41.6	221	58.6
1997	37.7	2.76	2.61	65.4	22.1	30.1	17.7	9.88	4.91	1.08	0.6	0.21
1998	2.35	1.26	6.98	2.62	1.54	0.83	0.63	1.8	1.65	0.98	0.51	2.93

Table 2.6 B: Estimated monthly river flow, excluding the flow from the Loerie River, into the Gamtoos Estuary in $\text{m}^3 \text{s}^{-1}$, based on measured river flow and additional hydrological modeling.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1990	0.78	2.19	1.18	0.69	0.59	0.99	0.89	0.66	0.75	1.08	0.61	1.66
1991	0.71	0.18	0.1	0.07	0.09	0.13	0.23	0.35	0.39	1.93	1.21	5.11
1992	2.54	5.22	2.47	0.4	0.3	0.35	0.57	2.5	1.31	9.38	8.96	1.76
1993	0.7	4.84	0.17	0.38	0.38	8.52	3.69	1.47	60	20.4	1.65	6.62
1994	8.39	1.41	6.08	1.2	0.63	0.36	0.64	8.93	4.88	5.97	0.65	6.34
1995	12.6	2.65	4.54	11.2	10.8	6.38	0.6	0.46	0.42	0.36	1.73	7.52
1996	3	2.03	0.36	0.9	0.23	0.19	0.28	0.48	0.5	31.7	200	54.1
1997	36.9	2.73	2.58	59.2	19.8	26.5	15.7	8.59	4	0.8	0.52	0.2
1998	1.81	0.96	5.44	2.07	1.07	0.72	0.57	1.43	1.23	0.72	0.4	2.31

Table 2.6 C: Estimated monthly river flow into the Gamtoos Estuary from the Loerie River in $\text{m}^3 \text{s}^{-1}$, based on measured river flow and additional hydrological modeling.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1990	0.02	0.73	0.31	0.23	0.12	0.49	0.25	0.21	0.35	0.35	0.14	0.6
1991	0.24	0.04	0.02	0.01	0.02	0.03	0.04	0.07	0.09	0.63	0.27	1.87
1992	0.64	1.93	0.68	0.07	0.05	0.05	0.14	0.99	0.37	4.34	3.57	0.47
1993	0.17	0.07	0.01	0.11	0.1	4.4	1.18	0.4	10.2	2.15	0.46	3.08
1994	0.45	0.12	0.27	0.11	0.09	0.04	0.15	0.87	0.45	0.33	0.12	2.27
1995	1.71	0.81	0.56	0.64	0.37	0.1	0.05	0.18	0.48	0.47	0.53	0.88
1996	0.53	0.07	0.08	0.03	0.01	0.01	0.03	0.09	0.11	9.83	20.6	4.53
1997	0.76	0.03	0.03	6.21	2.24	3.58	2.02	1.3	0.9	0.29	0.07	0
1998	0.54	0.3	1.53	0.55	0.47	0.11	0.06	0.37	0.42	0.26	0.11	0.61

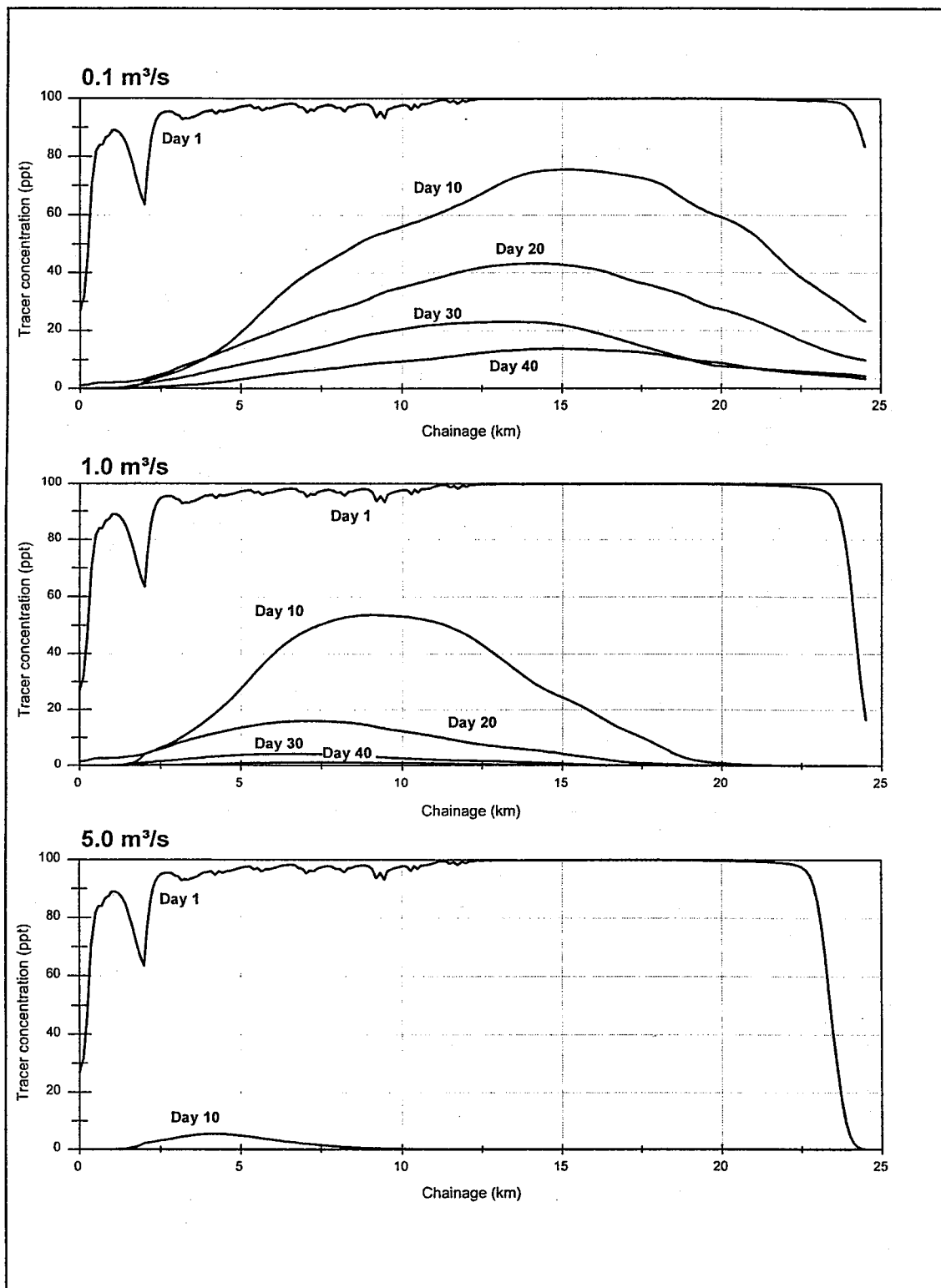


Figure 2.13: Simulated tracer distributions illustrating changes for indicated flow conditions after a 1996 flood event at 10 day intervals.

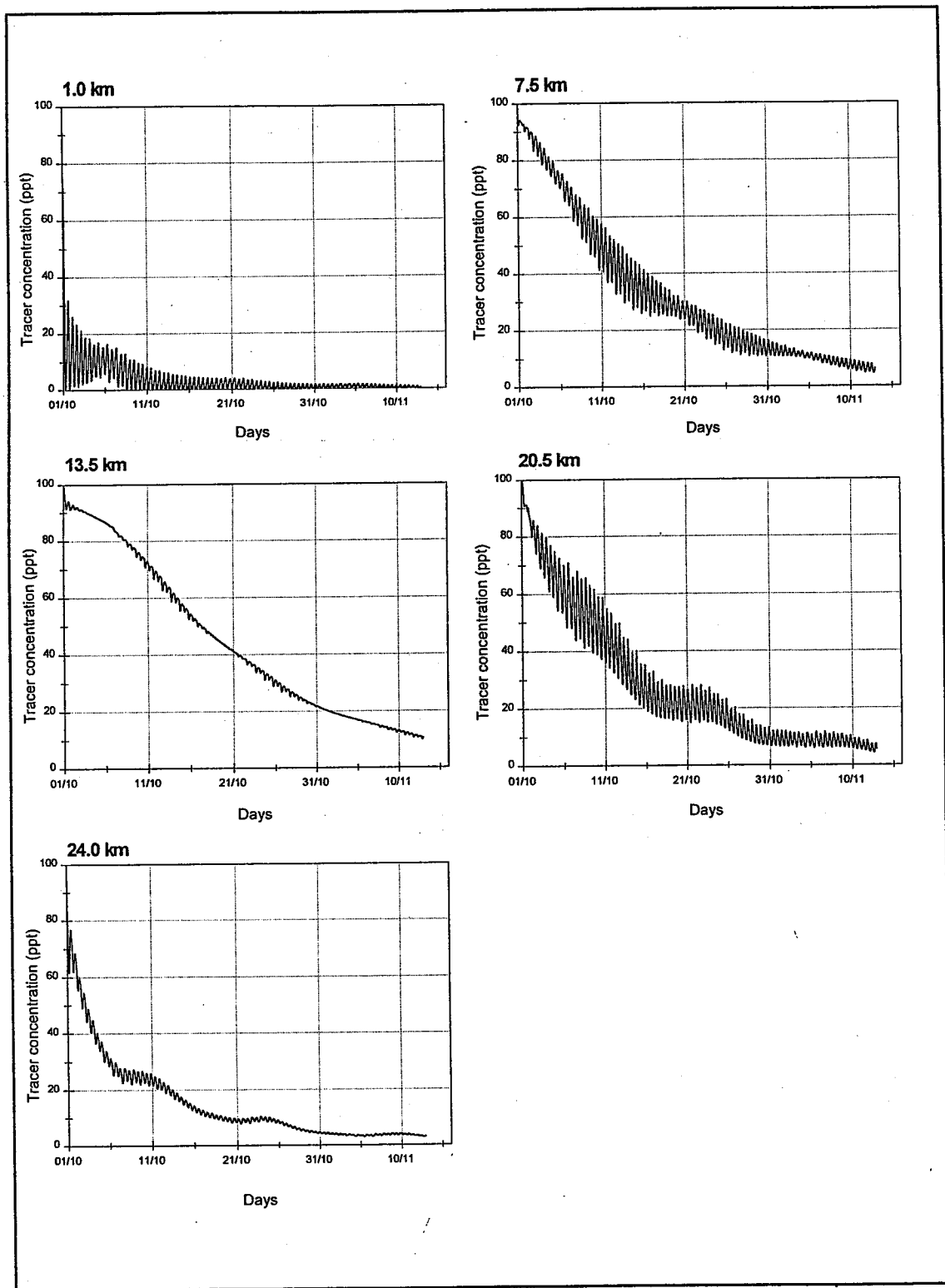


Figure 2.14 A: Simulated tracer concentrations at a river flow of $0.1 \text{ m}^3 \text{ s}^{-1}$ for cross-sections after a 1996 flood event.

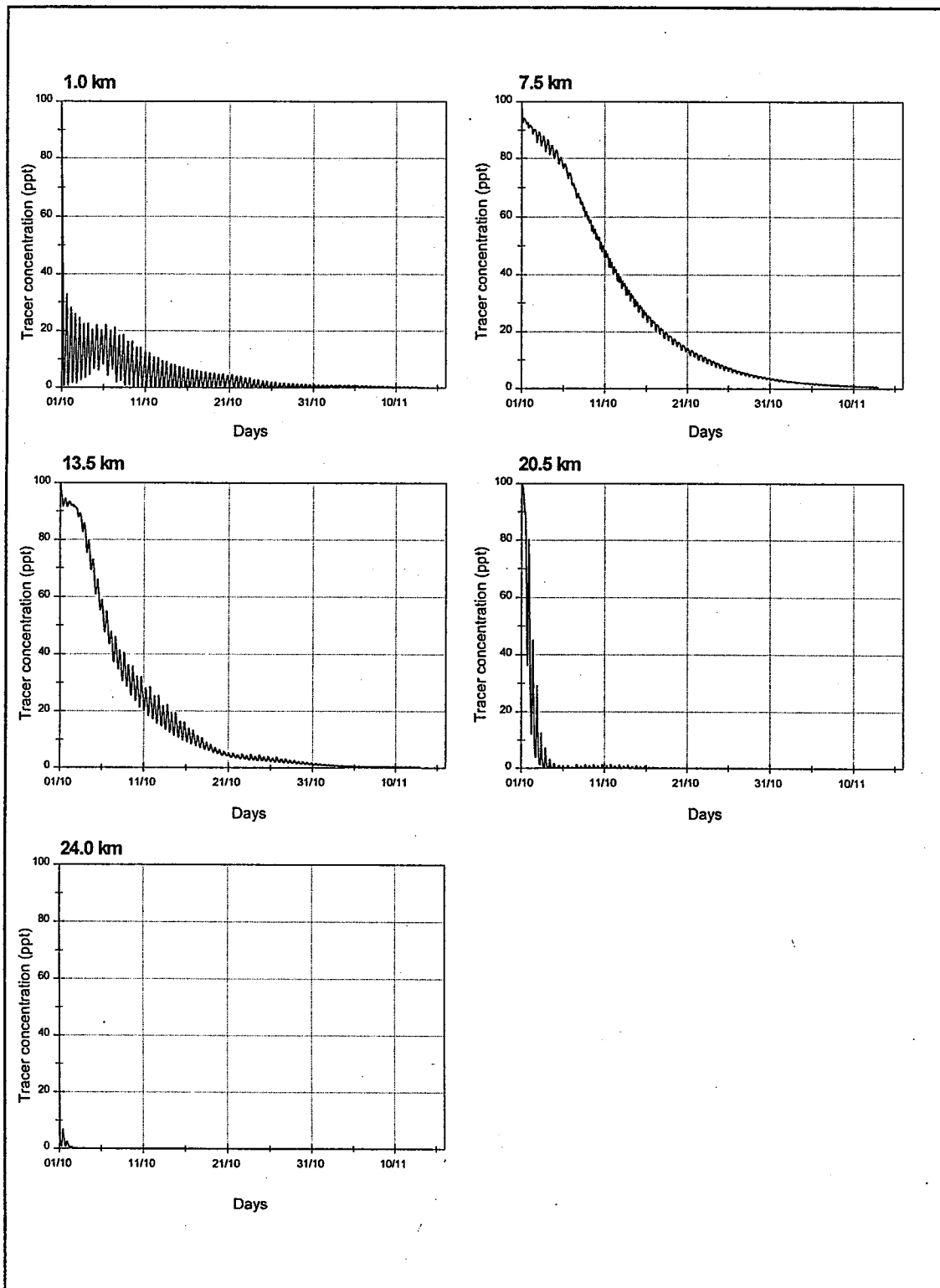


Figure 2.14 B: Simulated tracer concentrations at a river flow of $1.0 \text{ m}^3 \text{ s}^{-1}$ for cross-sections after a 1996 flood event.

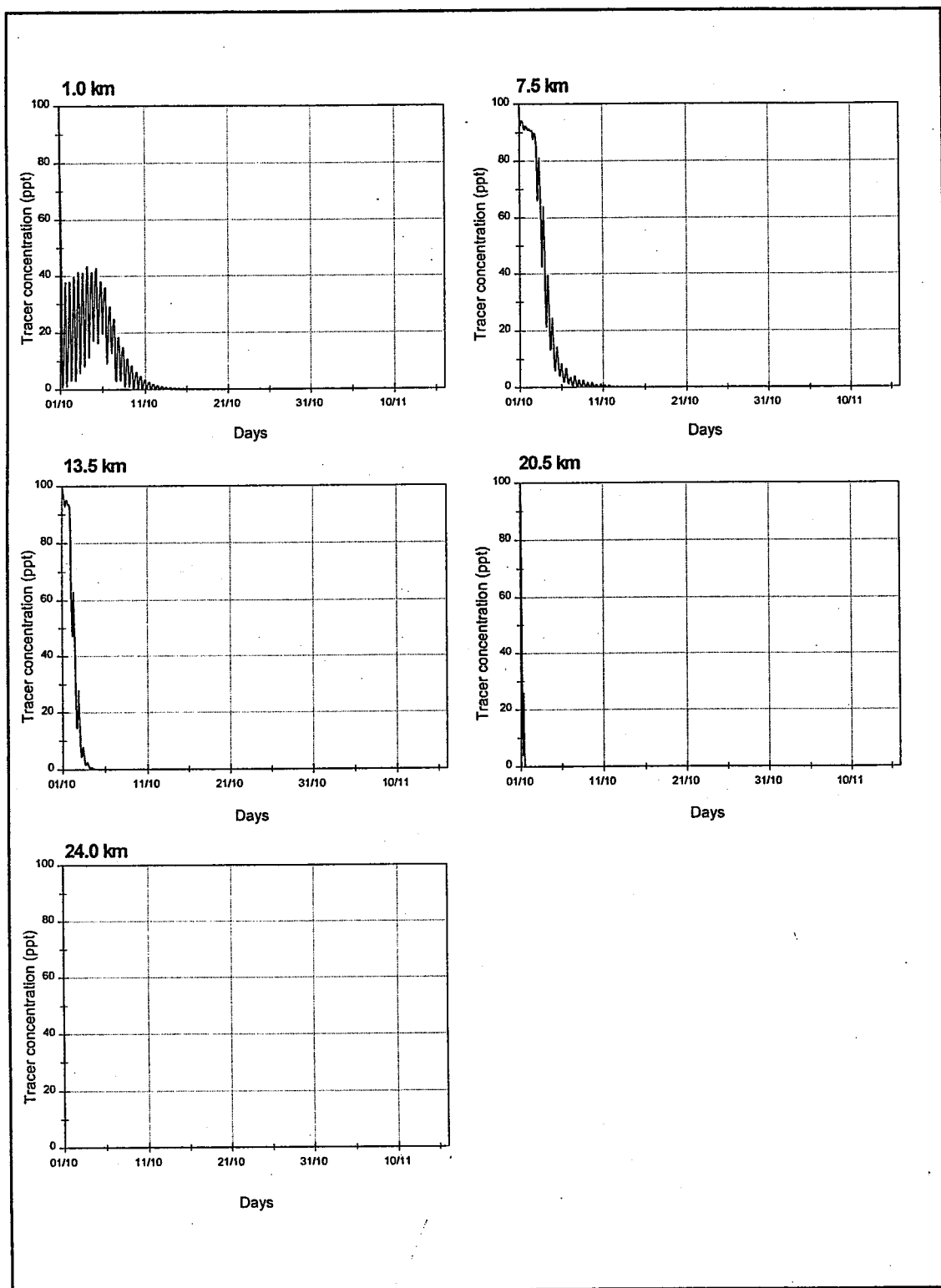


Figure 2.14 C: Simulated tracer concentrations at a river flow of $5.0 \text{ m}^3 \text{ s}^{-1}$ for cross-sections after a 1996 flood event.

2.3.6.1 Reduction in tracer concentrations at a river flow of $0.1 \text{ m}^3 \text{ s}^{-1}$

The longitudinal profiles of the salinity concentrations in Figure 2.13 show very clearly how the concentrations at different locations are being reduced with time. This reduction is highlighted in another way by the time history graphs in Figure 2.14 A. Near the mouth, the tracer concentrations are already strongly reduced within a few days because of the exchange of water with the sea. The reduction upstream is caused by the inflow of river water with a tracer concentration of zero. Higher concentrations can be observed in the middle of the estuary between 10.0 and 20.0 km from the mouth. However, the concentrations in the middle of the estuary have already dropped to levels of approximately 14.0 % after 40 days.

2.3.6.2 Reduction in tracer concentrations at a river flow of $1.0 \text{ m}^3 \text{ s}^{-1}$

The longitudinal profiles of the salinity concentrations at a river flow of $1.0 \text{ m}^3 \text{ s}^{-1}$ can be compared with those of the river flow of $0.1 \text{ m}^3 \text{ s}^{-1}$ in Figure 2.13 and in Figures 2.14 A and B. A much stronger reduction in concentrations is observed upstream because of the stronger river flow. Much lower levels are reached after 20, 30 and 40 days in the whole estuary. After 40 days, the maximum levels are approximately 2.0 % at about 7.0 km upstream of the mouth.

2.3.6.3 Reduction in tracer concentrations at a river flow of $5.0 \text{ m}^3 \text{ s}^{-1}$

The longitudinal profiles of the salinity concentrations at a river flow of $5.0 \text{ m}^3 \text{ s}^{-1}$ can be compared with those of the river flow of 0.1 and $1.0 \text{ m}^3 \text{ s}^{-1}$ in Figure 2.13 and in Figures 2.14 A to C. An even stronger reduction in concentrations is observed upstream because of the stronger river flow. Maximum levels of approximately 7.0% are present 8.0 km from the mouth after 10 days and the concentrations are below 1.0 % in the whole estuary after 20 days.

2.3.6.4 General

The reduction in concentrations of a conservative tracer in an estuary will be reduced with time if the concentrations are zero in the seawater and river water entering the estuary. However, in theory, an absolute zero concentration in the estuary will not be reached, even after a very long time. For this reason, a single retention time related to a certain flow does not exist. However, it is possible to use results such as those obtained with the model to estimate how long it will take until levels of, for example, 1% will be reached and what the effects of different river flows will be on the rate of reduction.

2.4 Conclusions and recommendations

The results obtained from the model simulations for pre- and post-flood topography conditions give a good overview of the effects of different river flow conditions on salinity distributions within the estuary. The results show the extent to which a decrease in river flow would result in increased salinities and an upstream shift of the salinity gradients within the estuary. The results, summarised in the previous graphs, have been used to interpret biotic structures within the estuary. Ecological requirements in terms of salinity distributions have been correlated with river flow requirements for the Gamtoos Estuary and the methodologies developed can also be applied to other estuaries. The abiotic and biotic linkages made during this project can be used for estuarine management support and will be especially useful in estimating the freshwater requirements of estuaries.

2.5 Acknowledgements

The cross-sections of the Gamtoos Estuary used for the configuration of the model were surveyed as part of a monitoring programme undertaken by the University of Port Elizabeth for the Department of Environmental Affairs and Tourism. Some hydrological data used in this report were provided by the Department of Water Affairs and Forestry. Tidal variation data, as recorded and predicted for Port Elizabeth, were provided by the Hydrographic Office, Silvermine. The estimates of the averaged monthly river flow into the estuary for the period 1990 to 1998 generated by Ninham Shand Consulting Engineers on a subcontract basis.

Susan Taljaard is thanked for her assistance with the analysis of field data. The cooperation and support provided by the University of Port Elizabeth, and especially the research teams (including the students) of Prof G Bate, Dr J Adams, Prof D Baird and Prof T Wooldridge during the field data collection, are much appreciated. Finally, the financial support of the project by the Water Research Commission is gratefully acknowledged.

2.6 References

Slinger, J.H., Taljaard, S., Rossouw, M. and Huizinga, P. 1998. Water quality modelling of estuaries.
Water Research Commission Report No. 664/1/98: 110 pp. + Appendices.

Chapter 3: Water column, interstitial and boundary layer nutrient status in and around the river-estuary interface (REI) zone of the Gamtoos Estuary

Pelagic Study: U. Scharler¹ and D. Baird¹

Benthic Study: H. Astill² and J. Adams²

¹Department of Zoology, University of Port Elizabeth

²Department of Botany, University of Port Elizabeth

Summary

The hypothesis of this study was as follows:

The quality and quantity of freshwater received at the river-estuarine interface region determines the nutrient status. The interface region is the most active of the estuarine reaches in terms of a sink/source behaviour for nutrients and thus influences the nutrient status of the whole system.

This was tested in two studies. The first study investigated the nutrient status of the water by determining the role of the Gamtoos River as a nutrient source under various freshwater inflow rates, determining the existence/influence of a river-estuarine interface (REI), and estimating the amount of freshwater inflow needed for a consistent nutrient input to the estuary. The second project addressed the relationship between bottom water and sediment porewater by investigating sediment porewater quality in the estuary, determining the influence of porewater nutrients on the water column nutrient status, and evaluating the influence of riverine inputs on the water column nutrient status.

Gamtoos River as a nutrient source and influence on water column nutrient status

Freshwater inflow was not ultimately responsible for variations in the nutrient status of the Gamtoos Estuary. Freshwater inputs under base flow conditions (0.35 to $1.6 \text{ m}^3 \text{ s}^{-1}$) did not influence mean estuarine nutrient concentrations significantly. Only phosphate concentrations were positively correlated with freshwater input (0.67 ; $p < 0.05$). Investigations of nutrient concentrations between spring and neap tides showed tidal movement also had no significant influence on nutrient status. As freshwater inflow had no direct significant influence on water column nutrient status, the amount of freshwater inflow required for consistent nutrient inputs to the estuary is unknown. However, it is predicted that a reduction in base flow will significantly influence water column nutrient status; the reasons for this are discussed in detail in this chapter.

Existence and influence of the REI on water column nutrient status

Although some nutrient species were found to occur at higher concentrations in the upper reaches (vertically averaged salinity of 17 ‰ and below), it was not possible to identify a REI confined within a certain salinity range for any of the nutrients measured. Overall, concentrations decreased towards the mouth in the case of nitrate, nitrite, dissolved organic carbon (DOC) and nitrogen (DON), particulate organic nitrogen (PON) and total particulate phosphorus (TPP). Concentrations of nitrate, nitrite, DON, PON, and TPP were higher in the upper reaches compared to the tidal head and the middle or lower estuarine reaches. Plots of nutrients against salinity did not reveal strong conservative behaviour for most nutrients, which points towards biological transformation and/or non-point source additions of nutrients along the estuarine gradient.

Sediment porewater quality and influence on water column nutrient status

As in the pelagic study, freshwater inputs were not found to significantly influence the inorganic nutrient status of benthic waters, while sediment porewater quality did significantly affect benthic water nutrient status. Porewater concentrations of nitrate and nitrite were 5 to 10 times higher than respective water column concentrations, while no concentration gradients existed between the water column and porewaters for ammonium and phosphate. Nitrate, nitrite, and ammonium porewater concentrations were significantly correlated with respective concentrations in the bottom waters of the water column (0.58, $p < 0.001$; 0.24, $p < 0.05$; -0.26, $p < 0.05$, respectively). Both porewater and bottom water phosphate concentrations were very low in the Gamtoos Estuary (both with respect to dissolved inorganic nitrogen concentrations and international observations), indicating a phosphate-limiting environment in this system. Such a phosphate-limiting environment would ensure any free reactive phosphorus at the sediment surface or in the water column would be utilised rapidly, making the detection of benthic fluxes difficult. It is for this reason that we think no significant relationship was found between sediment porewater phosphate and bottom water concentrations. These findings suggest that sediment porewater nutrients are a significant non-point source to water column nutrient status, throughout the estuary.

3.1 Introduction

Riverine freshwater input is often a major source of nutrients in estuaries, and the quantity and quality is a dominating factor determining the nutrient status and subsequent productivity of estuaries (Liss 1976; Aston 1980; Funicelli 1984). Thus, catchment practices may easily impact estuarine water quality, particularly through agricultural activities. Freshwater inputs from riverine sources generate a river-estuarine interface (REI), a region that is often recognised as having relatively increased

biochemical activity and biological biomass/diversity within an estuary (Morris *et al.* 1978; Montagna and Kalke 1992; Ahel *et al.* 1996; Rundle *et al.* 1998). Unfortunately, however, this region has received little attention in the scientific literature. It is therefore understandable that large-scale freshwater abstraction from an estuary has an indirect impact on water quality. This may be particularly influential to estuarine water quality and ecology in arid regions such as South Africa, where natural freshwater inputs are generally highly variable, with abstraction activities causing decreases in total freshwater received and the usual pulsing nature of deliverance (Puckridge *et al.* 1998).

In addition to allochthonous sources of estuarine nutrients such as freshwater inflow, autochthonous nutrient sources also exist. Sediment porewater nutrients have long been recognised as an important source of estuarine water column nutrient concentrations and primary productivity (Nixon *et al.* 1976; Boynton *et al.* 1980; Callender and Hammond 1982). At times, porewater nutrient concentrations have been found to influence estuarine water quality more than allochthonous sources from the surrounding catchment (Billen 1978; Lehtoranta 1998). As such, knowledge of porewater nutrient concentrations in combination with pelagic water quality is required in the planning of nutrient budgets and freshwater management proposals for estuarine systems.

The recently promulgated National Water Act of 1998 requires water reserve determinations for an estuary prior to any, or further, freshwater abstraction from a system. This legislation is an attempt to provide people with access to water resources whilst at the same time protecting estuarine ecology and health. Consequently many ecological research programmes determining freshwater requirements of estuarine systems have been, or are currently being, conducted throughout the country. Benthic nutrient processes are a significant component of estuarine ecology, as is the REI region. However, to date, the significance of these processes has not been considered in South African estuarine ecology. Thus, the importance of identifying the effects of freshwater inputs on both pelagic and benthic estuarine nutrient processes, in and around the REI has been recognised. The aim of this research was three-fold:

1. To investigate the influence of freshwater inputs on both water column and sediment porewater nutrient status.
2. To determine the influence of sediment porewater quality on water column nutrient concentrations.
3. To determine the existence and influence of the REI region on the nutrient status of the lower estuarine reaches.

The data presented here is the combination of two individual sampling programmes within the Gamtoos Estuary. The first investigated nutrient processes in the water column (conducted between November 1996 and November 1998), and the second investigated benthic nutrient processes (conducted between September 1999 and March 2000).

3.2 Materials and methods

3.2.1 Sampling and analysis

Physico-chemical measurements of the water column were conducted at 12 sampling points in the Gamtoos Estuary, on seven occasions (twice in November 1996, and once in October 1997, March, May, September and November 1998) (Figure 3.1). Monthly measurements of benthic waters (waters overlying the sediment) and sediment porewater were made at four sampling points between September 1999 and March 2000. Sites for the benthic study were selected according to sediment type along the horizontal estuarine axis (Heinecken 1981).

3.2.2 Water column

At 0.5 m intervals, salinity (‰) and temperature (°C) were measured throughout the water column using a CTDS Valeport Ser. 600. Dissolved oxygen and pH were measured at the surface, mid-water column, and bottom using a Jenway 9070 O₂-meter, and pH with a Jenway 3100, respectively. Light attenuation was measured using a Secchi Disk. Duplicate 500ml water samples were collected at the surface, 0.5 m, 1 m and at intervals of 1 m from thereon to the bottom. Samples were filtered in the field using Schleicher and Schuell (No. 6) glass fibre filters and kept on ice.

Dissolved nutrients were measured manually within 48 hours of collection. Nitrate + nitrite was measured using the method of Bate and Heelas (1975) as modified from Strickland and Parsons (1972). The nitrate data presented are nitrate + nitrite concentrations minus nitrite concentrations. Nitrite and soluble reactive phosphorus were measured according to Strickland and Parsons (1972), and ammonia using the method of Solórzano (1969). Total dissolved nitrogen (DON) and carbon (DOC) were analysed by the Municipality of Port Elizabeth using the Kjeldahl and Persulphate-Ultraviolet Oxidation methods (APHA 1992), respectively. Total Kjeldahl nitrogen, or particulate organic nitrogen (PON), was analysed after Bremner (1965), and particulate organic carbon (POC) was determined using the method of Strickland and Parsons (1972). Total suspended particulate organic phosphorus (POP) was measured by using the method of Black (1965).

After the length of the estuary had been covered during a sampling session, additional samples were taken in the lower salinity reaches [from a vertically average salinity of approximately 15 ‰, the River-Estuarine Interface (REI) to the tidal head]. These additional samples were taken at the following slack tide, so that at least two sets of samples (one for a high water slack tide, one for a low water slack tide) were available from the lower salinity reaches. These additional samples were used to describe the short-term variability in terms of nutrients within the estuary.

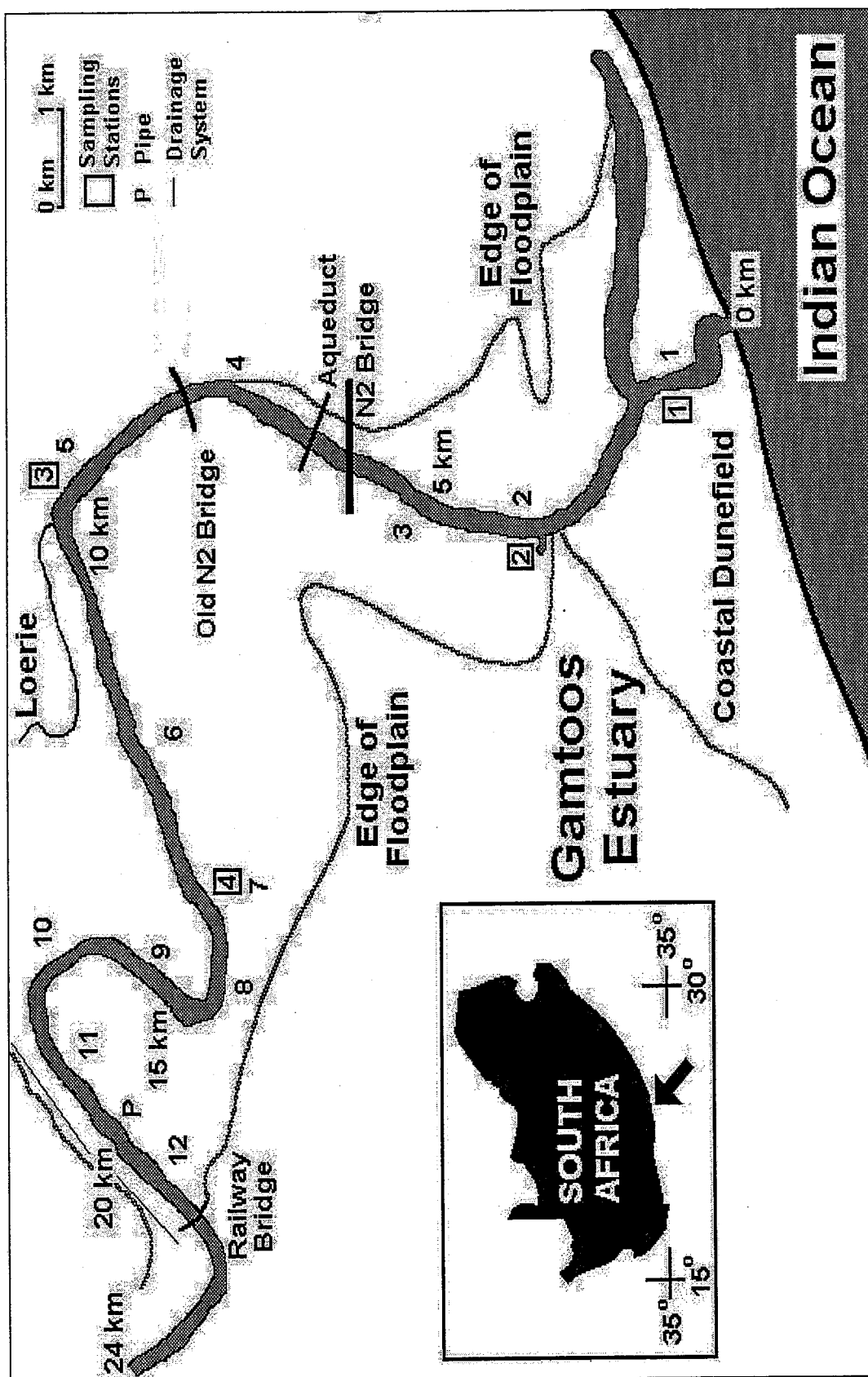


Figure 3.1: Location of sampling sites used to measure the water column (sites 1-12) and the benthos (square sites 1-4) in the Gamtoos Estuary.

3.2.3 Bottom waters

Bottom samples were collected just above the sediment surface, filtered in the field using Schleicher and Schuell (No. 6) glass fibre papers, and preserved immediately using mercuric chloride (Gearing 1991). Water samples were analysed manually for the inorganic nutrients nitrate + nitrite, nitrite, ammonium and phosphate using the methods described above.

Sediment samples were collected using 35mm (internal diameter) corers to a depth of 50mm. Three samples were bulked and squeezed for porewater extraction using a sediment squeezer modeled on the design of the Wildco 2214-c30 sediment squeezer. This process was performed three times at each site to obtain replicate samples. Porewater was filtered using Schleicher and Schuell (No. 6) glass fibre filter papers, preserved with mercuric chloride, and analysed for nitrate, nitrite, ammonium, and phosphate as described above. Additional sediment samples were collected for later analyses of redox potential, using platinum electrodes (Metrohm model 6.0451.100LL), and sediment organic content. Sediment organic content was determined by drying sediment at 100°C for 24h, then combusting at 550°C for 12h, after which organic content was measured as loss on ignition (% LOI; the difference in weight before and after combustion at 550°C) (Craft *et al.* 1991).

3.2.4 Rainfall and flow

3.2.4.1 Water column study

Freshwater inflow data for the Gamtoos Estuary were generated by the CSIR for both sampling sessions in November 1996 (see Chapter 2 for details). In October 1997, estuarine freshwater inflow was measured with a current meter. In 1998 freshwater input was estimated based on salinity profiles in the estuary in combination with simulated run-off conditions (Chapter 2).

3.2.4.2 Bottom waters study

Rainfall and water flow data were provided by the Department of Water Affairs, recorded at a sampling station located within the Gamtoos Estuary catchment. Rainfall data were measured at a station located at Kouga Dam (L8E001P), and flow data were obtained from a gauging station situated on the Groot River (L7H006), a primary tributary of the Gamtoos Estuary.

3.2.5 Data analyses

One-way repeated measures ANOVAs were used to analyse data for the independent factor site. Significant results were investigated with post-hoc Tukey tests. Data generated from water column samples measuring short-term variability in nutrient concentrations were analysed using a Wilcoxon matched pairs test, for every sampling session.

3.3 Results

3.3.1 Water column

3.3.1.1 Physico-chemical status

Freshwater inflow into the Gamtoos Estuary ranged from 0.4 to 1.6 m³ s⁻¹ under baseflow conditions. During a freshet, which occurred in November 1996, the average flow was estimated at 8 m³ s⁻¹ (see Chapter 2). The inflow at the agricultural drainage pipe ranged from 0.7 to 5 l s⁻¹, whilst the inflow rates of the Loerie tributary were generally negligible according to Schumann and Pearce (1997).

The mean salinity gradient (integrated over depth for all sampling sessions) in the Gamtoos Estuary ranged from approximately 30.6 ‰ at the mouth to 0.6 ‰ 19.5 km upstream (Figure 3.2). The salinity measured during the onset of the freshet event indicates 0 ‰ to within 10 km of the estuarine mouth. The temperature was slightly cooler at the mouth (17.5 °C) than at the tidal head of the estuary (19 °C) and slightly higher during the freshet (Figure 3.2). The dissolved oxygen content was lower at the mouth (7.3 mg l⁻¹) and increased to 8.4 mg l⁻¹ at the tidal head of the estuary (Figure 3.2). The percent oxygen saturation values showed a similar pattern and the measurements from the Loerie tributary were equivalent to those measured in the Gamtoos Estuary main channel (Figure 3.2). Dissolved oxygen content and saturation were lowered during the freshet by approximately 2 mg l⁻¹ near the mouth, and 3 mg l⁻¹ near the tidal head of the estuary. Measurements of pH ranged from 8.3 near the mouth to 8.4 at the tidal head, but were slightly lower in the middle reaches at 8 and less (Figure 3.2). Light attenuation showed a similar pattern to pH measurements in that slightly higher values were measured at both ends of the estuary (mouth = 77cm, tidal head = 58 cm) compared to the middle reaches (approximately 45cm) (Figure 3.2).

Stratification was evident for all physico-chemical parameters measured (Figure 3.3). Salinity stratification occurred on all sampling occasions and the difference between top and bottom salinities was most pronounced in the deeper lower and middle reaches. Temperatures were slightly lower at the bottom of the water column, and towards the mouth of the estuary. Oxygen and pH measurements indicated lower values near the bottom of the water column, and stratification was strongest in the middle reaches, but there was less difference between top and bottom values near the mouth and the tidal head of the estuary.

3.3.1.2 Nutrient status

3.3.1.2.1 Under base flow conditions (0.4 to 1.6 m³ s⁻¹)

Phosphate

Mean phosphate concentrations were very low relative to ammonium and nitrate concentrations. Mean phosphate concentrations were approximately 0.3 µM near the estuarine mouth and tidal head, while concentrations were slightly higher (ca. 0.5 to 0.7µM) in the middle reaches of the estuary

(Figure 3.4). Concentrations measured in the Loerie tributary were even lower ($0.3\ \mu\text{M}$) than in the Gamtoos Estuary main channel ($0.5\ \mu\text{M}$). Mean phosphate concentrations measured at the agricultural drainage pipe ($3.4\ \mu\text{M}$) were an order of magnitude higher than water column measurements.

Nitrite

As for phosphate, mean nitrite concentrations were also very low, decreasing from $2\ \mu\text{M}$ near the tidal head to $1\ \mu\text{M}$ at the estuarine mouth. Concentrations were highest ($2.9\ \mu\text{M}$) in the upper reaches between 16 and 18 km from the mouth (Figure 3.4). Mean nitrite concentrations were lower in the Loerie tributary ($0.7\ \mu\text{M}$) than in the Gamtoos Estuary main channel, and high in water flowing from the agricultural drainage pipe ($12.5\ \mu\text{M}$).

Nitrate

Mean nitrate concentrations steadily increased from the estuarine mouth ($12.5\ \mu\text{M}$), to the tidal head, where they were an order of magnitude higher ($102.3\ \mu\text{M}$) (Figure 3.4). Mean nitrate concentrations in the Loerie tributary ($11.6\ \mu\text{M}$) were lower than those measured in the Gamtoos Estuary main channel ($17.6\ \mu\text{M}$) at the confluence, and high in the water spilling from the agricultural drainage pipe ($195.2\ \mu\text{M}$).

Ammonium

Mean ammonium concentrations were higher at the mouth ($9.3\ \mu\text{M}$) than at the head ($5.5\ \mu\text{M}$) of the Gamtoos Estuary, but highest between 13 and 14.5 km from the mouth (11.7 and $12.2\ \mu\text{M}$, respectively) (Figure 3.4). High concentrations (127.1 and $119\ \mu\text{M}$) were measured on two sampling occasions 5.5 km from the mouth. Mean ammonium concentrations were similar in the Loerie tributary ($11\ \mu\text{M}$) to those in the Gamtoos Estuary main channel, but higher in the water spilling from the agricultural drainage pipe ($27.9\ \mu\text{M}$).

Dissolved organic carbon (DOC)

Mean DOC concentrations decreased from $908\ \mu\text{M}$ at the tidal head of the estuary to $484\ \mu\text{M}$ at the mouth (Figure 3.4). Similar to nitrate, mean DOC concentrations decreased by about $200\ \mu\text{M}$ within the first 5.5 km of the upper estuarine reaches, and a further $200\ \mu\text{M}$ along the remaining 14 km down-estuary. Mean DOC concentrations in the Loerie tributary were similar to those in the Gamtoos Estuary main channel ($597\ \mu\text{M}$), but considerably higher in waters of the agricultural drainage pipe ($4819\ \mu\text{M}$).

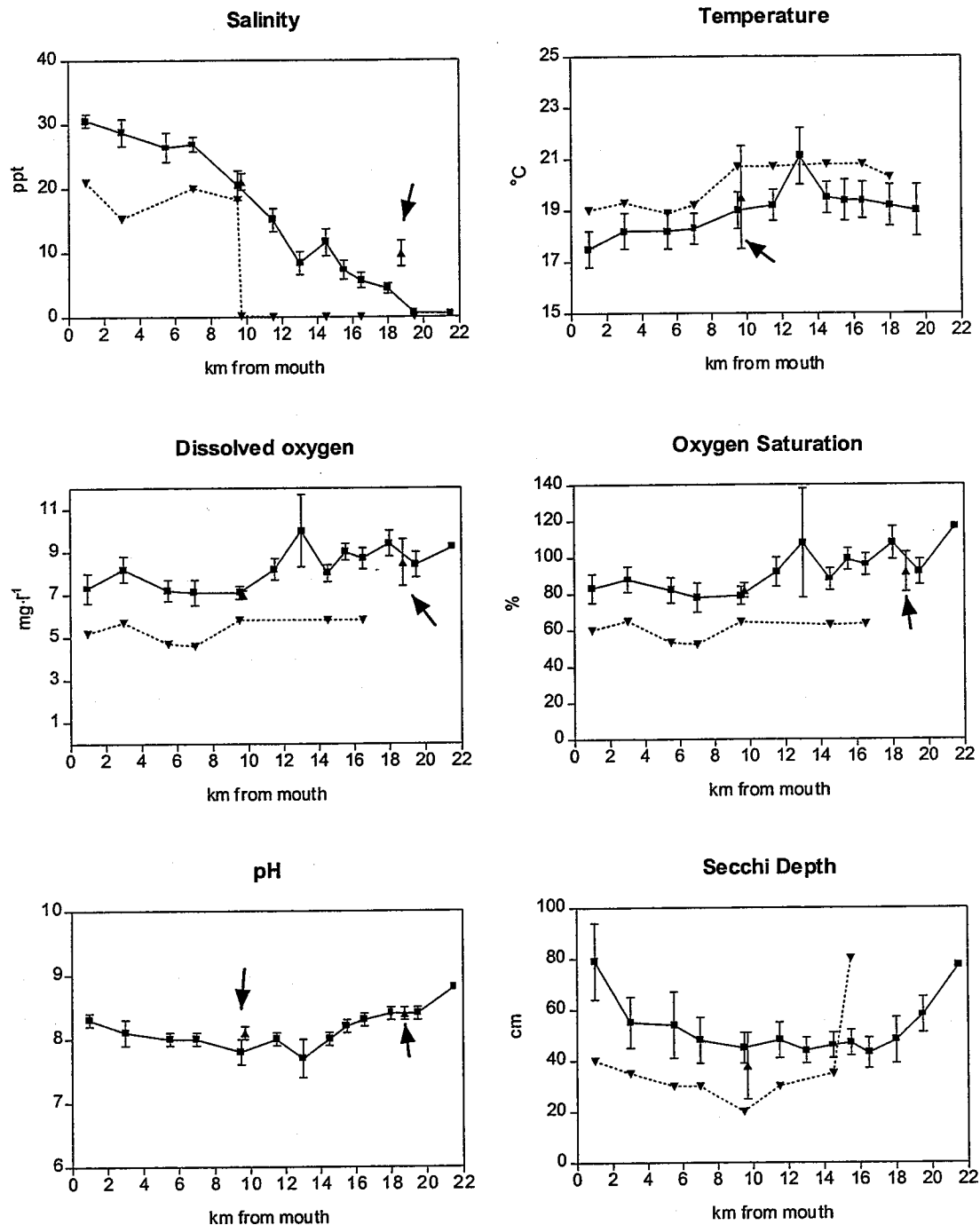


Figure 3.2: Mean values ($n = 42$ to 56 ; $\pm$ SE) of salinity (‰), temperature ($^{\circ}\text{C}$), dissolved oxygen (mg l^{-1}), oxygen saturation (%), pH, and light attenuation (Secchi Disk Depth in cm) in the Gamtoos Estuary. The solid line denotes mean concentrations during base flow conditions, the dotted line concentrations during the freshet. Arrows indicate concentrations in the Loerie tributary (ca. 9.6 km from the mouth) and the agricultural drainage pipe (ca. 18.7 km from the mouth).

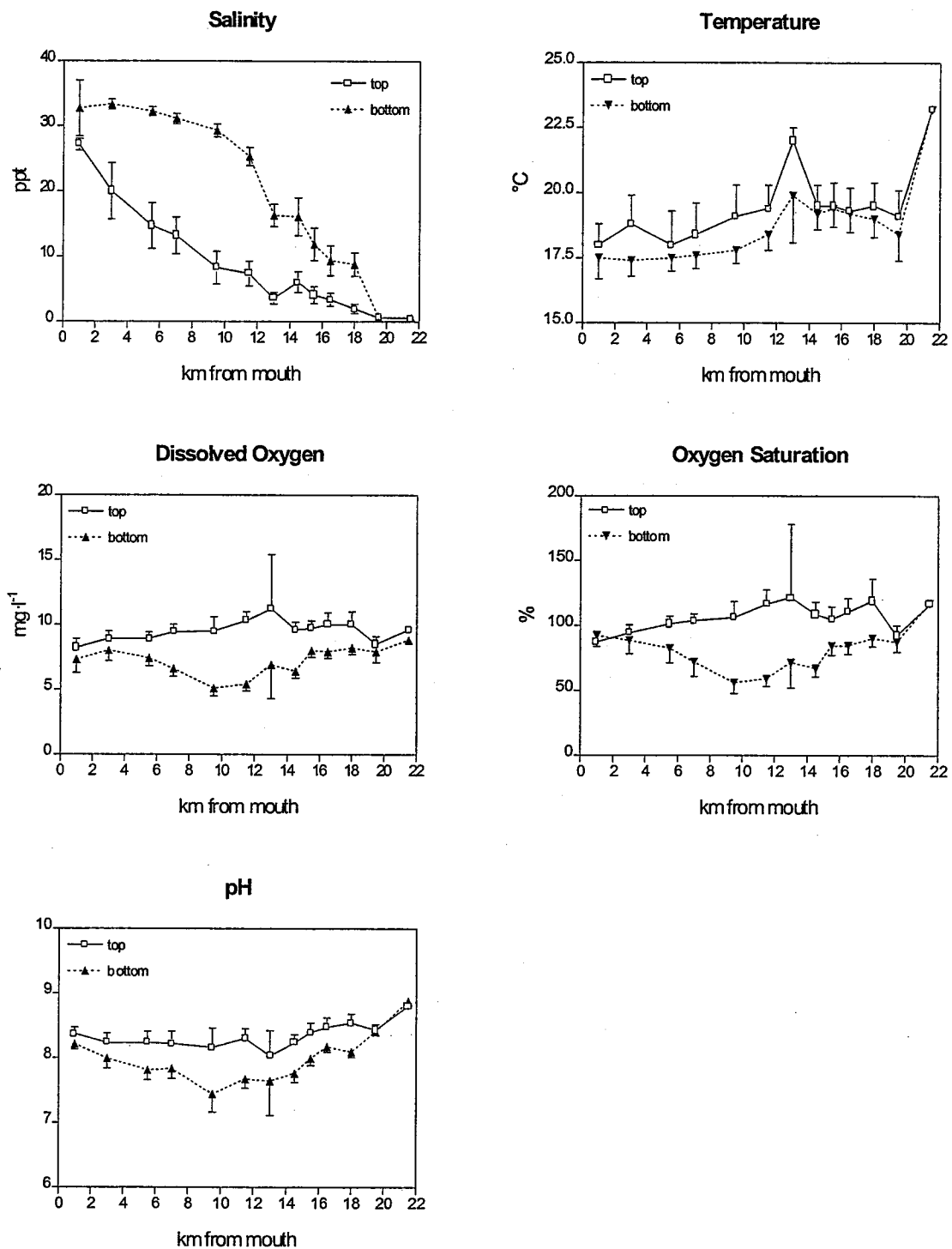


Figure 3.3: Mean values ($n=42$ to 56 ; $\pm$ SE) for the top (—) and bottom (---) of the water column, for salinity (‰), temperature ($^{\circ}\text{C}$), dissolved oxygen (mg l^{-1}), oxygen saturation (%), and pH, measured in the Gamtoos Estuary.

Dissolved organic nitrogen (DON)

Mean concentrations of DON ranged from 45 to 48 μM in the upper estuarine reaches (between 14.5 and 18 km), and decreased at the tidal head of the estuary (30.9 μM) and near the mouth (26.5 μM) (Figure 3.4). The mean concentration 3 km upstream of the mouth was 44 μM and no obvious trend was apparent. Mean DON concentrations in the Loerie tributary were lower (20.8 μM) than in the Gamtoos Estuary main channel, and high in the agricultural drainage pipe (107.6 μM).

Particulate organic carbon (POC)

Mean POC concentrations increased from the head (955 μM) to the mouth (3894 μM) of the estuary on all sampling occasions contrary to all other nutrients measured (Figure 3.4). Concentrations in the Loerie tributary were lower (2714 μM) than those at the confluence with the Gamtoos Estuary main channel (3665 μM). Concentrations in the agricultural drainage pipe were 2192 μM , higher compared to nearby estuarine waters (1839 μM).

Particulate organic nitrogen (PON)

Mean PON concentrations also showed a similar pattern to nitrate concentrations with noticeably higher concentrations measured in the upper estuarine reaches (Figure 3.4). Mean concentrations ranged from 15.8 μM at the tidal head (19.5 km from the mouth) to 24.3 μM 16.5 km from the mouth to 6.6 μM at the estuarine mouth. Concentrations in the Loerie tributary were slightly lower compared to the Gamtoos Estuary main channel (11.5 μM), and lower again in the agricultural drainage pipe (9.9 μM) compared with the adjacent estuary (19.1 μM).

Total particulate phosphorus (TPP)

Mean TPP concentrations were highest between 15.5 and 18 km from the estuarine mouth (6.4 to 7 μM), while lower at the tidal head (4.6 μM) and the mouth (2.1 μM) (Figure 3.4). Mean concentrations in the Loerie tributary (ca. 3.9 μM) were similar to those in the Gamtoos main channel, as were concentrations in waters of the agricultural drainage pipe (ca. 5.8 μM).

Water column nutrient patterns

Variability in nutrient concentrations was relatively high for all inorganic dissolved nutrients (Figures 3.5), while lowest for DOC and DON at approximately 40 % or less (Figures 3.5). For most nutrients measured there was no distinct pattern along the longitudinal axis of the estuary, although nitrate and nitrite show a lower variability in their upper reaches (Figures 3.5). The variability in POC concentrations appears to decrease towards the mouth (Figure 3.5).

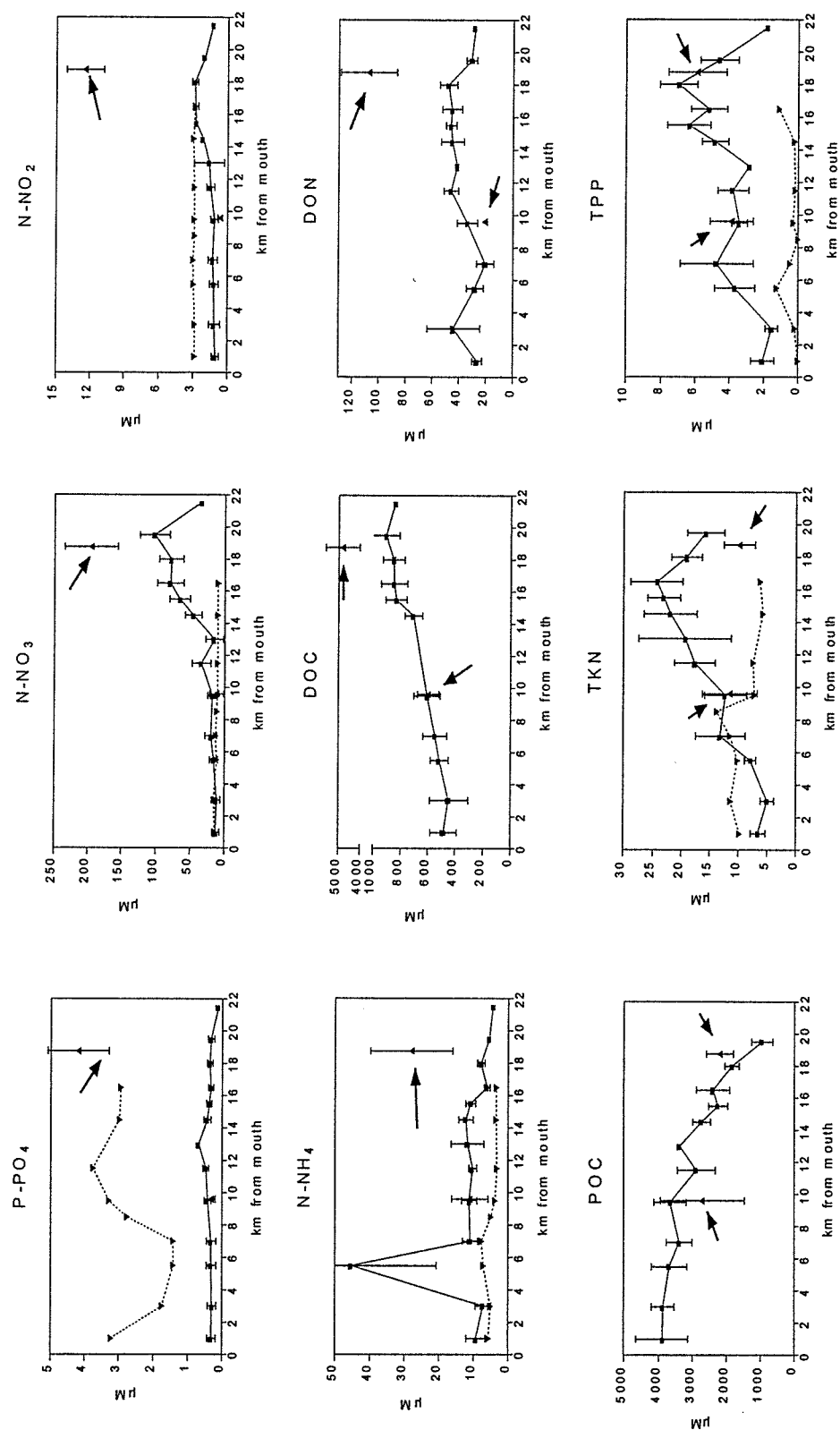


Figure 3.4: Mean nutrient concentrations (µM) ($n = 42$ to 56 ; $\pm$ SE) measured in the Gamtoos Estuary under baseflow conditions (—) and during the freshet (---). Arrows indicate mean nutrient concentrations in the Loerie tributary (ca. 9.6km from the estuary mouth) and in the agricultural drainage pipe (ca. 18.8 km from the estuary mouth).

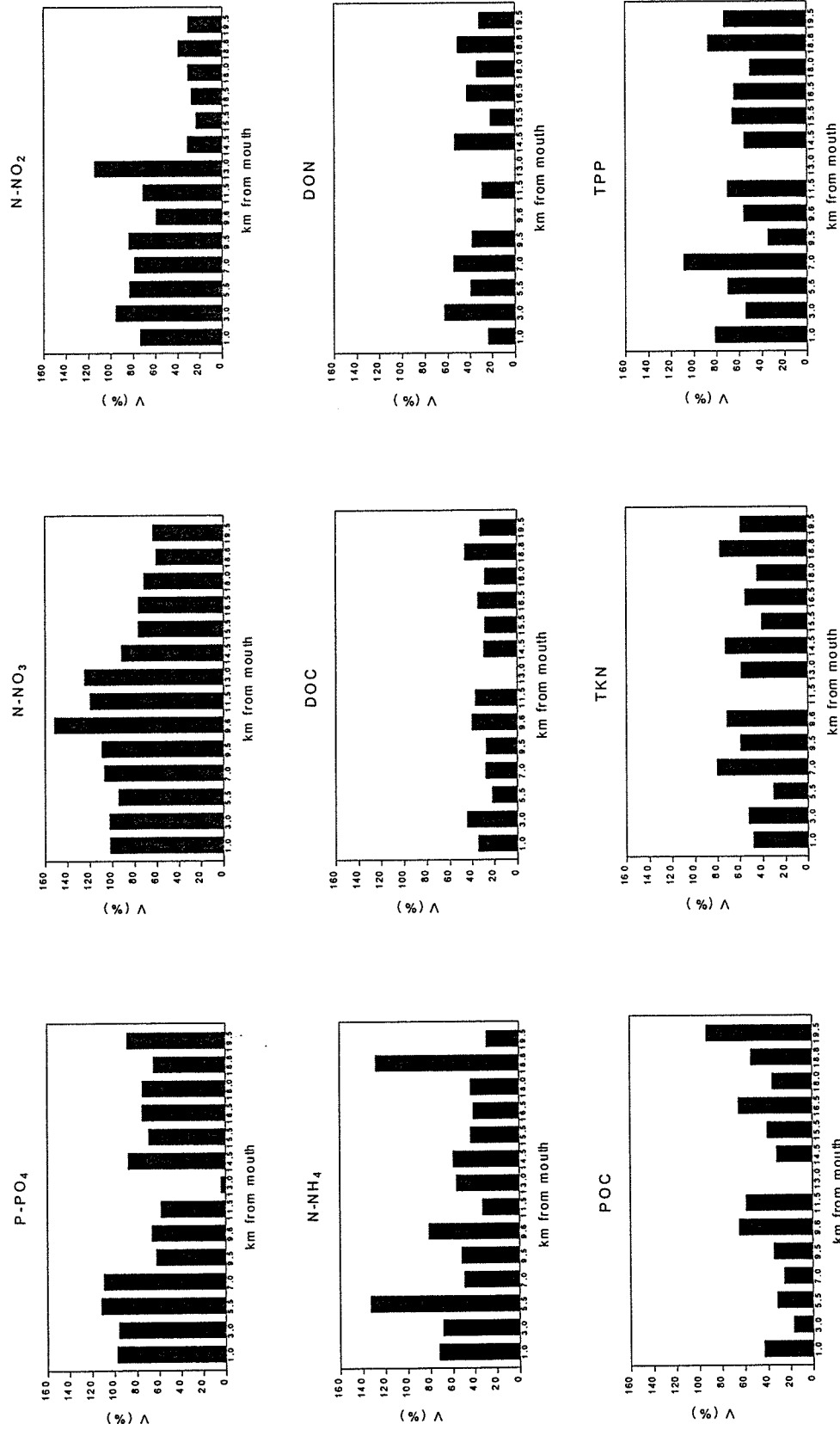


Figure 3.5: The coefficient of variation for all nutrients measured. All data were generated from baseflow data sets only. The columns at 18.8 km in the figures are data from the agricultural drainage pipe.

In summary, nitrate, nitrite, DOC, DON, PON and TPP concentrations generally decreased towards the mouth. Nitrate, nitrite, DON, PON, and TPP concentrations were higher in the upper estuarine reaches relative to those at the tidal head and the middle or lower estuarine reaches. The Loerie tributary showed similar or slightly lower concentrations for all nutrients compared to those measured at the confluence with the Gamtoos Estuary main channel. Waters from the agricultural drainage pipe 18.8 km from the estuary mouth showed distinctly higher concentrations compared to concentrations in the adjacent estuary with the exceptions of PON (concentrations were lower), and TPP (concentrations were equivalent).

3.3.1.2.2 During the freshet ($8 \text{ m}^3 \text{ s}^{-1}$)

During the freshet in November 1996, water quality was sampled at its onset and freshwater flow into the Gamtoos Estuary was approximately $8 \text{ m}^3 \text{ s}^{-1}$ (see Chapter 2). Mean salinity measurements had dropped to 0‰ 9.6km upstream from the estuary mouth (Figure 3.2). The freshet had therefore partially flushed out estuarine waters, while the lower 9.6 km were still euryhaline to marine. This flushing was also well represented in PON mean concentrations (Figure 3.4). Mean concentrations of phosphate and nitrite were elevated during the freshet (Figure 3.4), whereas nitrate, ammonium, PON and TPP decreased during the event (Figure 3.4).

3.3.1.2.3 C:N:P ratios

Inorganic dissolved N:P ratios (DIN:P-PO_4) generally decreased towards the mouth of the Gamtoos Estuary, reflecting mainly decreases in nitrate and nitrite concentrations with distance downstream (Figure 3.6). Exceptions to this trend were observed on the 4th November 1996 (Figure 3.6), where inorganic N:P ratios increased towards the mouth and on the 12th November 1996 (Figure 3.6), where the flushing effect at the onset of the freshet decreased nitrate and ammonia concentrations and increased phosphate concentrations. Nitrite concentrations during the flushing event were slightly elevated, but there was no longitudinal gradient apparent. The inorganic N:P ratios differed considerably between sampling occasions and exceeded the Redfield ratio in most cases. Fluctuations in inorganic N:P ratios can generally be attributed to changes in nitrate concentrations, which show the strongest correlation. This is not surprising, as nitrate showed the highest concentrations of the DIN components, especially in the upper estuarine reaches and a considerable amount of variation. Ammonium and phosphate concentrations showed lower, but significant correlations to DIN:P-PO_4 ratios (Table 3.1).

Total particulate N:P ratios were considerably lower compared to the inorganic N:P ratios (Figure 3.7) with only a few ratios close to 100. The particulate ratios showed no clear pattern along the longitudinal axis of the estuary. Particulate C:N ratios in general increased towards the estuarine mouth, reflective of the pattern of POC concentration in the estuary (Figure 3.8). The ratios exceeded the Redfield ratio, especially in the middle and lower reaches.

Table 3.1: Correlation/regression coefficients of inorganic N:P ratios with the inorganic nutrients measured (= 0.05, ** = $p < 0.001$).

Nutrient	μM	SD	r	r^2	t	p	n	p-level
PO ₄	0.4	0.3	-0.47	0.22	-9.4	< 0.001	313	**
NO ₃	40.2	48.3	0.75	0.56	19.9	< 0.001	313	**
NO ₂	1.8	1.2	0.36	0.13	6.8	< 0.001	313	**
NH ₄	12.9	47.3	0.47	0.22	9.5	< 0.001	313	**
DIN	54.9	66	0.89	0.8	35	< 0.001	313	**

3.3.1.2.4 Point sources

There are three potential point sources of nutrients to the Gamtoos Estuary, namely the Gamtoos River itself, the Loerie tributary entering the Gamtoos Estuary 9.5 km from the mouth, and an agricultural drainage pipe spilling into the Gamtoos Estuary 18.8 km from the mouth. The pipe drains an agricultural area of 50 ha to a depth of 1m. Further nutrient inputs to the Gamtoos Estuary are fertilizers, which constitute non-point sources, leached from adjacent agricultural fields on the floodplain of the estuary.

The rate of nutrient input from the Gamtoos River and the agricultural drainage pipe is shown in Table 3.2. The highest variation ($\geq 100\%$) in input at the tidal head of the Gamtoos Estuary was apparent from phosphate, nitrate, and particulate carbon and phosphorus. Overall, the highest input of a nitrogen component was nitrate, followed by dissolved and particulate organic nitrogen. Total particulate phosphorus inputs were a magnitude higher compared to inorganic dissolved phosphorus inputs.

Although nutrient concentrations were higher in water flowing from the agricultural drainage pipe, the absolute volume reaching the estuary via the pipe was lower compared to inputs at the tidal head because of the smaller volume of water discharging from the pipe. Compared to the input at the tidal head, the nutrient input from the drainage pipe was less variable for phosphate, nitrate, DOC and POC, but showed more variation for all other nutrients.

The Loerie tributary showed similar nutrient concentrations to the Gamtoos Estuary and is thus assumed to not be a major nutrient contributor to the estuary. This assumption is based on a few measurements only and future studies might prove otherwise. The variable nature of nutrient input during baseflow conditions from the Gamtoos River can be seen from correlations of freshwater input and nutrient input at the tidal head of the estuary (Table 3.3 A). Only phosphate concentrations were significantly (and positively) correlated with freshwater input with a correlation coefficient of 0.67. All other nutrients did not show a linear relationship to freshwater inflow. Only one significant correlation at the agricultural drainage pipe was found with DON, which showed an r value of 0.78 (Table 3.3 B).

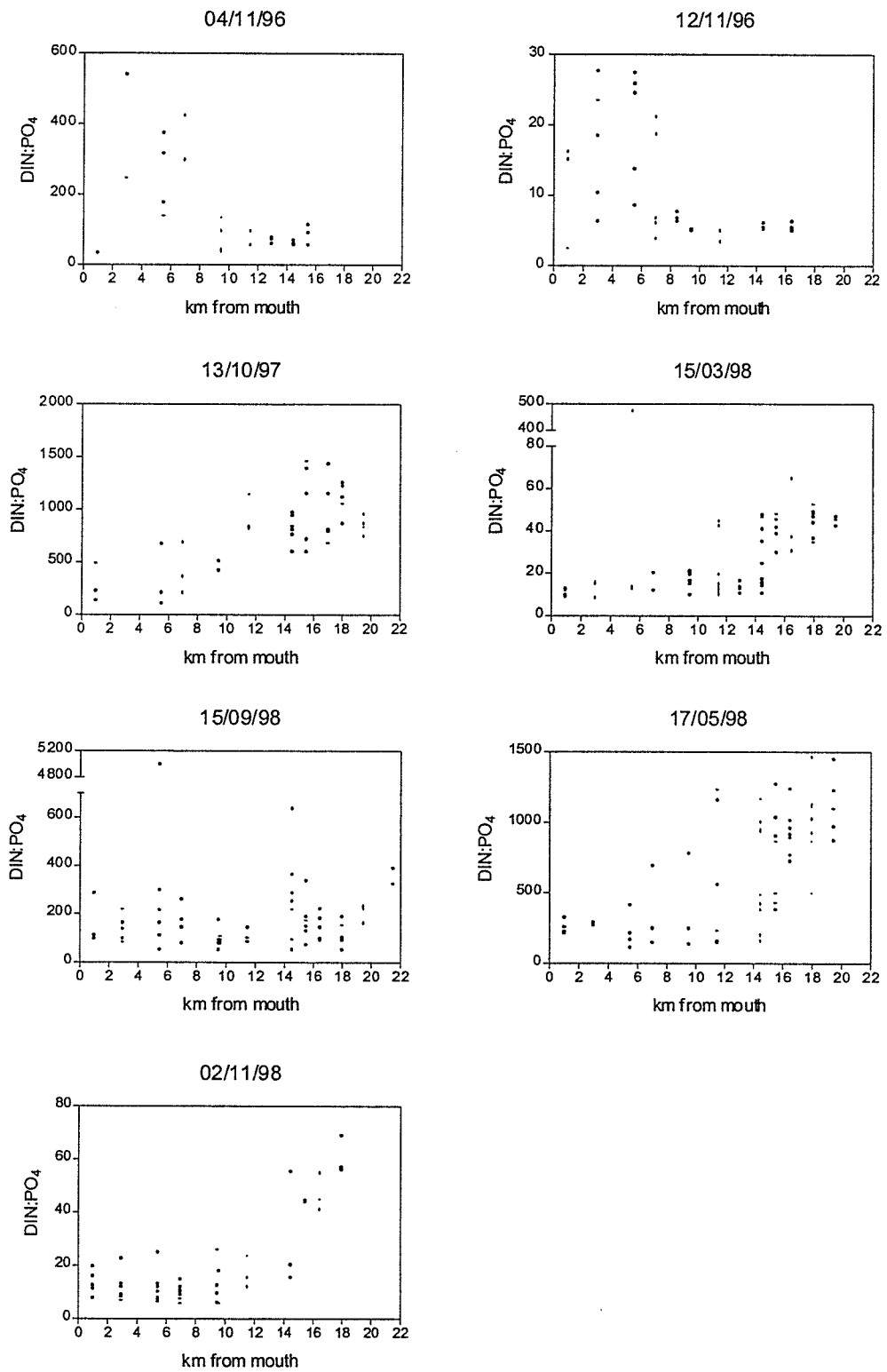


Figure 3.6: Dissolved inorganic N:P ratios during the freshet (13/10/97), and under baseflow conditions.

Table 3.2: Flow rate at the head of the estuary (in $\text{m}^3 \text{s}^{-1}$) and at the drainage pipe (in l s^{-1}), nutrient input at the tidal head of the estuary (in kg-at d^{-1}) and at the drainage pipe (in mg-at d^{-1}). The mean, range, and coefficient of variance (V) are provided. The results are representative of baseflow conditions only.

Variable	Head of estuary			Pipe		
	μ	range	V	μ	range	V
Flow rate	0.89	0.4 – 1.6	61	2.84	0.7 – 5	66.7
PO_4	0.04	0 – 0.12	121.5	1.1	0.20 – 2.7	84.8
NO_3	7.04	1.3 – 20.5	100	48.73	4.8 – 133.4	87.8
NO_2	0.34	0.1 – 2	38.4	3.39	0.6 – 7	66.8
NH_4	0.62	0.2 – 2.3	47.4	4.11	2.1 – 8.6	56.7
DOC	72.1	24.8 – 166	81.2	1391.6	266 – 2916	67
DON	3.31	0.6 – 5.1	62	37.15	0.2 – 58.7	65.3
POC	83.02	0 – 232.1	111.5	650.01	31.7 – 1549.7	84.8
PON	1.34	0.3 – 4.4	47	2.93	0.7 – 11.7	123.5
TPP	0.37	0 – 1.3	101	1.73	0.1 – 6.9	127.7

The freshwater inflow under base flow conditions (0.35 to $1.6 \text{ m}^3 \text{s}^{-1}$) did not influence the mean estuarine nutrient concentration in any apparent pattern (Figures 3.9 A to G). There are too few data points for nutrient input for any meaningful statistical analyses. During the freset ($8 \text{ m}^3 \text{s}^{-1}$), the mean estuarine nutrient concentrations remained mostly within the range measured during base flow conditions.

3.3.1.2.5 Conservative/non-conservative behaviour of nutrients

Plots of nutrients against salinity did not reveal strong conservative behaviour for most nutrients, which points towards biological transformation and/or non-point source additions along the gradient (Figure 3.10). In general, nutrients which showed some conservative mixing behaviour and decreasing concentrations at higher salinities, also exhibited a wider range of values in lower salinity reaches, with the range narrowing down in the higher salinity reaches. Nutrients showing this pattern include nitrate, nitrite, DOC and DON. Phosphate, ammonium and particulate phosphorus concentrations did not show any definite pattern according to the salinity regime and thus behaved non-conservatively. POC shows generally higher values at higher salinities.

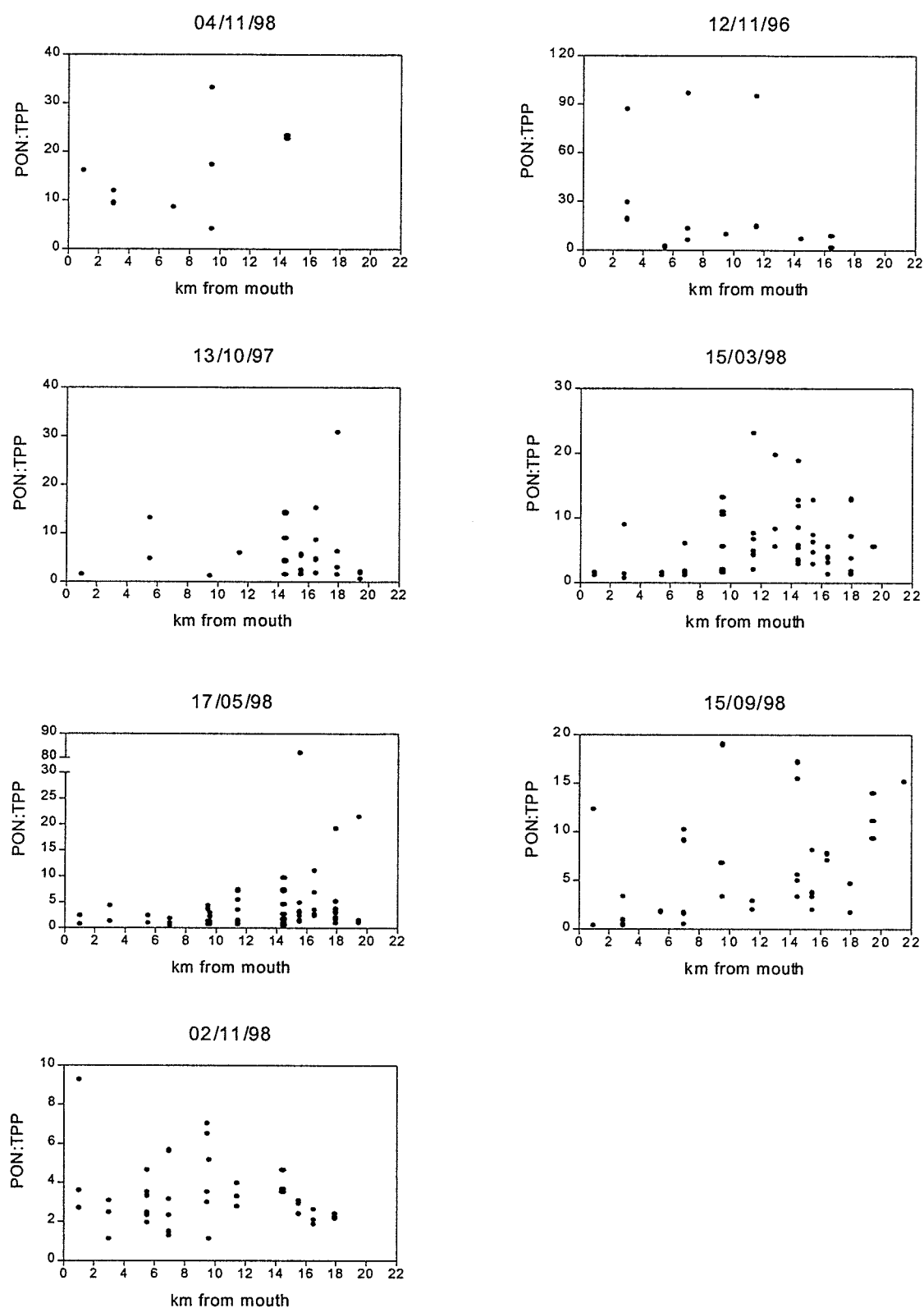


Figure 3.7: Particulate N:P ratios during a freshet (13/10/97), and under baseflow conditions.

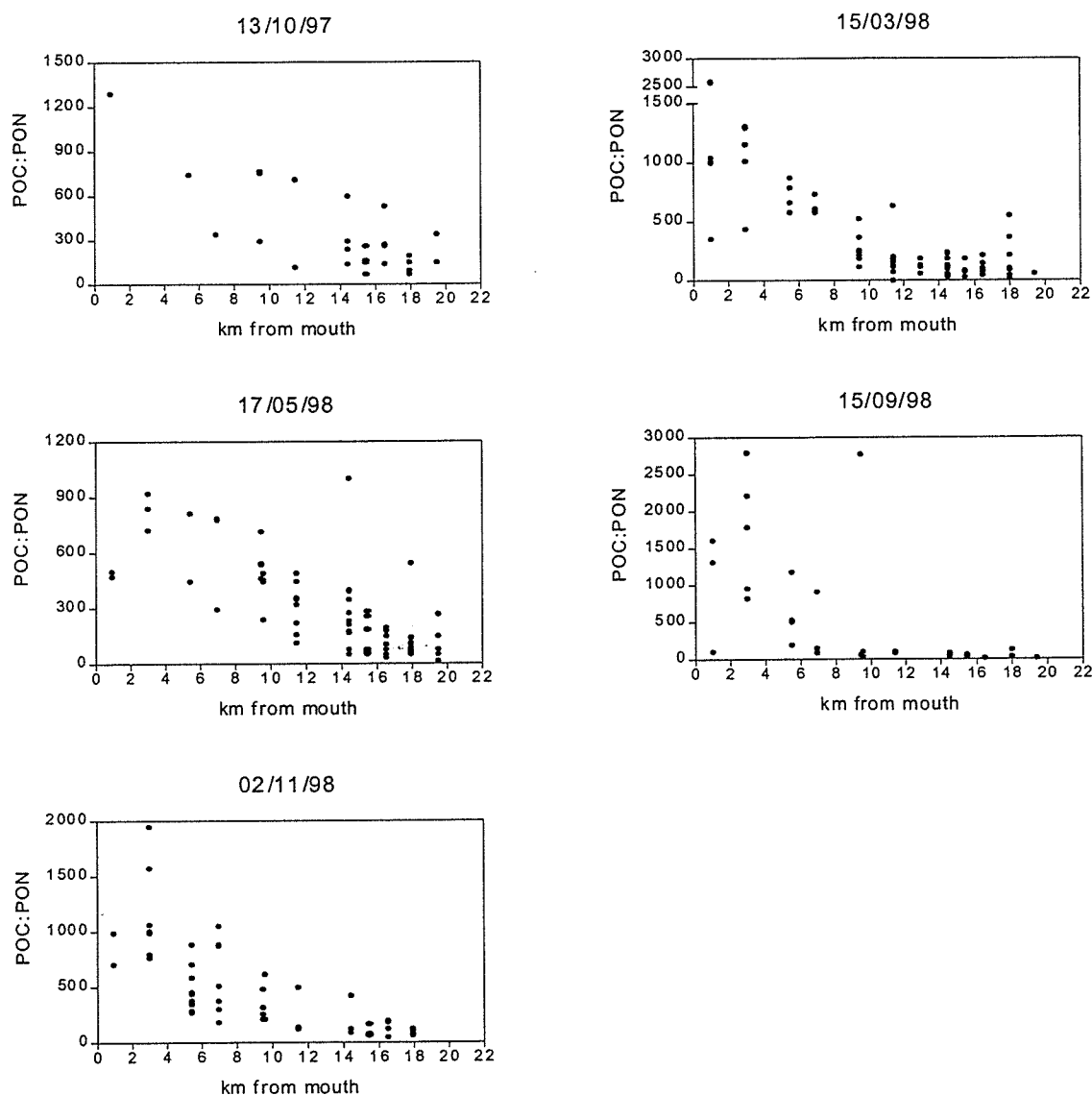


Figure 3.8: Particulate C:N ratios under baseflow conditions.

Table 3.4 investigates the existence of linear relationships of nutrients with salinity. A certain amount of variation in the nutrient concentrations can be explained by salinity for all nutrients except ammonium, which is highly susceptible to biological activity and is a relatively unstable compound. All other correlations were highly significant, despite low correlation coefficients in most cases. Highest coefficients were calculated for nutrients in high concentrations, including POC ($r = 0.40$), DOC ($r = -0.54$) and nitrate ($r = -0.44$).

Table 3.3: Correlation/regressions of freshwater flow versus nutrient concentration at the head of the estuary (A) and at the drainage pipe (B). Nutrient data were averaged over depth for baseflow conditions only. NS = not significant; * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$.

(A) Head of estuary:

Nutrient	μ	SD	r	r^2	t	p	n	p-level
PO ₄	0.36	0.29	0.67	0.45	2.55	0.03	10	*
NO ₃	96.6	63.8	-0.14	0.02	-0.4	0.697	10	NS
NO ₂	2.2	0.7	0.37	0.14	1.14	0.287	10	NS
NH ₄	6.1	2.5	-0.26	0.07	-0.75	0.477	10	NS
DOC	908.2	292.6	0.24	0.06	0.64	0.554	9	NS
DON	30.9	9.7	0.8	0.63	2.63	0.058	6	NS
POC	955.2	889.6	0.39	0.15	1.04	0.339	8	NS
PON	15.6	8.7	-0.27	0.07	-0.73	0.488	9	NS
TPP	4.1	3.5	0.09	0.01	0.26	0.8	10	NS

(B) Drainage pipe:

Nutrient	μ	SD	r	r^2	t	p	n	p-level
PO ₄	4.2	2.7	-0.54	0.3	-1.71	0.13	9	NS
NO ₃	125.2	118.3	-0.19	0.04	-0.5	0.63	9	NS
NO ₂	12.5	5	0.11	0.01	0.3	0.775	9	NS
NH ₄	27.9	35.7	-0.65	0.43	-2.28	0.057	9	NS
DOC	4819	2224.2	-0.04	0	-0.1	0.92	8	NS
DON	107.6	55	0.78	0.6	2.74	0.04	7	*
POC	2193	1184.5	0.4	0.16	1.16	0.283	9	NS
PON	9.9	7.7	0.04	0	0.1	0.927	8	NS
TPP	5.9	5.1	0.24	0.06	0.64	0.554	9	NS

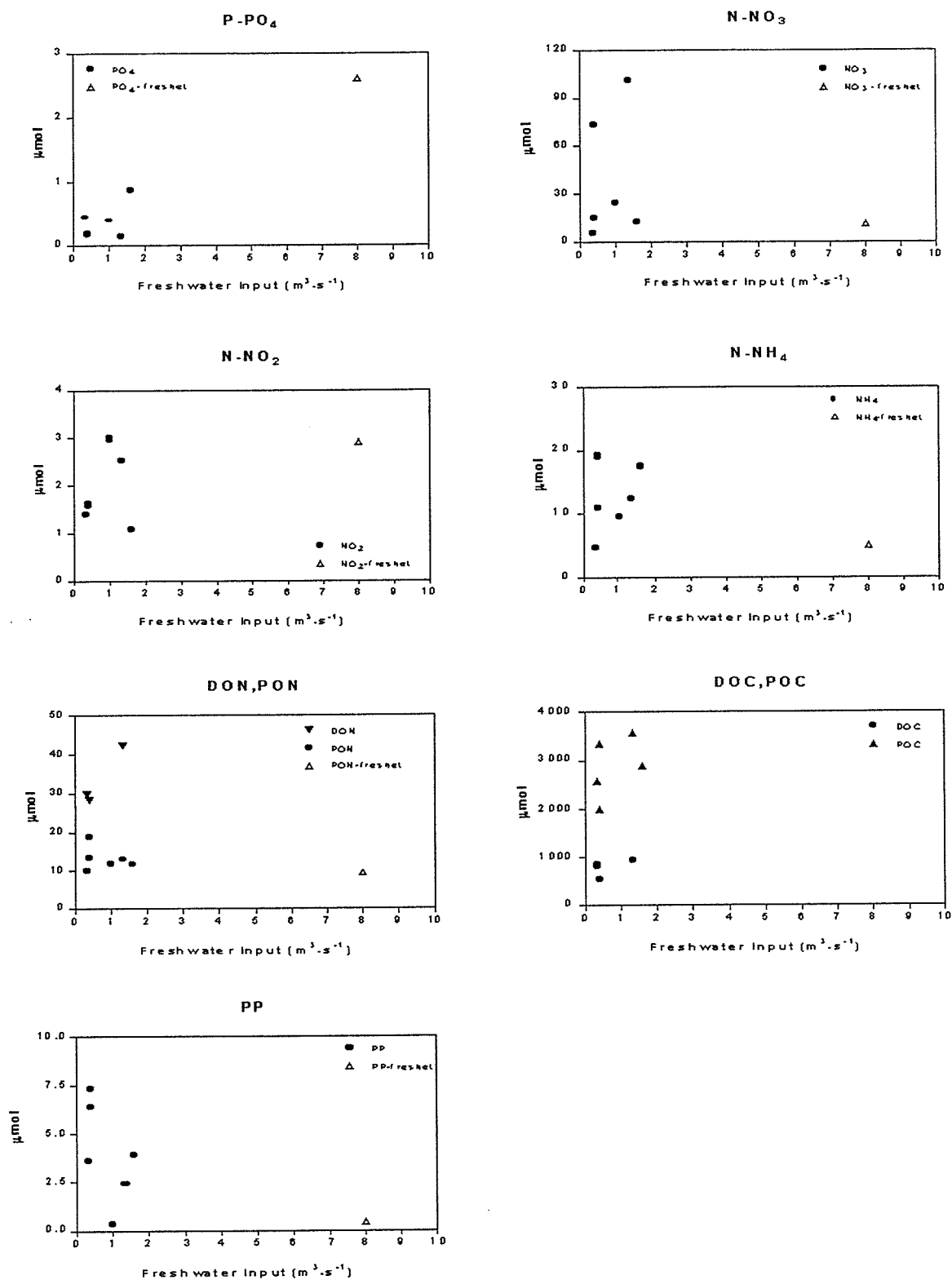


Figure 3.9: Mean estuarine nutrient concentrations (in μmol) plotted against freshwater inflow (in $\text{m}^3 \text{s}^{-1}$) at the tidal head of the Gamtoos Estuary.

Table 3.4: Correlation/regression of salinity against nutrients. All data were collected during baseflow conditions (0.4 to $1.6\text{m}^3\text{ s}^{-1}$). NS = not significant; * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$.

Nutrient	μ	SD	r	r^2	t	p	n	p-level
PO ₄	0.4	0.3	-0.2	0.04	-3.64	<0.001	329	***
NO ₃	39.6	47.8	-0.44	0.19	-8.74	<0.001	322	***
NO ₂	1.8	1.2	-0.57	0.32	-12.31	<0.001	324	***
NH ₄	12.7	46.4	0.13	0.02	2.27	0.024	325	NS
DOC	695.8	265.7	-0.54	0.29	-9.04	<0.001	202	***
DON	39	23.4	-0.3	0.09	-4.02	<0.001	161	***
POC	2719	1687.5	0.4	0.16	7.06	<0.001	270	***
PON	15.98	13.33	-0.35	0.12	-6.38	<0.001	293	***
TPP	4.45	4.7	-0.16	0.03	-2.85	0.005	310	**

3.3.1.2.6 A river-estuarine interface for nutrients?

One major objective of this study was to establish the possible existence of an REI region for nutrients. It was hypothesised that nutrient concentrations in the REI would be distinctly different to those in the rest of the estuary. As it is generally assumed that the river is a major source of nutrients to an estuary, the nutrient concentrations in the REI region would be expected to be higher, and that primary and secondary production in this area would also be higher relative to downstream regions.

Although some nutrients were found to occur at higher concentrations in the upper reaches, it was not possible to identify an REI region confined within a certain salinity range for any of the nutrients (Figure 3.11 A to I). Statistical analyses failed to identify a defined REI region for phosphate, ammonium, DON, POC and TPP. For all other nutrients, those reaches with nutrient concentrations different from other estuarine reaches were found in different salinity regimes for almost every set of samples. Despite these variations, it was generally at a vertically averaged salinity of 17 ‰ and below that higher concentrations of nutrients were measured. In the case where there was an REI region identified, concentrations of the nutrients were higher in those reaches. An exception was POC, for which concentrations were generally lower in the lower salinity reaches, increasing towards the mouth.

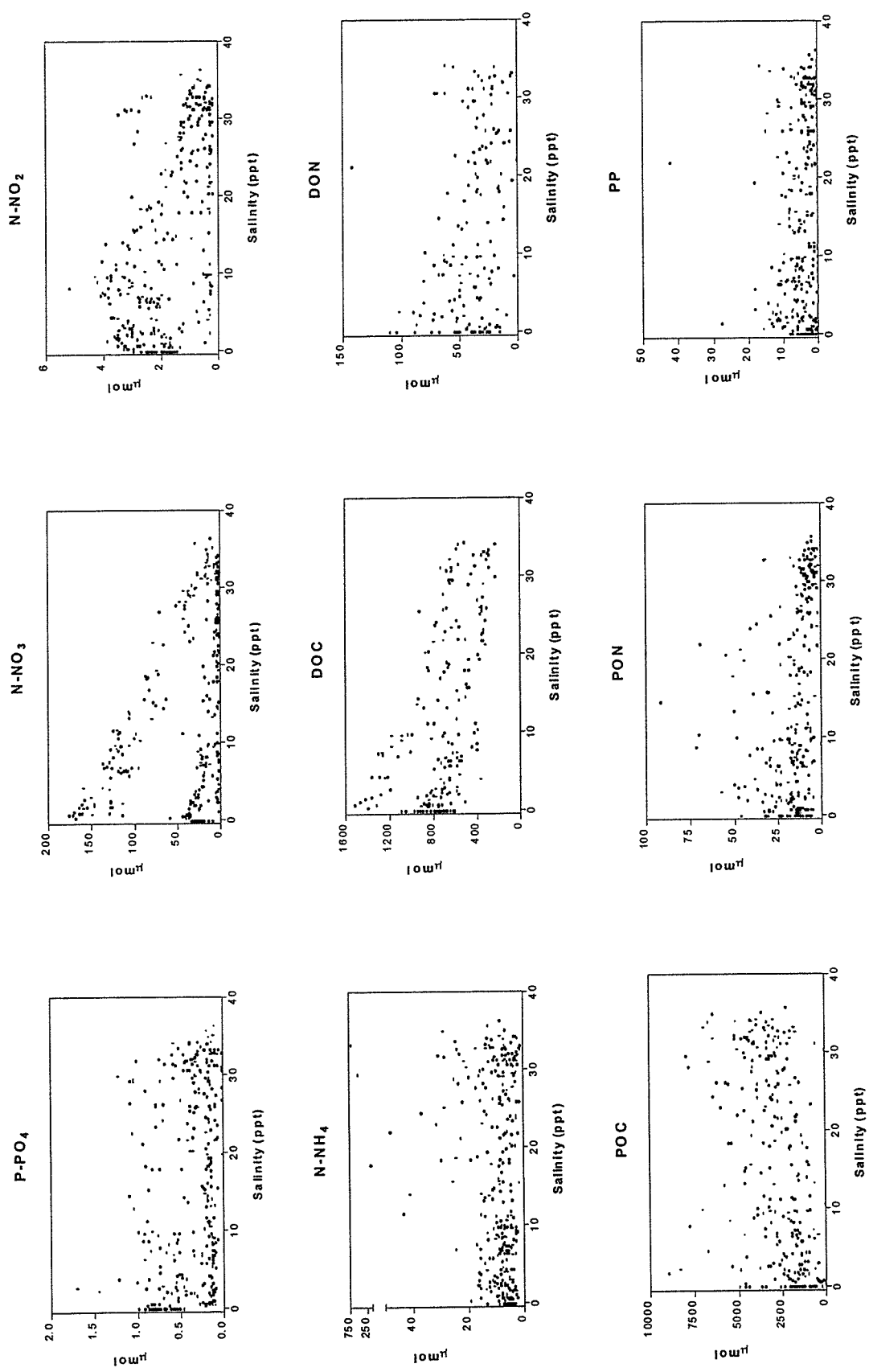


Figure 3.10: Nutrient concentration (in μmol) plots against salinity for all measurements taken during baseflow conditions in the Gamtoos Estuary.

3.3.1.2.7 Short-term variability in nutrient concentrations

Results of analyses showed that within a time-span of 6 hours (i.e. from one slack tide to the next), the nutrient status at one particular station did not generally change significantly (Table 3.5). From this it was concluded that freshwater inflow was not ultimately responsible for the nutrient status of this estuary, and that non-point sources in addition to biogeochemical recycling of elements may be influencing nutrient dynamics in this estuary. Should freshwater input at the tidal head of the estuary be a major determinant of the nutrient status in the Gamtoos Estuary, one would expect, (i) stronger linear correlations with salinity, and (ii) alterations of the nutrient concentrations in concordance with the salinity regime, particularly in the upper reaches (i.e. near the point source).

3.3.1.3 Comparison with other Eastern Cape estuaries

Mean freshwater inflow, salinity and dissolved nutrient concentrations were compared to other permanently open Eastern Cape estuaries. The Kromme, Swartkops, and Sundays estuaries were investigated in terms of their nutrient status in relation to freshwater input during the years 1993 to 1995 (Scharler *et al.* 1997). The Sundays Estuary exhibited the highest mean freshwater input, followed by the Gamtoos and Swartkops estuaries. The Kromme Estuary had the lowest freshwater input. Mean estuarine salinities were accordingly highest in the Kromme Estuary, but gradually decreased from the Swartkops to the Sundays to the Gamtoos estuaries (Figure 3.12).

3.3.2 Porewater and bottom water

Estuarine physico-chemical gradients in bottom waters and the sediment all changed significantly over the six-month sampling period. Generally, these changes were significantly correlated with salinity. The most notable result was the significant correlations observed between respective bottom water and porewater nutrients.

3.3.2.1 Bottom waters

Cumulative weekly rainfall and mean weekly water flow were measured over the six-month sampling period (Figure 3.13). Highest cumulative weekly rainfalls were recorded in March and February (124.4 mm and 56.2 mm, respectively), while the lowest weekly total was measured in October and November (0 mm). Peak water flows of 13.5 and 16.7 m³ s⁻¹ were recorded in December and February, respectively. Highest variations associated with mean weekly water flow, were also observed in December and February, indicating pulses of rainfall in these months.

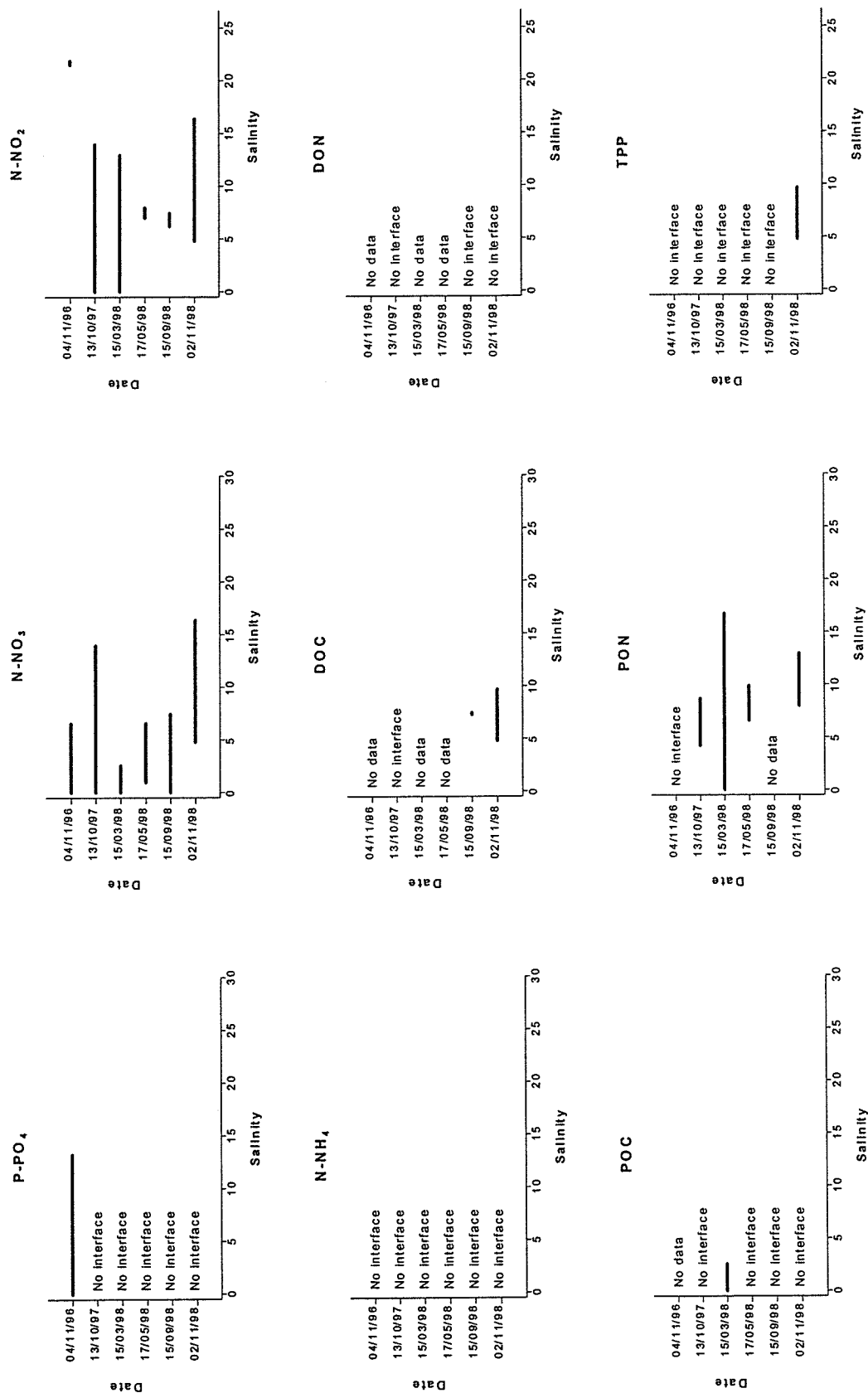


Figure 3.11: A river-estuarine interface region (<15 ‰) for nutrients calculated by ANOVA and post-hoc Tukey tests for several sampling sessions.

Table 3.5: Paired Wilcoxon on salinity and nutrient data in lower salinity reaches (ca. < 15 ‰). Results are given for individual sampling sessions and the state of the tide for which samples are compared. NS = not significant; * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$; n.d. = no data.

	13/10/97	15/03/98	15/03/98	15/03/98	17/05/98	17/05/98	17/05/98	15/09/98
Variable	High - low	High - low	Low - high	High - high	Low - high	High - low	Low - low	Low - high
Salinity	**	*	NS	NS	NS	NS	NS	NS
PO ₄	NS	NS	NS	NS	NS	*	*	NS
NO ₃	**	**	*	NS	NS	NS	NS	***
NO ₂	NS	*	*	*	NS	NS	NS	NS
NH ₄	NS	NS	NS	NS	NS	NS	NS	NS
DOC	*	NS	*	*	NS	NS	NS	n.d.
DON	NS	NS	NS	*	n.d.	n.d.	n.d.	n.d.
POC	NS	NS	NS	NS	NS	NS	NS	NS
PON	**	*	NS	*	*	*	NS	NS
TPP	NS	NS	NS	NS	*	NS	NS	NS

Gradients in salinity and temperature were significantly different between estuarine sites and months, and the interaction of spatial and temporal differences was also significant (Figures 3.14, Table 3.5). Salinity profiles were generally between 15 and 35 ‰. Water temperature increased from a minimum estuarine average of approximately 17°C in September to a maximum average of approximately 26°C in February. Post-hoc analyses found both salinity and temperature profiles to be significantly different in the upper regions, to those of lower regions (Table 3.7). Generally, salinity averages in the lower reaches were higher by approximately 10 to 30 ‰ than salinities in the upper reaches, while temperatures were approximately 3 to 5°C higher in the upper reaches than in the lower reaches. Salinities were significantly negatively correlated with a number of physico-chemical variables, including temperature, water column nitrate, nitrite, and ammonium concentrations, and porewater nitrate, nitrite and phosphate concentrations (Table 3.9).

Dissolved oxygen and pH changed significantly over time, and the interaction of distance upstream of the estuary mouth and time was significant for both variables (Figure 3.15, Table 3.6). In general, dissolved oxygen concentrations were between 6 and 12 mg l⁻¹, with maximum and minimum concentrations of 16 and 3.7 mg l⁻¹ recorded in September and March, respectively. Similarly, pH values were comparable between sampling points and months, ranging between 8 and 8.3 units, although values of 8.57 and 7.6 units were recorded in September and February, respectively.

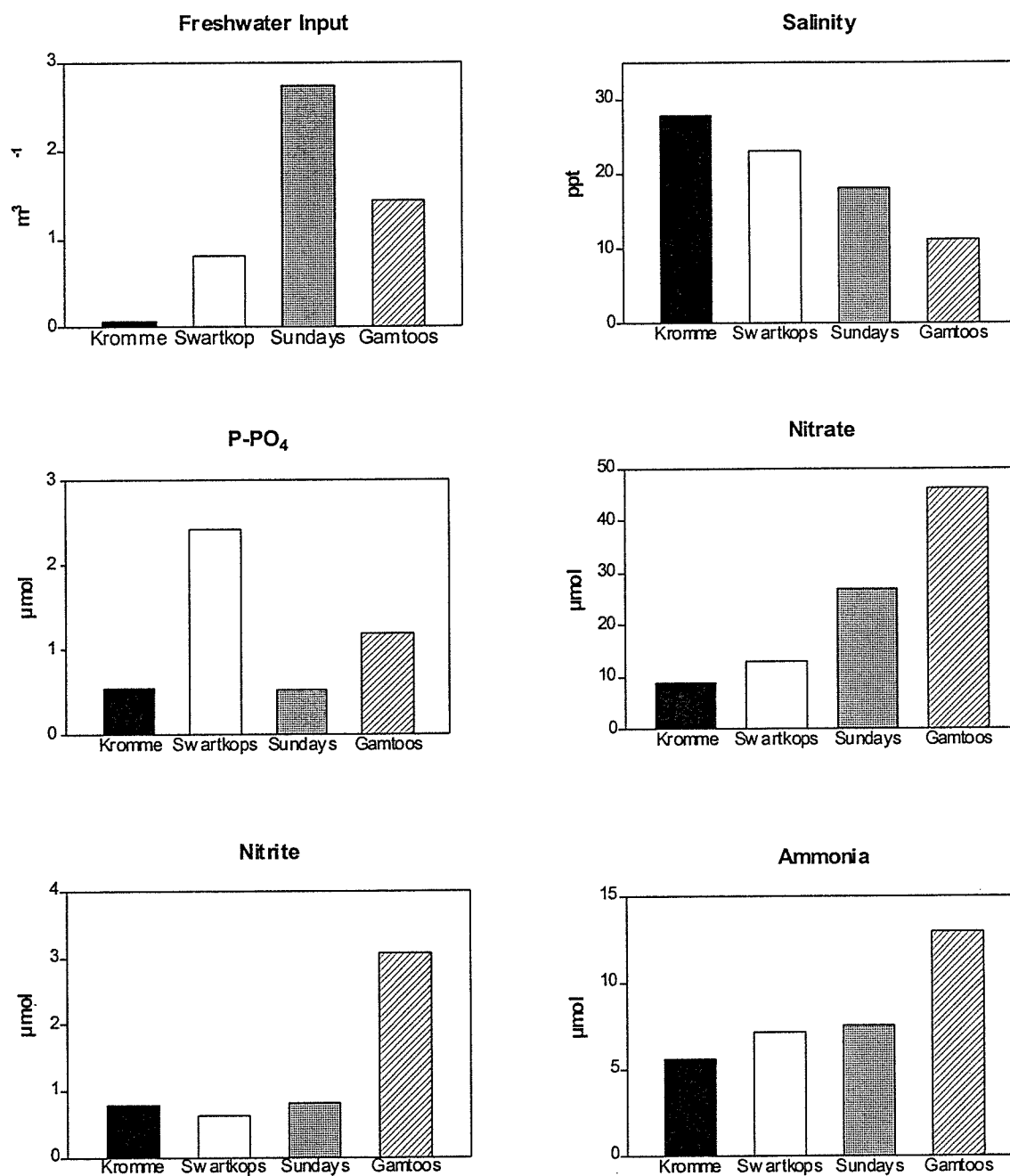


Figure 3.12: Mean values for freshwater inflow ($\text{m}^3 \text{s}^{-1}$), salinity, phosphate (μmol), nitrate (μmol), nitrite (μmol), and ammonium (μmol), in the Kromme, Swartkops, Sundays and Gamtoos estuaries.

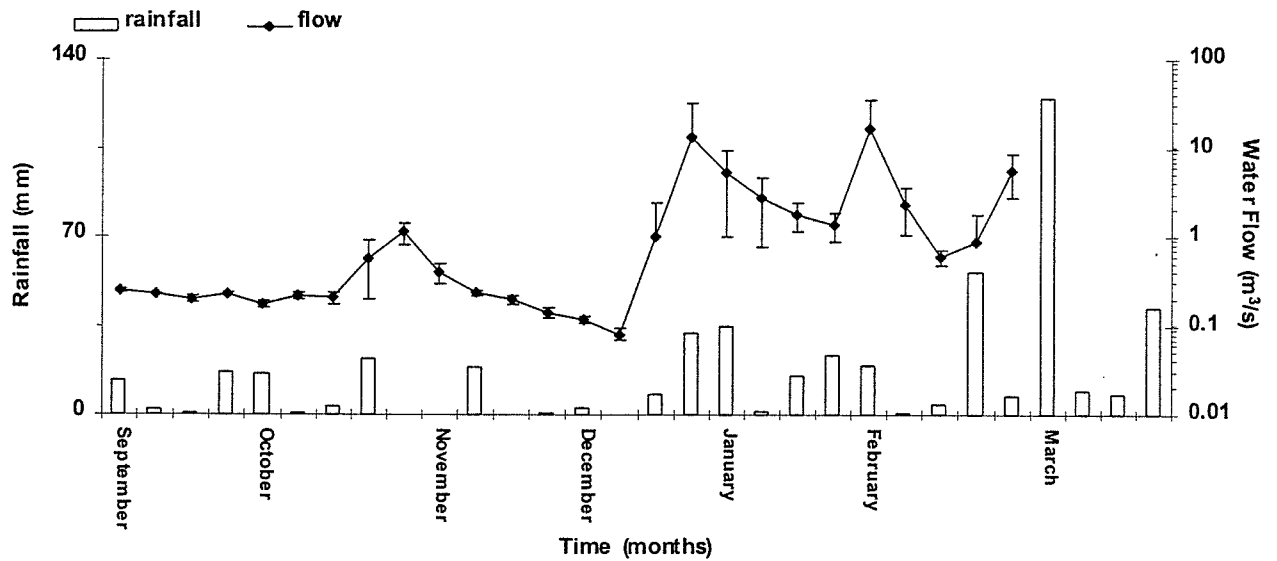


Figure 3.13: Cumulative monthly rainfall (□) and freshwater flow (—◆—) from September 1999 to March 2000, in the Gamtoos Estuary and catchment.

Light attenuation coefficients differed significantly between sites, increasing with distance upstream (Figure 3.16, Table 3.6). Coefficients were usually between 0.2 and 2, however, light attenuation coefficients as high as 2.5 and 5 were recorded in the upper estuarine regions in February. Chlorophyll *a* concentrations were measured as an indication of phytoplankton biomass. Concentrations were found to vary significantly between sites (Figure 3.16, Table 3.6). Chlorophyll *a* concentrations were approximately 1 to 10 $\mu\text{g l}^{-1}$ at sampling points 1 to 9 km upstream of the mouth, for all sampling times, with the exception of one measurement in March of approximately 20 $\mu\text{g l}^{-1}$ in surface waters, 1 km from the mouth. Concentrations were generally highest at the sampling point 13 km upstream of the mouth, where they were between 5 and 30 $\mu\text{g l}^{-1}$ for all sampling occasions. Concentrations as high as 55 and 65 $\mu\text{g l}^{-1}$ were measured at this site in September and March. Water column chlorophyll *a* concentrations were found to have a significant, positive correlation with temperature, and a negative correlation with salinity and sediment redox potential (Table 3.9).

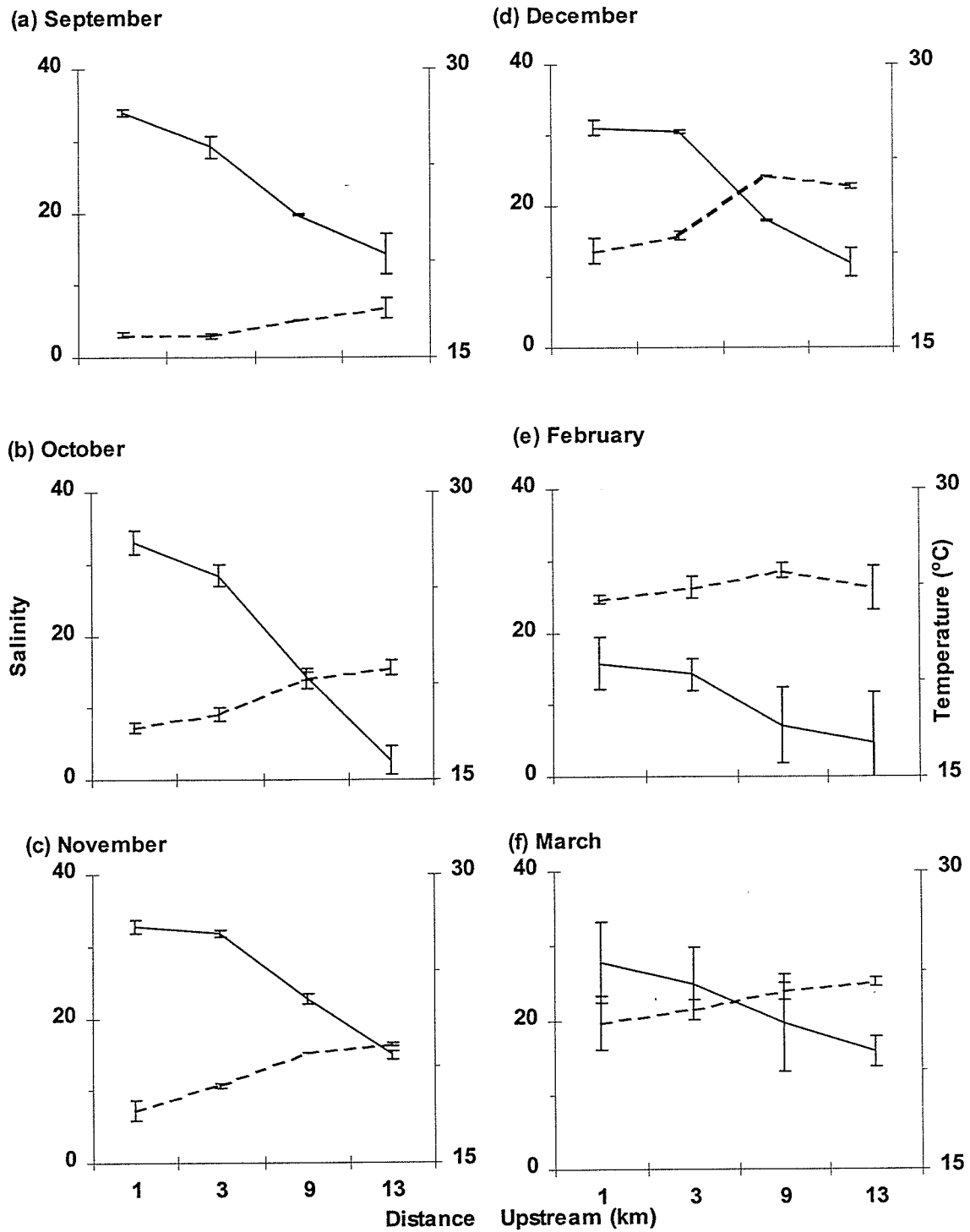


Figure 3.14 A to F: Mean salinity (—) (‰) and temperature (----) (°C) of the water column with increasing distance upstream (km) from the estuary mouth, from September 1999 to March 2000 ($n = 3$) $\pm$ SE.

Table 3.6: Results of repeated measures ANOVAs of parameters measured between September 1999 and March 2000 (* = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$, NS = not significant) (n=3).

Variable	Factors	d.f.	F	MS	p-level
Salinity	distance upstream	3	44.2889	1277.02	***
	month	5	55.1432	349.8	***
	interaction	15	5.1762	32.84	***
Temperature	distance upstream	3	35.7701	23.95	***
	month	5	682.7037	110.57	***
	interaction	15	7.8995	1.28	***
pH	distance upstream	3	1.2269	0.02	NS
	month	5	61.8689	0.15	***
	interaction	15	15.5978	0.04	***
Dissolved Oxygen	distance upstream	3	1.3187	4.32	NS
	month	5	20.2917	29.03	***
	interaction	15	3.2071	4.59	**
Redox Potential	distance upstream	3	61.5164	56883	***
	month	5	9.0279	26940.5	***
	interaction	15	7.5238	22452	***
Sediment Organic Content	distance upstream	3	23.6922	2.82	***
	month	5	8.4695	2.1	***
	interaction	15	2.6179	0.65	**
Light	distance upstream	3	3.508	2.49	*
	month	5	2.5595	1.8	NS
Chlorophyll <i>a</i>	distance upstream	3	3.1644	501.62	*
	month	5	1.5954	287.12	NS
WATER					
Nitrate	distance upstream	3	147.978	33.7962	***
	month (2)	5	1706.315	419.443	***
	1 x 2	15	179.326	44.0814	***
Nitrite	distance upstream	3	4.1537	22.4614	***
	month (2)	5	0.5287	2.3733	NS
	1 x 2	15	0.7339	3.2941	***
Ammonium	distance upstream	3	0.0015	0.7706	NS
	month (2)	5	0.0293	12.8234	***
	1 x 2	15	0.0025	1.1032	NS
Phosphate	distance upstream	3	0.0752	814.374	***
	month (2)	5	0.2272	1607.65	***
	1 x 2	15	0.0952	673.447	***

Table 3.6 continued

Variable	Factors	d.f.	F	MS	p-level
POREWATER					
Nitrate	distance upstream	3	9.3281	2565	NS
	Month	5	14.1794	30171.7	***
	interaction	15	1.6508	3512.57	NS
Nitrite	distance upstream	3	0.4602	24.84	NS
	Month	5	17.7626	295.06	***
	interaction	15	1.4896	24.74	NS
Ammonium	distance upstream	3	0.6377	0.31	NS
	Month	5	5.5958	5.53	***
	interaction	15	0.7096	0.7	NS
Phosphate	distance upstream	3	12.8024	1.24	**
	Month	5	63.9584	6.84	***
	interaction	15	6.6454	0.71	***

Water column nitrate and phosphate concentrations differed significantly with distance upstream of the estuary mouth and time, and interaction values of all combinations of these factors were also significant (Figures 3.17, Table 3.6). Post-hoc analyses found nitrate concentrations at the first and second sampling sites, 1 and 3km from the estuary mouth, respectively, were significantly lower than at other sites; nitrate concentrations fluctuated significantly between months, yet without obvious pattern (Tables 3.6 to 3.8). Mean water column nitrate concentrations were between 1 and 20 μM at all sites and depths, except in February and March when mean concentrations were between 16 and 52 μM . Post-hoc analyses found sites at 3 and 9 km upstream of the estuary mouth generally had significantly higher phosphate concentrations (Table 3.7). Analyses also found that phosphate concentrations fluctuated significantly between all months, yet without noticeable trends (Table 3.8). Mean water column phosphate concentrations were between 0.03 and 0.3 μM at all sites and depths, except in September and October 3 km from the estuary mouth, where concentrations were 0.69 and 1.02 μM , respectively.

Table 3.7: Results of post-hoc Tukey analyses for all parameters with significant differences between sites 1 to 4 along estuarine length, monitored during the sampling period of September 1999 to March 2000 (* = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$, NS = not significant) (n=3).

Variable	p-level			
Salinity	1	2	3	4
1				
2	NS			
3	***	**		
4	***	***	*	
Temperature	1	2	3	4
1				
2	NS			
3	***	**		
4	***	**	NS	
Redox	1	2	3	4
1				
2	***			
3	NS	***		
4	***	NS	***	
Sediment Organic Content	1	2	3	4
1				
2	*			
3	**	***		
4	NS	NS	**	
WATER COLUMN				
Nitrate	1	2	3	4
1				
2	NS			
3	***	**		
4	***	***	NS	
Nitrite	1	2	3	4
1				
2	NS			
3	NS	NS		
4	***	**	**	

Table 3.7 continued

Variable	p-level			
Phosphate	1	2	3	4
1				
2	***			
3	NS	***		
4	**	***	*	
POREWATER				
Phosphate	1	2	3	4
1				
2	NS			
3	NS	NS		
4	**	**	**	

Table 3.8: Results of post-hoc Tukey analyses for all parameters with significant differences between months along estuarine length, monitored during the sampling period of September 1999 to March 2000 (* = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$, NS = not significant) (n=3).

Variable	p-level					
Salinity	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	**					
Nov	NS	***				
Dec	NS	*	NS			
Feb	***	***	***	***		
Mar	NS	NS	*	NS	***	
Temperature	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	***					
Nov	***	NS				
Dec	***	***	***			
Feb	***	***	***	***		
Mar	***	***	***	***	***	
pH	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	***					

Table 3.8 continued

Variable	p-level					
Nov	***	NS				
Dec	**	NS	NS			
Feb	***	***	***	***		
Mar	NS	***	***	*	***	
Dissolved Oxygen	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	***					
Nov	***	**				
Dec	***	NS	NS			
Feb	***	**	NS	NS		
Mar	***	NS	**	NS	**	
Redox	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	NS					
Nov	NS	NS				
Dec	NS	NS	NS			
Feb	*	*	NS	NS		
Mar	***	***	*	*	NS	
Sediment Organic Content	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	NS					
Nov	**	**				
Dec	NS	NS	***			
Feb	NS	NS	NS	**		
Mar	NS	NS	NS	NS	NS	
WATER COLUMN						
Nitrate	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	NS					
Nov	NS	NS				
Dec	***	***	***			
Feb	***	***	***	***		
Mar	NS	NS	NS	***	***	

Table 3.8 continued

Variable	p-level					
Ammonium	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	NS					
Nov	**	NS				
Dec	***	*	NS			
Feb	NS	*	***	***		
Mar	**	NS	NS	NS	***	
Phosphate	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	***					
Nov	***	***				
Dec	NS	***	***			
Feb	***	***	NS	***		
Mar	***	***	***	***	***	
POREWATER						
Nitrate	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	NS					
Nov	NS	NS				
Dec	NS	NS	NS			
Feb	***	***	***	***		
Mar	NS	NS	NS	NS	***	
Nitrite	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	NS					
Nov	NS	*				
Dec	NS	NS	NS			
Feb	***	***	***	***		
Mar	NS	NS	NS	NS	***	
Ammonium	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	NS					
Nov	***	NS				
Dec	NS	NS	NS			
Feb	NS	NS	NS	NS		
Mar	*	NS	NS	NS	NS	

Table 3.8 continued

Variable	p-level					
Phosphate	Sep	Oct	Nov	Dec	Feb	Mar
Sep						
Oct	NS					
Nov	NS	NS				
Dec	NS	NS	NS			
Feb	***	***	***	***		
Mar	NS	*	NS	NS	***	

Bottom water nitrite concentrations varied significantly with distance from the estuary mouth, with the interaction of the factors distance and time significant (Figure 3.18, Table 3.6). Post-hoc analyses found that relatively higher nitrite concentrations at the sampling site 13 km from the estuary mouth were significantly different from other sites (Table 3.7). Mean water column nitrite concentrations were normally between 1 and 2.5 μM however, mean concentrations between 2.7 and 3.5 μM in the months of September, October, December and February, were observed at sites 9 and 13 km upstream of the estuary mouth.

Ammonium concentrations varied significantly in the water column between months (Figure 3.18, Table 3.6). Post-hoc analyses found ammonium concentrations measured in September, October, November and December varied significantly with each other and with trends recorded in the other three months (Table 3.8). Mean water column ammonium concentrations were generally between 0.01 and 0.1 μM . However, relatively low mean concentrations were recorded in November (≤ 0.01 μM), and relatively high concentrations were observed in September and February (between 0.1 and 0.4 μM). All bottom water nutrient concentrations were found to correlate significantly with respective porewater nutrient concentrations, with the exception of phosphate, where correlations were positive between bottom water and porewater concentrations for nitrate and nitrite, and negative for ammonium (Table 3.9).

3.3.2.2 Sediment

Sediment redox potential varied significantly with both distance upstream of the estuary mouth and time, and the interaction of these two factors was also significant (Figures 3.19 A to F, Table 3.6). Significant differences in redox potentials between sites was due to more positive values at sampling sites 1 and 9 km upstream from the mouth, than values recorded at sites 3 and 13 km upstream from the mouth (Table 3.7). Sites 1 and 9 km upstream of the mouth constantly recorded mean redox potentials between 125 and 195 mV (except in March), while sites 3 and 13 km upstream of the mouth recorded mean redox potentials that fluctuated between 212 and 221 mV.

Table 3.9: Results of significant correlations between variables measured (* = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$, NS = not significant) (n=3). Temp = temperature, DO = dissolved oxygen, Chl *a* = chlorophyll *a*, PW = porewater, SOC = sediment organic content, WC = water column.

Factors	Salinity	Rainfall	Temp	DO	PW NO ₂	PW NO ₃	Redox	pH	SOC	PW NH ₄	PW PO ₄	Chl <i>a</i>	WC NO ₂	WC NO ₃
Salinity														
Rainfall	NS													
Temp	-0.59 (***)	NS												
DO	NS	NS	-0.4 (***)											
PW NO ₂	-0.49 (***)	NS	0.39 (**)	NS										
PW NO ₃	-0.4 (**)	NS	0.37 (**)	-0.29 (*)	0.39 (**)									
Redox	NS	NS	-0.31 (**)	NS	NS	NS								
pH	NS	NS	-0.64 (***)	0.59 (***)	NS	NS	NS							
SOC	NS	-0.45 (*)	NS	NS	NS	NS	-0.4 (***)	NS						
PW NH ₄	NS	NS	NS	-0.33 (**)	-0.24 (*)	NS	NS	-0.46 (***)	NS					
PW PO ₄	-0.47 (**)	NS	0.51 (***)	-0.32 (**)	0.54 (***)	0.53 (***)	-0.34 (**)	NS	NS	NS				
Chl <i>a</i>	-0.27 (*)	NS	0.28 (*)	NS	NS	NS	-0.29 (*)	NS	NS	NS	NS			
WC NO ₂	-0.51 (***)	NS	NS	0.38 (**)	0.24 (*)	NS	NS	0.29 (*)	NS	-0.33 (***)	NS	NS		
WC NO ₃	-0.5 (**)	NS	0.51 (***)	-0.33 (**)	0.65 (***)	0.58 (***)	-0.24 (*)	NS	NS	NS	0.79 (***)	NS	NS	
WC NH ₄	-0.28 (*)	NS	NS	NS	0.36 (**)	0.36 (***)	NS	0.42 (***)	NS	-0.26 (*)	0.47 (***)	NS	0.29 (*)	0.42 (***)

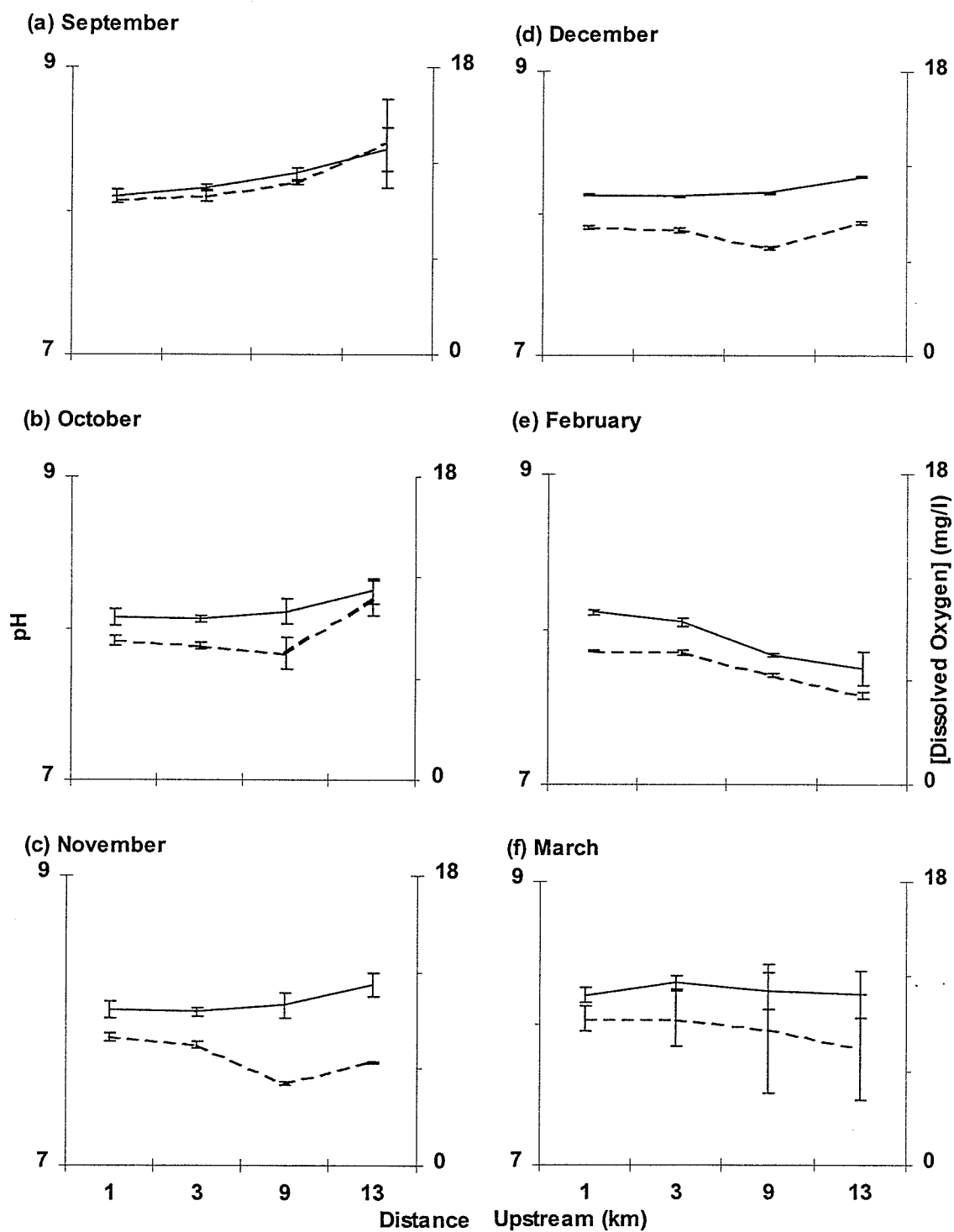


Figure 3.15 A - F: Mean pH (—) and dissolved oxygen (----) of bottom waters with increasing distance upstream (in km) from the Gamtoos Estuary mouth, for the months September 1999 to March 2000 ($n=3$) $\pm$ SE.

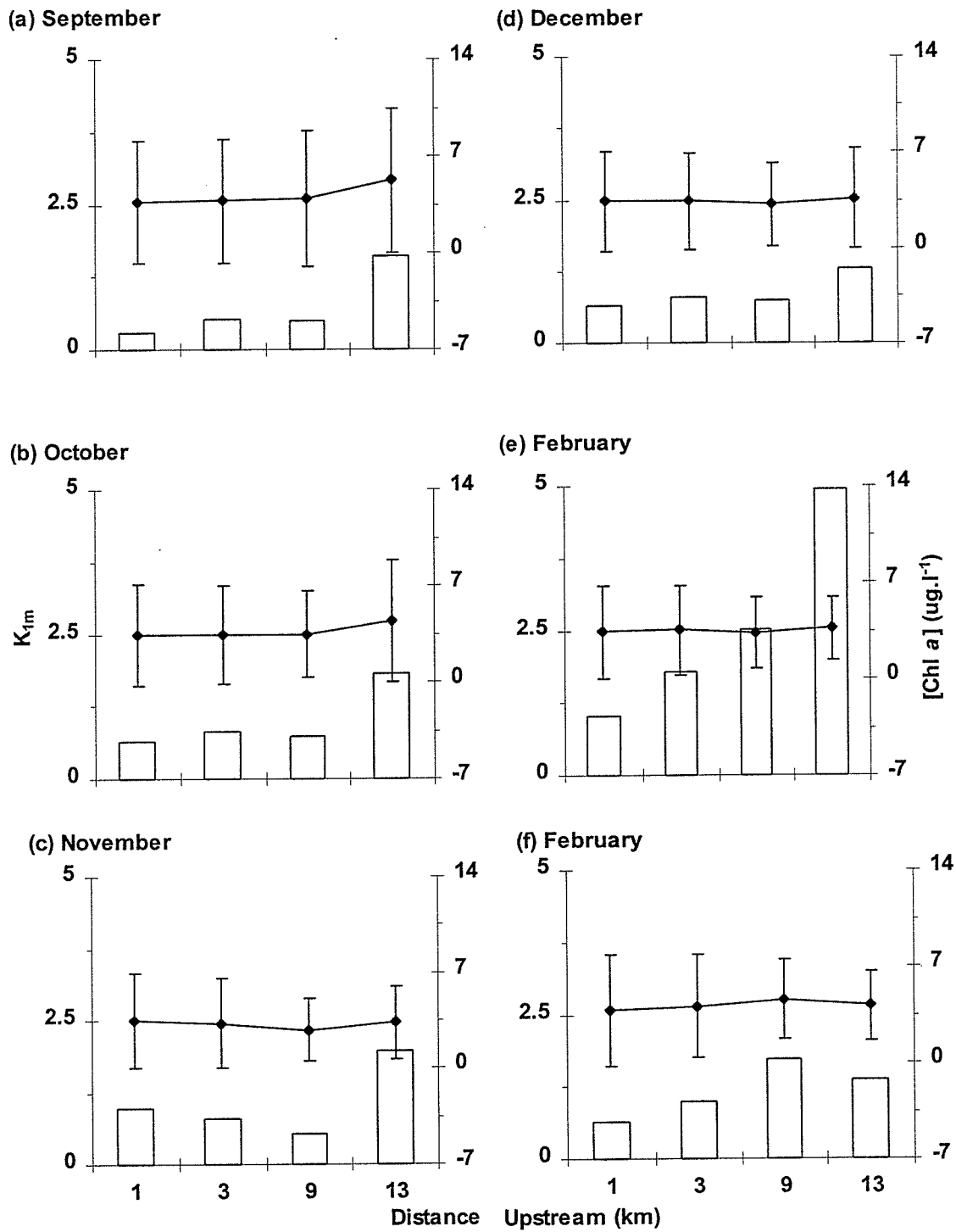


Figure 3.16: Light attenuation coefficients ($\square$) ($\mu E\ m^{-2}\ s^{-1}$) and mean chlorophyll *a* concentrations ($\blacklozenge$) ($\mu g\ l^{-1}$) at 1 m depth, with increasing distance upstream (km) from the Gamtoos Estuary mouth, for the months September 1999 to March 2000 ($n=3$) $\pm$ SE.

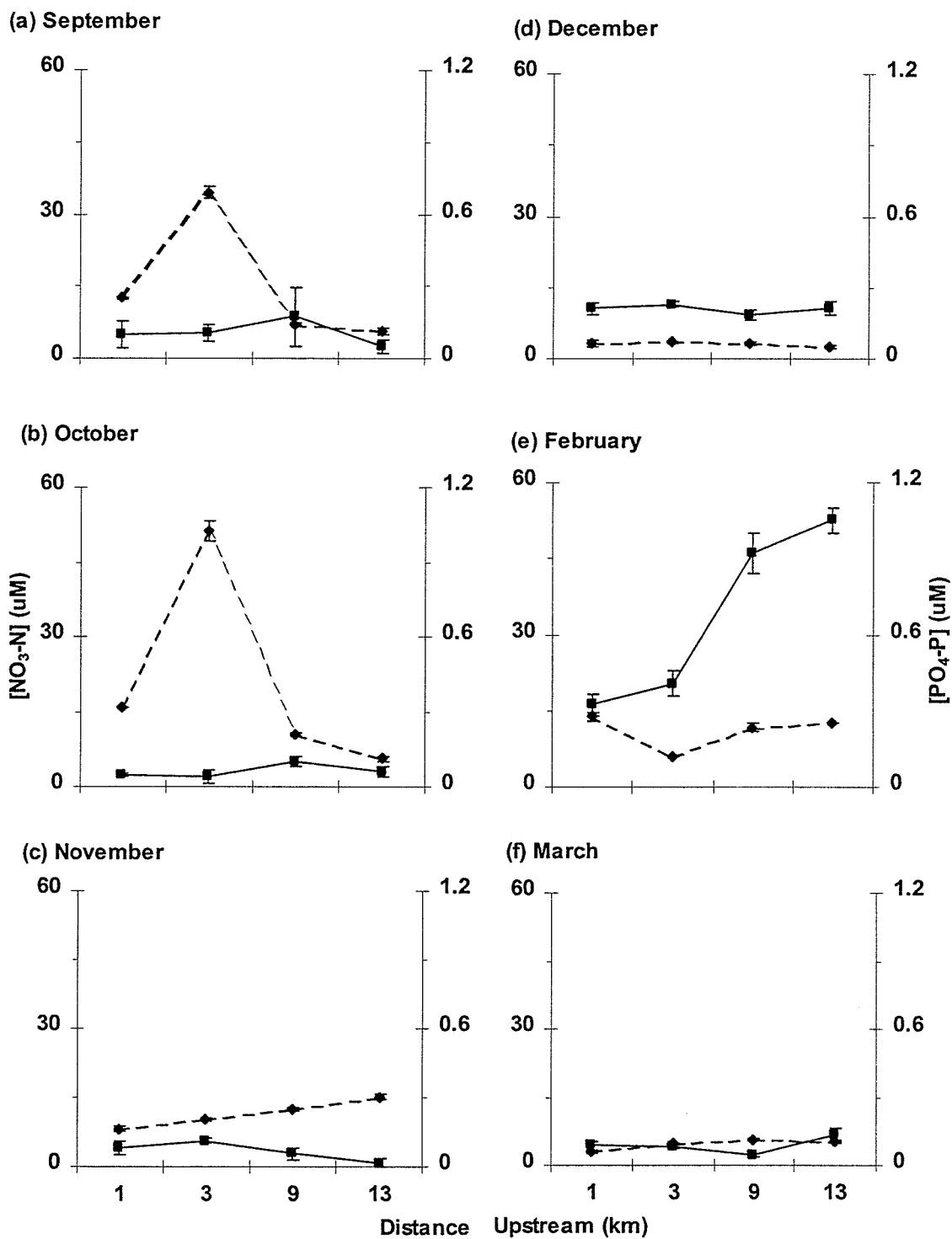


Figure 3.17 A - F: Mean bottom water nitrate (—■—) and phosphate concentrations (- ♦ -) (in μM), with increasing distance upstream (in km) from the Gamtoos Estuary mouth, for the months September 1999 to March 2000 ($n=3 \pm \text{SE}$).

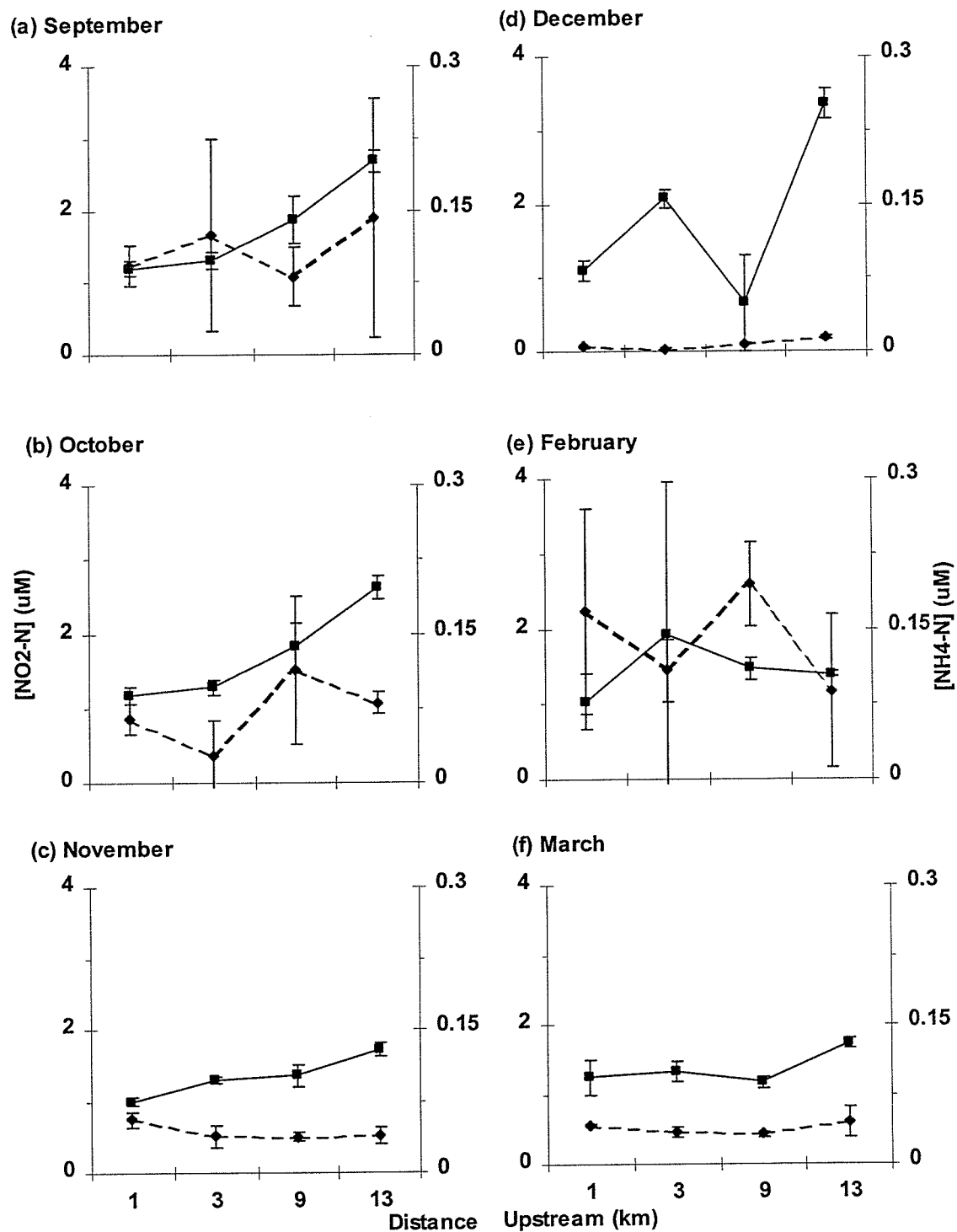


Figure 3.18 A - F: Mean bottom water nitrite (—■—) and ammonium concentrations (- ♦ -) (in μM), with increasing distance upstream (in km) from the Gamtoos Estuary mouth, for the months September 1999 to March 2000 ($n = 3$) $\pm$ SE.

Sediment organic content varied significantly with both distance upstream of the estuary mouth and time, and the interaction of these two factors was also significant (Figures 3.19 A to F, Table 3.6). Organic contents at all sites, for all seasons ranged between 0.4 and 2.7 %. Post-hoc analyses found organic contents to be significantly lower at the sampling site located 9 km upstream of the mouth, than at other sampling points, and also to be significantly lower in the month of November than in other months (Table 3.7). Sediment organic content was found to have a significant negative correlation with rainfall (Table 3.9).

Significant variation in porewater nitrate and nitrite concentrations was due to the factor of time (Figures 3.20 A to F, Table 3.6). Post-hoc analyses revealed that the significance of time was the result of higher nitrate and nitrite porewater concentrations observed in February (Table 3.7). Mean porewater nitrate concentrations were generally between 50 and 100 in μM , however, relatively high mean porewater nitrate concentrations of between 150 and 200 μM were recorded in November and February, while relatively low mean concentrations (10 to 30 μM) were recorded in September, October, November and March. Mean porewater nitrite concentrations were approximately one magnitude lower than nitrate concentrations, with concentrations between 5 and 15 μM for most sites and months. Relatively low mean concentrations between 1 and 3 μM were recorded in November and December, and relatively high mean concentrations between 18 and 20 μM were measured in February.

With respect to porewater nitrate, both mean ammonium and phosphate porewater concentrations were one to two orders of magnitude lower (Figures 3.20 A to F). Both ammonium and phosphate porewater concentrations varied significantly over time, and the interaction of time and distance upstream of the estuary mouth were significant, while phosphate concentrations also varied significantly over space (Table 3.6). Significant temporal variation in porewater ammonium concentrations was due to relatively low concentrations recorded in September ($\leq 0.1 \mu\text{M}$; Table 3.7). Generally, mean porewater ammonium concentrations were between 0.5 and 2 μM , although low means of 0.2 and 0.3 μM were recorded in October and December, respectively, and a high mean of 2.2 μM was measured in November. Porewater ammonium concentrations had a significant negative correlation with the pH of the overlying benthic water (Table 3.9). Significant variation in porewater phosphate concentrations with distance upstream of the estuary mouth was due to relatively high concentrations recorded at the sampling site 13 km upstream of the mouth, while variation over time was a result of relatively high concentrations measured in February and March (Table 3.7). Mean porewater phosphate concentrations were between 0.15 and 1.5 μM , although means above and below this were observed in November and December (0.007 to 0.009 μM), and February (1.6 and 3.8 μM). Porewater phosphate concentrations had a significant negative correlation with sediment redox potential (Table 3.9).

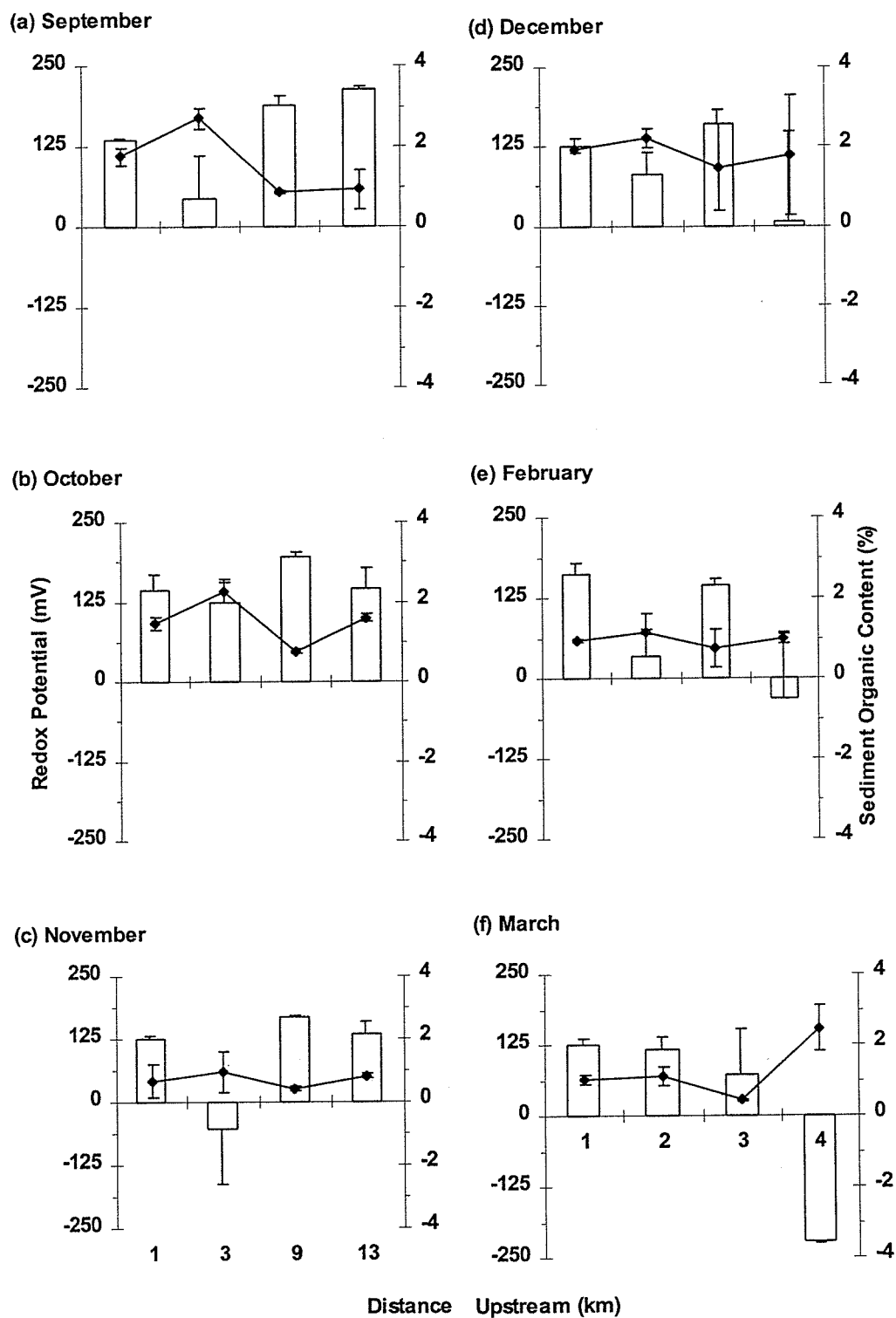


Figure 3.19 A - F: Mean sediment redox potential ($\square$) (in mV), and organic content ($\bullet$) (%LOI), with increasing distance upstream from the Gamtoos Estuary mouth, for the months September 1999 to March 2000 ($n = 3$) $\pm$ SE.

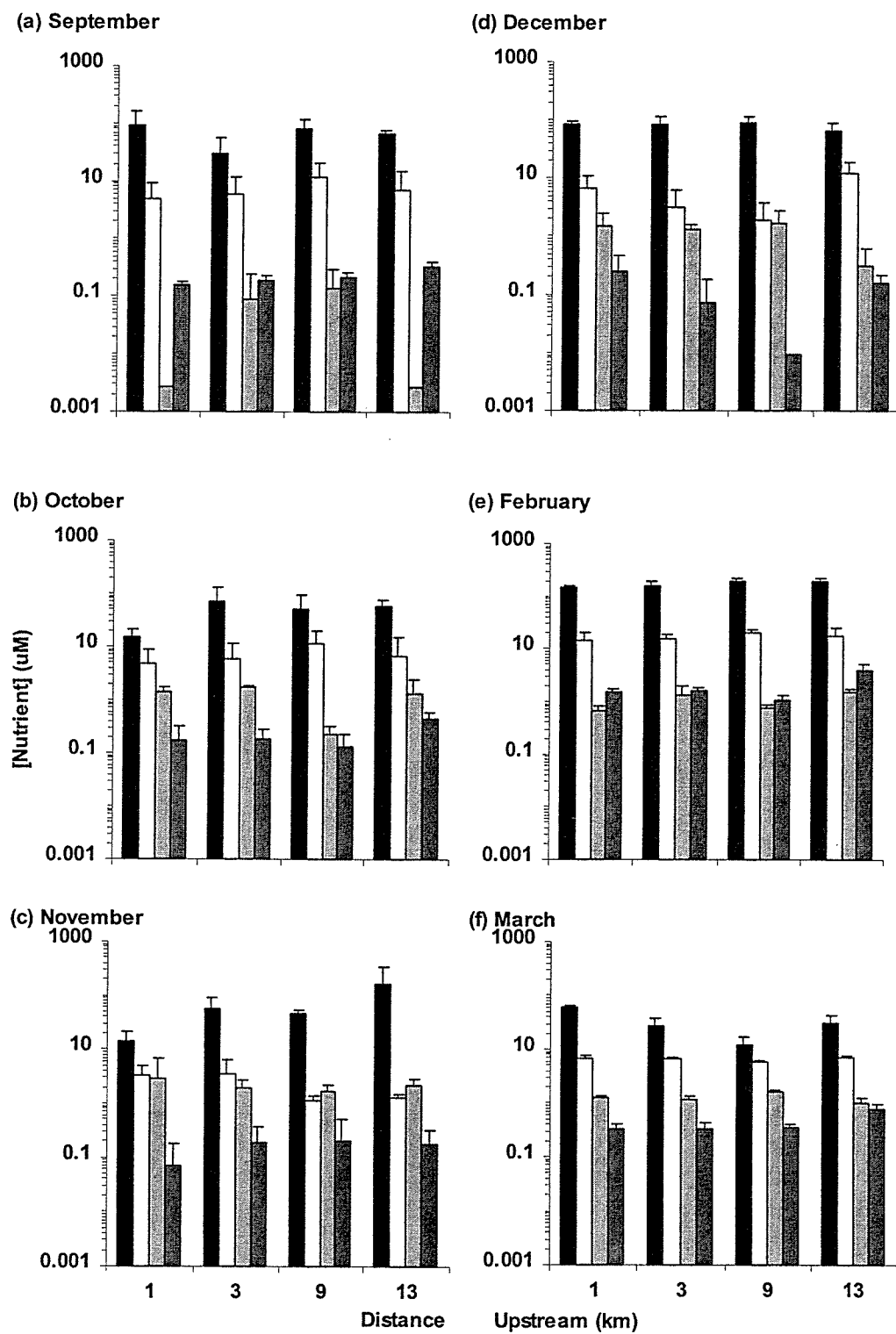


Figure 20 A - F: Mean sediment porewater nitrate (■), nitrite (□), ammonium (▨), and phosphate (▩) concentrations (in μM), with increasing distance upstream (in km) from the Gamtoos Estuary mouth, for the months September 1999 to March 2000 ($n = 3$) $\pm$ SE.

With the exception of ammonium, sediment porewater nutrient concentrations were significantly correlated with salinity (negatively) and temperature (positively) (Table 3.9). Porewater nutrients (except for nitrite) were also found to have a significant negative correlation with dissolved oxygen concentrations of the overlying benthic water. Many significant correlations, both positive and negative were found between porewater and water column nutrients, however, the strongest was between porewater phosphate and water column nitrate concentrations.

3.4 Discussion

3.4.1 Water column

3.4.1.1 Nutrient sources and the REI

The impact of agricultural runoff from the catchment area and the quantity of freshwater input on estuarine water quality has been documented from numerous estuaries on different continents (Thimakorn 1990; Eyre 1994; Pearce and Schumann 1997; Rendell *et al.* 1997). The water quality of the Gamtoos River is equally impacted by agricultural activities in its catchment area and is reflected by enriched nitrate input to the estuary during baseflow conditions. These nitrate concentrations are diluted during freshets and flushing resulting in lower concentrations throughout the estuary. Similar dilution effects for various nutrients in high concentrations in the river water have been documented by Thimakorn (1990), Balls *et al.* (1997), and Scharler *et al.* (1997). In this instance, the river can clearly be identified as a major source point for a particular nutrient during low flow conditions. Phosphate and nitrite concentrations demonstrated different trends than nitrate, in that their concentrations increased during the freshet. The exact reason for the trend cannot easily be explained.

Concentrations of ammonium and PON were relatively lower in inflowing river water during the freshet, but due to the large volume of water reaching the estuary ($8 \text{ m}^3 \text{ s}^{-1}$), overall loads were higher during the freshet than during baseflow conditions. Ammonium and PON could be regenerated in the estuary during baseflow conditions and were diluted during the freshet. The residence time following a freshwater input of $10 \text{ m}^3 \text{ s}^{-1}$ during a freshet is approximately 1 day in the upper reaches and 4 days for the whole estuary (Pearce and Schumann 1997). This period is too short for eutrophic conditions to develop, even if high nutrient loads (of ammonium, nitrite or phosphate) do occur. The prevention of severe estuarine pollution requires an open tidal inlet and efficient water exchange with the adjacent ocean. The relatively high nitrate loads in the estuary during baseflow conditions makes it more prone to eutrophication under low flow conditions. It may be considered that the low phosphate concentrations are unable to support high phytoplankton productivity, but flagellate and dinoflagellate blooms do occur (Scharler unpublished data). Regeneration of phosphate within the system may therefore be a possible source of phosphate to enhance or stimulate primary production.

A similar situation has been observed in the Sundays Estuary, approximately 30 km east of Port Elizabeth, which exhibits phytoplankton blooms based on high nitrate concentrations (received from riverine water enriched with fertilisers), but low phosphate concentrations (Hilmer and Bate 1990; Scharler *et al.* 1997).

The magnitude of river flow into the Gamtoos Estuary could not be identified as the ultimate determinant of water column nutrient status. Only those nutrients for which the REI region appears to be the biggest sink exhibited negative and highly significant correlations with salinity. Although the river was the biggest source for nitrate, this was not apparent for other nutrients such as nitrite and PON. Water from the drainage pipe had in most cases higher concentrations of salts and nutrients compared to the estuarine water in the upper reaches. The volume of water entering the estuary via the pipe ranged from 0.8 to 5 l s⁻¹, and nutrients contained in such small volumes are normally dispersed within the estuary without changing its water quality other than in the immediate vicinity of the drainage pipe. The drainage system drains an area of approximately 50 ha to a depth of 1 m during non-flood conditions (Pearce and Schumann 1997). The nutrient loading from surface runoff and seepage at any one point along the length of the estuary will not be as high as that in the drainage pipe, but nevertheless will supplement nutrients to the estuary. Dense microalgal mats growing in the intertidal zone in the middle and upper reaches are an indication of the agricultural impact on the estuary. The extent of the seepage has unfortunately not been quantified. Pearce and Schumann (1997) measured estuarine groundwater infiltration from fields on the eastern banks of the estuary from 1992 to 1994. The study estimated that groundwater discharge into the estuary from a small stretch (7.7 km) in the Loerie flats was approximately 276 m³ per day. The nitrate loading from a small area of the Loerie flats into the Gamtoos Estuary has roughly been calculated between 10 kg (near Station 9) and 2.7 tons (near Stations 6 and 8), annually (Pearce and Schumann 1997). If that area of the floodplain used for crop cultivation is taken into account, as well as the estimated volume of water discharged into the estuary, then it is clear that the impact on estuarine water quality can be substantial. As a result of these non-point sources of nutrients, it is impossible to nominate the Gamtoos River as the sole source of nutrient loading to the estuary, even for those nutrients which show increasing concentrations towards the head of the estuary. The vegetation on the estuarine banks and the substantial microalgal mats in the intertidal areas may absorb some proportion of the nutrients, and so reduce the influence of seepage and groundwater flow on nutrient concentrations in the water column. The flux of nutrients from the sediment to the overlying water column is considered in the following section, this process may be a significant source of dissolved nitrogen and carbon compounds (Baird *et al.* 1995). The extremely high DIN:PO₄ ratios observed in this study indicate a nitrogen surplus, which is converted to PON:TPP ratios nearer to and below ideal Redfield ratios in the seston (all living and non-living forms present in the water column) component. Furthermore, particulate N:P ratios in the sediment are equally lower than inorganic N:P ratios. Overall, the Gamtoos Estuary acts as a filter for land derived products, which is in concordance with other Eastern Cape estuaries (Scharler *et al.* 1997).

3.4.1.2 Eutrophication in the Gamtoos Estuary

Standing stocks (measured as chlorophyll *a* concentration) of inter- and subtidal benthic microalgae in the Gamtoos Estuary are high relative to phytoplankton standing stocks (Garcia-Rodriguez 1993). Phytoplankton biomass appears to be more affected by water column nutrient status of the estuary than benthic microalgae, as phytoplankton chlorophyll *a* concentrations were high in the Gamtoos Estuary and the similarly impacted Sundays estuary (approximately 10 to 100 times higher compared to more pristine systems). However, benthic microalgal biomass showed similar chlorophyll *a* concentrations for agriculturally impacted and more pristine systems (Hilmer and Bate 1990; Garcia-Rodriguez 1993 and Scharler *et al.* 1997).

Although it was not possible to identify an REI for all nutrients, the low salinity upper reaches appeared to act as a sink for nitrite, nitrate, DOC and PON. In the case of PON, the REI serves as both a source and a sink. The interface region has no marked influence on phosphate concentrations, and ammonium may be generated here. Most of the phosphorus enters the estuary attached to particles as TPP and not in the dissolved inorganic form. TPP concentrations gradually decrease from head to mouth of the estuary without any rapid depletion in the REI. POC possibly has its source in the estuary and ocean, since its concentrations gradually increase from the REI to the mouth. The REI produces PON for the lower reaches of the estuary, whereas its influence on other particulate components (TPP and POC) is not clear. In the case where there was an REI identified for some nutrients, it was in general the salinity reaches of below 17 ‰ that showed higher nutrient concentrations. Morris *et al.* (1978) in turn talk of a freshwater-seawater interface (FSI) of between 0.1 and 1 ‰ which seems an important site for chemical processes regarding nutrients. Rundle *et al.* (1998) recognised a distinct macrobenthic community just before the FSI at < 0.2 ‰.

An REI of < 17 ‰ could be present in the Gamtoos Estuary even under low flow scenarios (see Chapter 2). The importance of high flows (freshets) in the Gamtoos Estuary lies in the dilution of high nutrient concentrations in the estuary and in a flushing effect to counteract nutrient accumulation in the estuary. Floods are necessary to prevent mouth closure and maintain the estuary-ocean connection. Tidal flushing of especially the upper reaches is, compared to the lower reaches of the estuary, fairly restricted (Pearce and Schumann 1997). In the case of the Gamtoos Estuary, therefore, we see for the first time a need for increased flows not as a supply for nutrients, but to dilute and flush accumulated nutrients from the system.

Thus, the water column nutrient status of the Gamtoos Estuary is governed primarily by the agricultural practices along the riverine and estuarine banks. It will vary with fertiliser application, groundwater contamination, rain-induced runoff, and seepage from the banks. Low flow conditions create situations where there is a surplus of nutrients in the estuary. Eutrophication of the system is prevented by the filtering action of microalgae (macroalgae and submerged macrophytes only comprise a very small part of the estuarine flora), in dense mats along its banks. In addition, the estuarine-ocean exchange dilutes nutrient concentrations in the estuary. Should the mouth close as a

result of excessive water abstraction in the catchment area or be severely constricted due to sand accumulation at the flood-tide delta near the mouth, inadequate water exchange may lead to eutrophication of the system. Several scenarios of the residence time of a tracer under different freshwater inflow conditions were modeled by the CSIR (see Chapter 2). Results showed that the concentrations of a tracer (and potentially a nutrient) is reduced to only 14 % in the middle reaches of the estuary after 40 days under low flow conditions of $0.1 \text{ m}^3 \text{ s}^{-1}$. In this and other scenarios it was assumed that the tracer is only present in the estuarine water and not renewed by riverine or oceanic input. Therefore it is the middle reaches which has the longest residence time for these specific scenarios. Under a flow condition of $1 \text{ m}^3 \text{ s}^{-1}$, the tracer is reduced to 7 % of its concentration after 40 days. Only under fairly high freshwater inflow conditions of $5 \text{ m}^3 \text{ s}^{-1}$ was the tracer reduced to < 1 % after 20 days. These results clearly showed that there is a threat of eutrophication under low flow conditions and inadequate flushing with seawater, especially when the nutrient pool is continually renewed from riverine inputs and non-point source leaching of nutrients along the length of the estuary.

Not only the inorganic nutrients are responsible for phytoplankton growth, but dissolved organic nitrogen can also contribute substantially to eutrophication (Deegan *et al.* 1994; Seitzinger and Sanders 1997; Berman *et al.* 1999). Berman *et al.* (1999) state that the breakdown of DON and the subsequent release of biologically available compounds such as urea and ammonium could be an important nutrient supplier to bacteria and phytoplankton. Measurement of DIN accounts for only a portion of the biologically available nitrogen and Seitzinger and Sanders (1997) estimate that from 20 to over 90 % of nitrogen might enter an estuary in its organic form. This has clear implications for managing nutrient input into estuaries. In addition, an average of 25 to 41 % of the dissolved inorganic nitrogen taken up by phytoplankton is released as DON (Bronk *et al.* 1994). The inorganic nitrogen incorporated into phytoplankton biomass is therefore not passed on to higher trophic levels, but instead recycled and made available for further use by bacteria and phytoplankton.

The increased nutrient loadings result in excessive primary production, with algal blooms being a nuisance factor in several ways. For example, blooms of certain species might lead to avoidance by and/or poisoning of secondary consumers (Turner and Tester 1997), and the algal biomass which accumulates during a bloom, will eventually die off and can cause anoxic conditions as the biomass decays and decomposes (Nixon *et al.* 1976; Oviatt *et al.* 1986). Also, as certain algal species show a preference for nutrient rich environments, the diversity in eutrophic ecosystems is likely to be impacted, with consequences for secondary consumers (Gabric and Bell 1993; Hassett *et al.* 1997; Rendell *et al.* 1997; McClelland and Valiela 1998).

3.4.2 Bottom waters

Porewater concentrations recorded in the Gamtoos Estuary for nitrate, nitrite, ammonium, and phosphate ranged from 50 to 100, 5 to 15, 0.2 to 2.2 and 0.15 to 1.5 μM , respectively. Comparative sediment porewater nutrient concentrations for inorganic and phosphorus species reported for several international water bodies are presented in Table 3.10.

Generally, porewater nitrate and nitrite concentrations in the Gamtoos Estuary are comparable to, or slightly higher than, figures listed for systems in Table 3.10, while ammonium and phosphate porewater concentrations are much lower. The combination of low water column and porewater phosphate concentrations indicates that the estuary receives very little allochthonous loading of phosphorus via surface runoff, groundwater infiltration, or riverine inputs, and presumably therefore, internal benthic regeneration of phosphate. In particular, with such severe inorganic phosphate limitation apparent in the system (relative to nitrogen levels), any phosphate that reaches the water column will be rapidly utilised before it may reach and be bound within the sediments. Unlike phosphate, ammonium concentrations present in both the water column and porewater often originate from the bacterial mineralisation of organic nitrogen present in the sediments (ammonification) (Kelso *et al.* 1997). Thus, in environments with high sediment organic content, porewater and water column ammonium concentrations are generally high. However, in the Gamtoos Estuary, sediment organic content was never above 2.7 %, which is relatively low (e.g. Klump and Martens 1989; Tomaszek 1995; Underwood *et al.* 1998). Ammonium may also be generated by the process dissimilatory nitrate reduction to ammonium (DNRA) (Kelso *et al.* 1997). This process is in direct competition with denitrification, and is favoured in nitrate-limited environments, unlike the Gamtoos Estuary. Thus, with low sediment organic content, and conditions not favouring DNRA activity, it is expected that most ammonium present within the sediment will be utilised in the process of nitrification, where ammonium is converted to nitrate via nitrite (Seitzinger 1988).

In this study, significant correlations were found between respective nitrogen species in the porewater and overlying bottom water. Porewater nutrient concentrations have been found to significantly influence bottom water quality in several other systems (Billen 1975; Jenkins and Kemp 1984), while other authors report no significant influence (Kelso *et al.* 1997). Benthic nutrient fluxes are influenced by the hydrodynamic processes diffusion, advection, and sediment resuspension, and these processes may then be influenced by environmental and biological variables present at the location of measurement (Oldham and Lavery 1999). Thus, it is not surprising that conflicting observations are reported. In the Gamtoos Estuary, porewater nutrient concentrations were significantly correlated with bottom water nutrient concentrations for inorganic nitrogen species only. That no significant correlation was found between porewater and bottom water phosphate concentrations is understandable. The high phosphate limitation in the system means available phosphate will be utilised quickly by primary producers present, both at the sediment surface (e.g.

microphytobenthos), and in the water column (e.g. phytoplankton). Therefore, it would be extremely difficult to detect sediment-water phosphate fluxes in this estuary. Alternatively, in estuarine systems with high porewater phosphate concentrations, benthic fluxes are often noticeable (e.g. Berelson *et al.* 1998). In addition, a negative correlation was found between porewater and bottom water ammonium concentrations. That porewater concentrations were generally higher than bottom water concentrations would indicate that water column ammonium is being absorbed by sediments, and here ammonium is inevitably being consumed by the process nitrification (Kelso *et al.* 1997).

Table 3.10: Sediment porewater nutrient concentrations for inorganic nitrogen and phosphorus species reported for various locations.

Reference	Location	Time	Depth(cm)	Sediment			
				[NO ₃] (μ M)	[NO ₂] (μ M)	[NH ₄] (μ M)	[PO ₄] (μ M)
Heggie <i>et al.</i> (1999)	Port Phillip Bay, Australia	Jan	0-1	1	-	-	-
Oldham and Lavery (1999)	Swan River Estuary, Western Australia	Feb	0-1	>140	-	<10	1.25-1.5
Underwood <i>et al.</i> (1998)	Colne Estuary, England	Dec	0-1	40-700	3 - 10	8-270	-
Kemp <i>et al.</i> (1990)	Chesapeake Bay, Maryland	> 2 years	0-1	1 - 65	-	150- 1450	-
Lehtoranta (1998)	Neva Estuary, Gulf of Finland	Jul - Aug	0-1	-	-	16-50	16-50
Overnell <i>et al.</i> (1995)	Loche Linnhe, Scotland	Feb - Jun	0-1	<1-13	-	-	-
Klump and Martens (1989)	Cape Lookout Bight, North Carolina	> 1 year	0-1	-	-	1000	<100-200
Lomstein <i>et al.</i> (1989)	Bering Shelf	Jun - Aug	0-2	-	-	19-55	-
Nishio <i>et al.</i> (1982)	Tama Estuary, Tokyo	May - Sep	0-2	1 - 3	-	-	-
Kaspar <i>et al.</i> (1985)	Tangaroa, New Zealand	Jan	0-7.5	20-62	-	40-450	25-153

No significant correlations were found between rainfall/river flow and water column nutrient concentrations. Other studies have found riverine inputs are both influential (Eyre and Twigg 1997; Eyre and Balls 1999) and not influential on water column nutrient concentrations (Kelso *et al.* 1997; Hubertz and Cahoon 1999). Temperate estuaries generally exhibit flow regimes that are relatively constant (in comparison to tropical estuaries) (Eyre and Balls 1999). A constant flow pattern to the estuary makes nutrient inputs derived from terrigenous leachable materials smaller and more consistent than nutrient loadings that would be associated with riverine flows in tropical estuaries. In addition, small constant flows generate a higher degree of stratification within the estuary, making the likelihood of relatively nutrient-rich freshwaters at the top of the water column mixing with relatively nutrient-poor marine waters at the bottom of the water column less likely (Eyre and Balls 1999). To further compound this scenario, the Gamtoos Estuary has dams in place that no doubt minimises flow rates and water column mixing. Thus, no significant influence of rainfall or river flow on water column nutrient concentrations in the Gamtoos Estuary is apparently understandable.

3.4.3 Conclusions and management implications

In considering the final results and conclusions from these studies, it should be kept in mind that during the periods of investigation, all measurements and observations were made under baseflow conditions with the exception of one freshet. Therefore, the conclusions made here should perhaps be considered in the context of baseflow scenarios of the Gamtoos Estuary, with the future recommendation of qualifying these conclusions with sampling programmes held on the Gamtoos Estuary specifically during freshet and flood periods.

3.4.3.1 Water column

As is apparent from the discussion, it is not only imperative to allow adequate freshwater inflow into estuaries to ensure a continued renewal of the nutrient pool, but also to manage excessive nutrient input into estuaries. Fertiliser application has increased dramatically over the past few decades and the increase in nitrogen and phosphorus application is estimated at approximately 220 and 160%, respectively, in developing countries (Gabric and Bell 1993). Furthermore, atmospheric deposition and groundwater input of allochthonous nutrients is increasingly recognised as a potential source of eutrophication.

In terms of establishing the DWAF ecological reserve for estuaries and determining freshwater requirements, it is imperative to not only consider saline habitats but also to review the nutrient status. If a reduction in nutrient loading to an estuary is not feasible for whatever reason, the estuarine freshwater requirement might increase so as to prevent eutrophication of the system through dilution and ensuring an open tidal inlet.

3.4.3.2 Bottom waters

In this study, sediment porewater inorganic nutrient concentrations were more influential on bottom water nutrient concentrations than riverine inputs. In addition, water column and porewater phosphate concentrations were extremely low, resulting in the Gamtoos estuarine ecology being phosphorus limited. From these observations a number of points for consideration in future management decisions can be made for the system:

1. Sediment porewater nutrient concentrations need to be measured in future water quality programmes as they may significantly influence the nutrient status of the estuary.
2. The application of phosphorus within the catchment needs to be monitored carefully in the hope of maintaining phosphorus-limited conditions within the estuary, considering the water quality problem of inflated inorganic nitrogen levels already apparent. If inorganic phosphorus concentrations increase within the system, so too will the biomass of primary producers, with the potential of management issues such as toxic microalgal blooms arising, over and above those that have already presumably taken place.

With increased freshwater abstraction from the Gamtoos Estuary will come a reduction in the intensity and variability of river flows and subsequent water column mixing in the system. In considering riverine inputs as a point source of nutrients, it appears that this will have few ramifications on water quality due to the large influence of autochthonous sources under baseflow conditions. However, with reduced freshwater inputs and water column mixing, the only parameter able to remove nutrient-rich bottom waters is tidal forcing, which is a relatively consistent variable. In addition, tidal influence will decrease with decreases in freshwater inputs, due to narrowing of the estuarine mouth and possible closure. There is a necessity for river flows with the potential of entire water column mixing that will ensure the occasional removal of nutrient-rich bottom waters from the estuary, or potential water quality problems as observed in other marine-influenced systems (e.g. Swan River Estuary, Western Australia; Thompson and Hosja 1996).

3.5 Acknowledgements

The authors thank the Water Research Commission, the National Research Foundation, and the University of Port Elizabeth for funding this study. The provision of data by the Department of Water Affairs and Forestry, the Municipality of Port Elizabeth, and CSIR is also acknowledged.

3.6 References

- APHA. 1992. Standard Methods of the examination of water and wastewater. American Public Health Association, Washington.
- Aston, S.R. 1980. Nutrients, dissolved gases and general biochemistry in estuaries. In: Chemistry and biochemistry in estuaries (ed. E. Olausson and I. Cato). Wiley and Sons, New York: pp. 233-262.
- Ahel, M., Barlow, R.G. and Mantoura, R.F.C. 1996. Effect of salinity gradients on the distribution of phytoplankton pigments in a stratified estuary. *Marine Ecology Progress Series* 143: 289-295.
- Baird, D., Ulanowicz, R.E. and Boynton, W.R. 1995. Seasonal nitrogen dynamics in Chesapeake Bay: a network approach. *Estuarine Coastal and Shelf Science* 41: 137-162.
- Balls, P.W., MacDonald, A., Pugh, K.B. and Edwards, A.C. 1997. Rainfall events and their influence on nutrient distributions in the Ythan Estuary (Scotland). *Estuarine Coastal and Shelf Science* 44: 73-81.
- Bate, G.C. and Heelas, B.V. 1975. Studies on the nitrate nutrition of two indigenous Rhodesian grasses. *Journal of Applied Ecology* 12: 941-952.
- Berelson, W.M., Heggie, D., Longmore, A., Nicholson, G. and Skyring, G. 1998. Benthic nutrient cycling in Port Phillip Bay, Australia. *Estuarine Coastal and Shelf Science* 46: 917-934.
- Berman, T., Bechemin, C. and Maestrini, S.Y. 1999. Release of ammonium and urea from dissolved organic nitrogen in aquatic ecosystems. *Aquatic Microbial Ecology* 16: 295-302.
- Billen, G. 1975. Nitrification in the Scheldt Estuary (Belgium and Netherlands). *Estuarine and Coastal Marine Science* 3: 79-89.
- Billen, G. 1978. A budget of nitrogen recycling in North Sea sediments off the Belgian Coast. *Estuarine and Coastal Marine Science* 7: 127-146.
- Black, C.A. 1965. Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties (ed. C.A. Black). American Society of Agronomy, Madison.
- Boynton, W.R., Kemp, W.M. and Osborne, C.G. 1980. Nutrient fluxes across the sediment-water interface in the turbid zone of a coastal plain estuary. In: Benthic Nutrient Fluxes (Estuarine Perspectives) (eds W.R. Boynton and W.M. Kemp) Academic Press, New York: pp. 93-109.
- Bremner, J.M. 1965. Total Nitrogen. Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties (ed. C.A. Black). American Society of Agronomy, Madison: pp. 1149-1178.
- Bronk, D.A., Glibert, P.M. and Ward, B.B. 1994. Nitrogen uptake, dissolved organic nitrogen release, and new production. *Science* 265: 1843-1846.
- Callender, E. and Hammond, D.E. 1982. Nutrient exchange across the sediment-water interface in the Potomac River Estuary. *Estuarine Coastal and Shelf Science* 15: 395-413.
- Craft, C.B., Seneca, E.D. and Broome, S.W. 1991. Loss on ignition and Kjeldahl digestion estimating organic carbon and total nitrogen in estuarine marsh soils: calibration with dry combustion. *Estuaries* 14: 175-179.

- Deegan, L.A., Finn, J.T., Hopkinson, C.S., Giblin, A.E., Peterson, B.J., Fry, B. and Hobbie, J.E. 1994. Flow model analysis of the effects of organic matter - nutrient interactions on estuarine trophic dynamics. In: *Changes in fluxes in estuaries: implications from science to management* (ed. K.R. Dyer and R.J. Orth). Olsen and Olsen, Fredensborg: pp. 273-282.
- Eyre, B. 1994. Nutrient biogeochemistry in the tropical Moresby river estuary system North Queensland, Australia. *Estuarine and Coastal Marine Science* 39: 15-31.
- Eyre, B. and Balls, P.W. 1999. A comparative study of nutrient behaviour along the salinity gradient of tropical and temperate estuaries. *Estuaries* 22: 313-326.
- Eyre, B. and Twigg, C. 1997. Nutrient behaviour during post-flood recovery of the Richmond River estuary Northern NSW, Australia. *Estuarine, Coastal and Shelf Science* 44: 311-326.
- Funicelli, N.A. 1984. Assessing and managing effects of reduced freshwater inflow to two Texas estuaries. In: *The estuary as a filter* (ed. V.S. Kennedy). Academic Press, Orlando: pp. 435-446.
- Gabric, A.J. and Bell, R.F. 1993. Review of the effects of non-point nutrient loading on coastal ecosystems. *Australian Journal of Marine and Freshwater Research* 44: 261-283.
- Garcia-Rodriguez, P. 1993. The determination and distribution of microbenthic chlorophyll *a* in selected South Cape Estuaries, South Africa. Botany Department, University of Port Elizabeth. Unpublished Report.
- Gearing, J. N. 1991. The study of diet and trophic relationships through natural abundance ^{13}C . In: *Carbon isotope techniques* (ed. D.C. Coleman and B.F. Coleman). Academic Press, New York: pp. 201-218.
- Hassett, R.P., Cardinale, B., Stabler, L.B. and Elser, J.J. 1997. Ecological stoichiometry of N and P in pelagic ecosystems: comparison of lakes and oceans with emphasis on the zooplankton-phytoplankton interaction. *Limnology and Oceanography* 42: 648-662.
- Heggie, D.T., Skyring, G., Orchard, J., Longmore, A. Nicholson, G. and Berelson, W.M. 1999. Denitrification and denitrifying efficiencies in sediments of Port Phillip Bay: direct determinations of biogenic N_2 and N-metabolite fluxes with implications for water quality. *Marine and Freshwater Research* 50: 589-96.
- Heineken, T.J.E. 1981. Estuaries of the Cape. Part II: Synopses of available information on individual systems. CSIR Report No. 7: Gamtoos (CMS48). CSIR Research Report 406: 40 pp.
- Hilmer, T. and Bate, C. 1990. Covariance analysis of chlorophyll distribution in the Sundays River estuary, Eastern Cape. *South African Journal of Aquatic Science* 16(1/2): 37-59.
- Hubert, E.D. and Cahoon, L.B. 1999. Short-term variability of water quality parameters in two shallow estuaries of North Carolina. *Estuaries* 22: 814-823.
- Jenkins, M.C. and Kemp, W.M. 1984. The coupling of nitrification and denitrification in two estuarine sediments. *Limnology and Oceanography* 29: 609-619.

- Kaspar, H.F., Asher, R.A. and Boyer, I.C. 1985. Microbial nitrogen transformations in sediments and inorganic nitrogen fluxes across the sediment/water interface on the South Island West Coast, New Zealand. *Estuarine Coastal and Shelf Science* 21: 245-255.
- Kelso, B.H.L., Smith, R.V., Laughlin, R.J. and Lennox, S.D. 1997. Dissimilatory nitrate reduction in anaerobic sediments leading to river nitrite accumulation. *Applied and Environmental Microbiology* 63: 4679-4685.
- Kemp, W.M., Sampou, P., Caffrey, J., Mayer, M., Henriksen, K. and Boynton, W.R. 1990. Ammonium recycling versus denitrification in Chesapeake Bay sediments. *Limnology and Oceanography* 35: 1545-1563.
- Klump, J.V. and Martens, C.S. 1989. The seasonality of nutrient regeneration in an organic-rich coastal sediment: kinetic modelling of changing pore-water nutrient and sulfate distributions. *Limnology and Oceanography* 34: 559-577.
- Lehtoranta, J. 1998. Net sedimentation and sediment-water nutrient fluxes in the Eastern Gulf of Finland (Baltic Sea). *Vie Milieu* 48: 341-352.
- Liss, P.S. 1976. Conservative and non-conservative behaviour of dissolved constituents during estuarine mixing. In: *Estuarine Chemistry* (eds. Burton, J.D. and Liss, P.S.). London, Academic Press.
- Lomstein, B.A., Blackburn, T.H. and Henriksen, K. 1989. Aspects of nitrogen and carbon cycling in the northern Bering Shelf sediment. I. The significance of urea turnover in the mineralisation of ammonium. *Marine Ecology Progress Series* 57: 237-247.
- Montagna, P.A. and Kalke, R.D. 1992. The effect of freshwater inflow on meiofaunal and macrofaunal populations in the Guadalupe and Nueces estuaries, Texas. *Estuaries* 15: 307-326.
- McClelland, J.W. and Valiela, I. 1998. Changes in food web structure under the influence of increased anthropogenic nitrogen inputs to estuaries. *Marine Ecology Progress Series* 168: 259-271.
- Morris, A.W., Mantoura, R.F.C., Bale, A.J. and Howland, R.J.M. 1978. Very low salinity regions of estuaries: important sites for chemical and biological reactions. *Nature* 274: 678-680.
- Nishio, T., Koike, I. and Hattori, A. 1982. Denitrification, nitrate reduction, and oxygen consumption in coastal and estuarine sediments. *Applied and Environmental Microbiology* 43: 648-653.
- Nixon, S.W., Oviatt, C.A. and Hale, S.S. 1976. Nitrogen regeneration and the metabolism of coastal marine bottom communities. In: *The role of terrestrial and aquatic organisms in decomposition processes* (eds. Anderson, J.M. and MacFayden, A.). Oxford, Blackwell: 269-283.
- Oldham, C.E. and Lavery, P.S. 1999. Porewater nutrient fluxes in a shallow fetch-limited estuary. *Marine Ecology Progress Series* 183: 39-47.
- Overnell, J., Edwards, A., Grantham, B.E., Harvey, S.M., Jones, K.J., Leftley, J.W. and Smallman, D.J. 1995. Sediment-water column coupling and the fate of the spring phytoplankton bloom in Loch Linnhe, a Scottish Fjordic Sea-Loch. Sediment processes and sediment-water fluxes. *Estuarine Coastal and Shelf Science* 41: 1-19.

- Oviatt, C.A., Keller, A.A., Sampou, P.A. and Beatty, L.L. 1986. Patterns of productivity during eutrophication: a mesocosm experiment. *Marine Ecology Progress Series* 28: 69-80.
- Pearce, M.W. and Schumann, E.H. 1997. The effect of land use on Gamtoos estuary water quality, WRC Report No. 503/1/97: 170 pp.
- Puckridge, J.T., Sheldon, F., Walker, K.F. and Boulton, A.J. 1998. Flow variability and the ecology of large rivers. *Marine and Freshwater Research* 49: 55-72.
- Rendell, A.R., Horrobin, T.M., Jickells, T.D., Edmunds, H.M., Brown, J. and Malcom, S.J. 1997. Nutrient cycling in the Great Ouse Estuary and its impact on nutrient fluxes to The Wash, England. *Estuarine and Coastal Marine Science* 45: 653-668.
- Rundle, S.D., Attrill, M.J. and Arshad, A. 1998. Seasonality in macroinvertebrate community composition across a neglected ecological boundary, the freshwater-estuarine transition zone. *Aquatic Ecology* 32: 211-216.
- Scharler, U.M., Baird, D. and Winter, P.E.D. 1997. Diversity and productivity of biotic communities in relation to freshwater inputs in three Eastern Cape estuaries. Water Research Commission Report No. 463/1/98.
- Schumann, E.H. and Pearce, M.W. 1997. Freshwater inflow and estuarine variability in the Gamtoos estuary: Implications for watershed management. *Estuaries* 20(1): 124-133.
- Seitzinger, S.P. 1988. Denitrification in freshwater and coastal marine ecosystems: ecological and geochemical significance. *Limnology and Oceanography* 33: 702-724.
- Seitzinger, S.P. and Sanders, R.W. 1997. Contribution of dissolved organic nitrogen from rivers to estuarine eutrophication. *Marine Ecology Progress Series* 159: 1-12.
- Solórzano, L. 1969. Determination of ammonia in natural waters by the phenolhypochlorite method. *Limnology and Oceanography* 14: 799-801.
- Strickland, J. and Parsons, T. 1972. A practical handbook of seawater analysis. *Bulletin of Fisheries Research Board Canada* 167: 81-89.
- Thimakorn, P. 1990. Management of nitrate loadings in the Chao Phraya estuary. In: Estuarine water quality management: monitoring, modelling and research. Coastal and Estuarine Studies 36 (ed. Michaelis, W.). Heidelberg, Springer Verlag.
- Thompson, P.A. and Hosja, W. 1996. Nutrient limitation of phytoplankton in the upper Swan River estuary, Western Australia. *Australian Journal of Marine and Freshwater Research* 47:659-667.
- Tomaszek, J.A. 1995. Relationship between denitrification and redox potential in two sediment-water systems. *Marine and Freshwater Research* 46: 27-32.
- Turner, J.T. and Tester, P.A. 1997. Toxic marine phytoplankton, zooplankton grazers, and pelagic food webs. *Limnology and Oceanography* 42: 1203-1214.
- Underwood, G.J.C., Phillips, J. and Saunders, K. 1998. Distribution of estuarine benthic diatom species along salinity and nutrient gradients. *European Journal of Phycology* 33: 173-183.

Chapter 4: Structure and dynamics of estuarine microalgae in the Gamtoos Estuary in relation to the river-estuary interface (REI) region

G.C. Snow, G.C. Bate, H. L. Astill and J.B. Adams

Department of Botany, University of Port Elizabeth

Summary

The increasing demand for freshwater in South Africa has led to an urgent need to determine the freshwater requirements for the ecological maintenance of estuaries (ecological freshwater reserve). A study was initiated on the structure and biomass of microalgae in the Gamtoos Estuary in order to improve our capability to predict changes and determine the acceptability of reduced freshwater input on estuaries. The mean annual runoff of the Gamtoos River has been halved by major impoundments. Agriculture in the Gamtoos Estuary valley contributes high levels of nutrients through groundwater seepage and an extensive sub-surface drainage system.

This study was aimed at determining the responses of microalgae, within the river-estuary interface (REI) region, to the volume of river discharge. A strong longitudinal salinity gradient was present at water flow rates ranging from 0.3 to $30.5 \text{ m}^3 \text{ s}^{-1}$. There was strong vertical stratification at flows above $1 \text{ m}^3 \text{ s}^{-1}$. Nitrate concentration increased with flow from $1.9 \pm 0.2 \text{ } \mu\text{M}$ (at $0.3 \text{ m}^3 \text{ s}^{-1}$) to $83.6 \pm 13.8 \text{ } \mu\text{M}$ ($30.5 \text{ m}^3 \text{ s}^{-1}$). Phytoplankton chlorophyll-a was highest at flows of 0.8 and $1.2 \text{ m}^3 \text{ s}^{-1}$, with subtidal and intertidal benthic chlorophyll-a highest at flow rates of 1 and $1.2 \text{ m}^3 \text{ s}^{-1}$ respectively. A riverwater input rate of $\sim 1 \text{ m}^3 \text{ s}^{-1}$ appears optimal for overall phytoplankton, phytoplankton in the REI and benthic microalgal growth. At this flow rate, the estuary acts effectively at removing excess nutrients from the water before it discharges to the sea.

High levels of mineral elements are imported from the surrounding agricultural areas within the catchment and have resulted in the Gamtoos Estuary becoming eutrophic. The high concentration of nutrients resulted in high numbers of phytoplankton cells, up to a maximum average of $20958 \pm 1750 \text{ cells ml}^{-1}$ at a flow of $0.8 \text{ m}^3 \text{ s}^{-1}$. Phytoplankton group diversity and evenness were highest at a river flow rate of $1 \text{ m}^3 \text{ s}^{-1}$. Flagellates and diatoms were the dominant phytoplankton groups during the entire study. Average flagellate numbers were highest in the upper reaches of the estuary, when the flow rate was less than $9.7 \text{ m}^3 \text{ s}^{-1}$. As river flow and average nitrate concentration decreased, flagellate numbers increased reaching a maximum of $8497 \pm 773 \text{ cells ml}^{-1}$ at a flow of $0.3 \text{ m}^3 \text{ s}^{-1}$. Diatom numbers increased with flow rate of the river, reaching a maximum of $7199 \pm 2387 \text{ cells ml}^{-1}$ at $1.2 \text{ m}^3 \text{ s}^{-1}$. Dinoflagellates and coccolithophorids were dominant but only during stable, low flow conditions when vertical stratification was relatively weak.

The influence of sediment characteristics and conditions in the interstitial waters (porewater) of the sediment on subtidal and intertidal microphytobenthos communities in the REI region of the Gamtoos Estuary were also investigated. Porewater phosphate concentrations were positively correlated with microalgal biomass. Benthic microalgal biomass was significantly higher in the intertidal region compared to the subtidal region. Microalgal biomass was highest in those reaches of the estuary where the salinity was between 10 and 15 ‰. This region would be negatively affected by freshwater abstraction, as the expanse of mesohaline waters would be reduced and the longitudinal position of the REI region altered.

If base flow is removed from a permanently open estuary the longitudinal and vertical salinity gradients are lost. The range of salinity produced by riverwater input creates a suite of niches along the longitudinal axis of the estuary. This supports a broad range of microalgal species occurring in the estuary, on which a number of higher trophic levels are dependent. In addition to the loss of biodiversity, a lack of river water prevents the necessary biotic and abiotic functions to occur within the REI. Amongst these functions, base flow introduces mineral nutrients essential for the increased phytoplankton biomass associated with the REI. The Gamtoos is a eutrophic estuary, further reduction in river input may lead to a lower phytoplankton biomass and increased dominance of flagellates.

4.1 General Introduction

The requirements for freshwater in a developing country such as South Africa are considerable. As a result of the rapidly growing population, there is an urgent need to determine the freshwater requirements for the ecological maintenance of estuaries (ecological reserve). A multidisciplinary project undertaken by the Consortium for Estuarine Research and Management (CERM) for the Water Research Commission (WRC) aimed to improve the understanding of the interactions between river flow, salinity distribution and the ecology of estuaries. A joint project between the University of Port Elizabeth's Botany and Zoology Departments, the CSIR in Stellenbosch and the JLB Smith Institute of Ichthyology in Grahamstown was initiated to address a number of aims, with particular reference to the river estuary interface (REI) region.

Hilmer and Bate (1990) identified a turbid, low salinity region of increased phytoplankton chlorophyll-a in the Sundays Estuary. Mean chlorophyll-a was lowest in river water and water near to the mouth of the estuary. A maximum in chlorophyll-a occurred at a salinity of 3 to 7 ‰, two thirds of the length of the estuary from the mouth, and near to the water surface (zero to 0.5 m depth). The results of the Sundays Estuary study focused attention on this particular region (REI) of the estuary. The REI is considered to be the ebb and flow region where the vertically averaged salinity is usually < 10 ‰, but may rise above this value under low flow conditions. It is defined as the distance (km) from the mouth of the estuary where the maximum in phytoplankton chlorophyll-a occurs.

To be able to address the botanical (microalgal) component of the overall REI study, a flow diagram (Figure 4.1.1) was developed, which includes the biotic and abiotic factors that potentially affect estuarine microalgae. The diagram assisted in highlighting poorly understood links between microalgae, physico-chemical factors and other trophic levels.

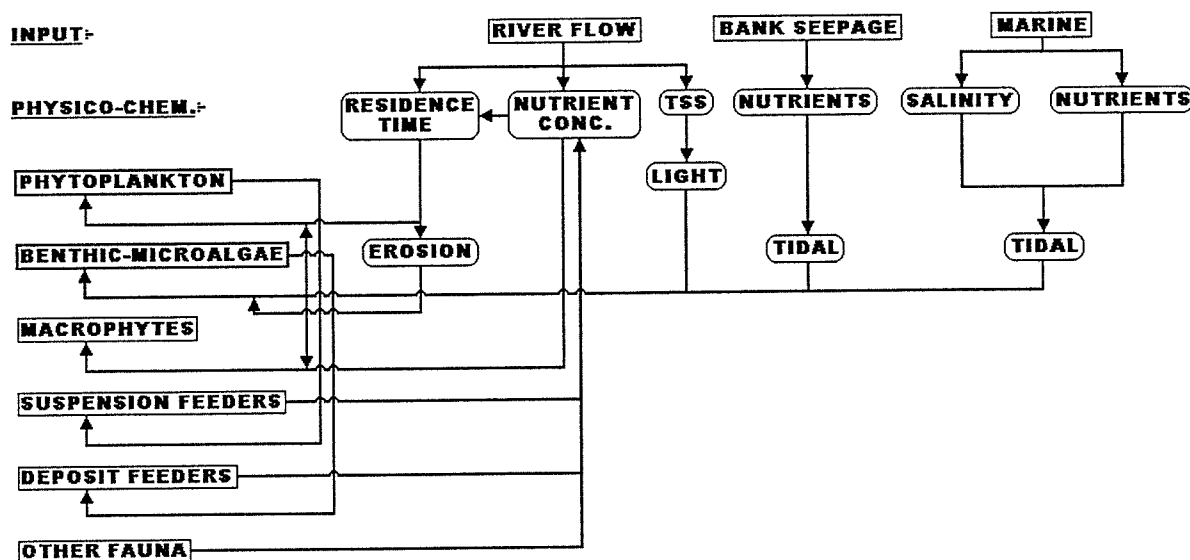


Figure 4.1.1: Flow diagram of biotic and abiotic factors that potentially affect estuarine microalgae.

The most important factor governing the productivity of microalgae in the estuary was assumed to be river flow. River flow affects residence time, nutrient input from the catchment, and total suspended solids (TSS). The combined effects of river flow were considered to be of greater influence on microalgae within the REI than groundwater seepage or marine water input. Once the flow diagram was complete a number of hypotheses were proposed:

1. *Phytoplankton biomass would peak at a flow rate of between 1 and 8 m³ s⁻¹ in the Gamtoos Estuary.* The basis of this hypothesis was that earlier observations had indicated that very low flow rates resulted in low measurable levels of chlorophyll-a in the water. Very high flow rates caused the water to pass through the estuary too quickly for algae to grow to a high population density.
2. *Three tidal cycles (spring-neap-spring) are required for maximum phytoplankton production.* This hypothesis was based on the results of a similar study in the nearby Sundays River Estuary (Hilmer 1990). It was related to the residence time of water within the estuary.
3. *Benthic chlorophyll-a is high when water flow is low because sediment erosion is low.* The basis of this hypothesis was that intertidal sediment appears to be very stable in the Gamtoos Estuary, due partly to the presence of algal mats. Observations have shown that these can be

eroded at high rates of water flow. High flow rates also appeared to be associated with very coarse and mobile subtidal sediments. Hence, the impression was that increased sediment stability at low flow would likely result in a high benthic microalgal population.

4. *As the flow of water entering the estuary increases, so the average measurable plant mineral nutrient concentration of the water increases.* The basis of this was that at low flow, a high water residence time would result in the almost complete absorption of dissolved mineral nutrients. At very high flow rates, the mineral uptake would be almost zero.

4.2 Effect of river flow on microalgal biomass and distribution in the Gamtoos Estuary

4.2.1 Introduction

Much of South Africa is semi-arid, necessitating construction of impoundments on larger rivers to supply the water needs of a growing population. As a consequence, the volume of water flowing into estuaries and the intensity of floods has been reduced (Allanson and Read 1995; Reddering 1988; Schumann and Pearce 1997). The present use of water in South Africa is often wasteful and inefficient. As a result, the National Water Act of 1998 recognizes two rights, namely the right to sufficient water to meet basic human needs and the right to have the environment protected for the benefit of present and future generations, i.e. a freshwater reserve.

This study reports on a component of a multi-disciplinary research project. The aim was to improve the capacity to predict changes and determine acceptability of reduced water input, with special reference to the structure and biomass of microalgal populations within the river-estuary interface (REI) region. The REI is defined as the distance (km) from the mouth of the estuary where a phytoplankton chlorophyll-a maximum occurs. The hypothesis tested was that the structure and functioning of the REI is governed by the quality and supply pattern of freshwater received. Results from this study will provide essential information on the microalgal biomass in the REI and its degree of dependence on freshwater for essential processes. Knowledge gained will also lead to a better understanding of the consequences of freshwater attenuation in estuaries.

4.2.2 Study area

The Gamtoos River Estuary is a permanently open estuary (Figure 4.2.1), with a shallow (< 1.5 m) tidal inlet, a narrow channel (< 50 m) and, for the first 1.5 km upstream, an extensive flood tide delta where the shallow channel widens to ~250 m. Beyond the flood tide delta, the main channel deepens to ~4 m in the middle reaches, but from 8 km upstream from the mouth the estuary becomes progressively narrower (< 100 m) and shallower (< 2.5 m). This trend continues to the tidal head (~20 km) where water is less than one metre deep (Schlacher and Wooldridge 1996a).

The estuary is shallow, mud-dominated and turbid, entering the sea through a dynamic coastal dunefield (Heineken 1981). The estuary above the flood-tide delta is channelled, with steep banks and a narrow intertidal region. The tributaries flowing into the Gamtoos River have an extensive drainage area, about 34 500 km² extending far into the interior. The large Kouga Dam supplying Port Elizabeth, and the agricultural requirements of the area have reduced the supply of riverwater to the estuary. The total reservoir capacity within the Gamtoos catchment is 255 x 10⁶ m³ (Midgley *et al.* 1994). The mean annual runoff of the catchment is 495 x 10⁶ m³, which is only twice total reservoir capacity. Evaporation and abstraction of river water further reduce flow to the estuary. In 1997, the estimated average flow into the estuary was less than 1 m³ s⁻¹ (Schumann and Pearce 1997).

An extensive surface and subsurface agricultural drainage system directs runoff from the low-lying Loerie Flats region, approximately 50 ha north and east of site 9 (Figure 4.2.1), towards the estuary. An agricultural drainage pipe empties into the estuary 16.5 km from the mouth. Outflow at the pipe has been measured at between 0.07 and 11 l s⁻¹ (Schumann and Pearce 1997). The contribution of water to the estuary was considered insignificant, but nutrient concentrations, particularly in the upper estuary, were significant. The agricultural drainage pipe which discharges runoff from an area of approximately 50 ha and throughflow from the upper 1.2 m of soil into the estuary at site 9, gives an indication of the contribution of nutrients from surface runoff of the agricultural fields to the estuary. Pearce (1996) estimated that on an annual basis, approximately 75 kg of phosphorous, 129 kg of nitrate and 16 kg of nitrite entered the estuary from the surface agricultural drainage system alone. A mixing zone exists around the discharge point 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.

Total groundwater discharge also contributes significant levels of nutrients to the Gamtoos Estuary. Pearce (1996) estimated that between 10 and 2 700 kg of nitrate-N, 1 and 89 kg of nitrite-N and 10 and 400 kg of phosphorous-P were contributed by total groundwater discharge to the estuary per annum. These estimates were regarded as 'conservative' and only represented inputs from an area that comprises less than 20 % of the total area bordering the estuary.

4.2.3 Materials and Methods

4.2.3.1 Flow

Accurate measurements of river flow into the Gamtoos Estuary were not possible during this study as the nearest streamflow gauges were over 50 km from the head of the estuary, just before the confluence of the two major tributaries. These flow gauges are sited at the Kouga River (L8R001) and the Groot River (L9R001). Variation in the river flow between these gauges and what enters the

estuary are as a result of evaporation, point-source and diffuse seepage, localised rainfall and abstraction for agriculture. It is assumed that these factors have a minimal effect under flow conditions greater than $2 \text{ m}^3 \text{ s}^{-1}$, but under low flow conditions, the effect becomes more pronounced. Combined streamflow recordings of the two gauges are displayed in Figure 4.2.2.

The effects of selected flow conditions on salinity distributions within the Gamtoos Estuary, utilising the Mike 11 modeling system, are given in Chapter 2. This model is designed to yield accurate simulations of water level variations, volume flows, velocities and cross-sectional average salinity along the length of the estuary over time periods from days to months. The model was calibrated for neap (3 November 1996) and spring tide (11 November 1996) conditions. A pulse in freshwater occurred between these two measurements, making the data less suitable for model calibration. However, in the absence of other data they were used here, and are considered reasonably accurate salinity simulations. The Mike 11 model was used to simulate longitudinal salinity distributions for flows from 0.1 to $20 \text{ m}^3 \text{ s}^{-1}$.

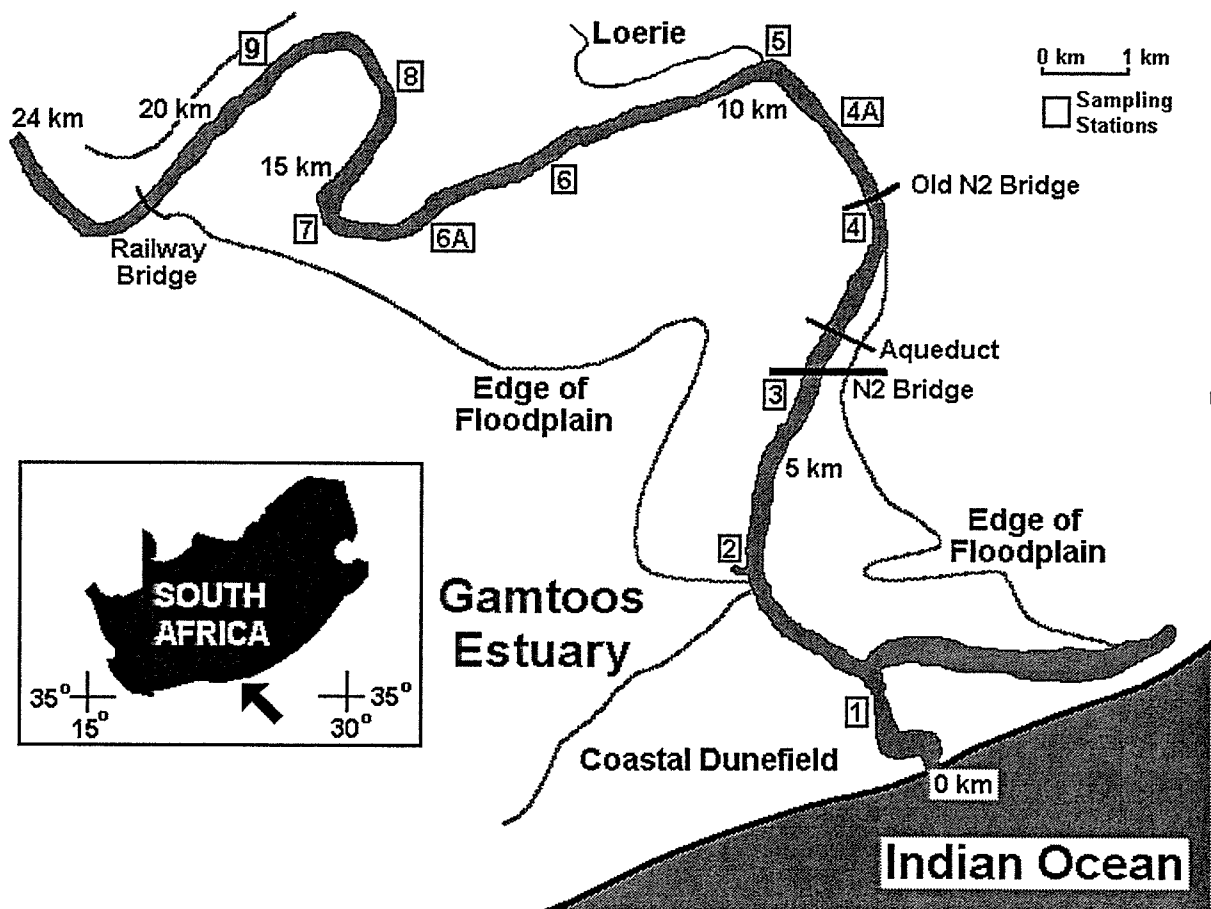


Figure 4.2.1: Locality map of the Gamtoos River Estuary indicating the position of sampling sites (squared numbers).

Flow estimates were determined from the Mike 11 simulations by using the distance (in kilometers) from the mouth of the estuary where mean vertical salinity was 10 ‰. A major flood event (22 November 1996) changed the morphology of the mouth of the estuary. This means that estimates of water flow may be in error. However, we believe that the error is unlikely to be greater than 20%, which means that the interpretation of the results into acceptable volumes of base flow will still be reasonable.

4.2.3.2 Sampling

Ten sampling sessions took place between November 1996 and February 1998. Site 1 was near the estuary mouth within the flood-tide delta, and site 9 near the tidal head of the estuary. Further upstream was considered riverine and above the estuarine tidal influence. Data from 293 depth profiles were collected and salinity, nitrate and chlorophyll-a measured. Water samples were collected using a 500 ml pop-bottle at zero, 0.5 and 1 m and thereafter at 1 m intervals to the bottom, then filtered through glass-fibre filters (Schleicher and Schüll No. 6). Salinity prior to 25 October 1997 was measured using a WTM model LFG 191 conductivity / temperature meter (CTD) until it became unserviceable. Later, salinity was measured using a calibrated refractometer (American Optical).

4.2.3.3 Light attenuation

Light attenuation was determined using a Li-Cor Li 192 S quantum sensor attached to a Li 185a light meter. The vertical attenuation coefficient was determined for the top 1 m of each profile as: $K \text{ (m}^{-1}\text{)} = 1.7 (\ln [I \text{ surface} / I \text{ 1m}])$, where I is the scalar irradiance just below the surface and at one metre depth. A secchi disc was also used to determine light attenuation. The vertical attenuation coefficient was determined as described by Dawes (1981): $K \text{ (m}^{-1}\text{)} = (1.7 / \text{secchi depth})$. There was no significant difference between the attenuation coefficients determined by light meter and Secchi disc and the values were averaged for each site.

4.2.3.4 Nitrate-N

Nitrate N was determined using a method adapted from Bate and Heelas (1975). The nitrate in the filtered (Whatman GF/C filters) water samples was reduced to nitrite then determined as NO_2^- -N plus NO_3^- -N. Nitrite was quantitatively incorporated into a diazo-couple (red/purple) compound and then the concentration determined at 540 nm using a GBC UV-VIS spectrophotometer. All results were expressed in μMolar values as these units obviate any problems of interpretation between N and other oxidized nitrogen (NO_x) compounds.

4.2.3.5 Phytoplankton chlorophyll-a

Water samples (500 ml) were gravity filtered through plastic Millipore filter towers with Whatman (GF/C) glass fibre filters. The chlorophyll-a was extracted by placing the filters into glass vials with 10 ml of 95 % ethanol (Merck 4111). The samples were then stored overnight at 1 to 2 °C. The contents of the vials were filtered and the light absorbance at 665 nm of the supernatant was determined, before and after adding two drops of 0.1 N HCl, using a GBC UV-VIS spectrophotometer. Chlorophyll-a was calculated using the equation of Hilmer (1990) that had been derived from that of Nusch (1980).

4.2.3.6 Plotting

Surfer for Windows (Golden Software®), version 6, was used to create contour profiles of nitrate, phytoplankton chlorophyll-a and salinity. Kriging interpolation was the gridding method used to plot the profiles.

4.2.3.7 Benthic chlorophyll-a

Water content is one of the most important factors influencing the extraction of chlorophyll-a in benthic samples. This changes in relation to sediment type (sand, silt or clay). Rodriguez (1994) determined that chlorophyll-a extracted from a 10 mm deep and 20 mm wide core of sediment placed in 30 ml of 95 % ethanol would be least affected by sediment type and water content. The final ethanol concentration of the sample would be > 90 %, which is the minimum concentration necessary for optimal chlorophyll-a abstraction (Nusch 1980). Chlorophyll-a from the intertidal and subtidal benthic samples was extracted at low temperature (1 to 2 °C) overnight then determined using high performance liquid chromatography (HPLC).

The benthic chlorophyll-a content of the sediment was determined by first centrifuging and then filtering the samples through glass fibre filters (Whatman GF/C). The extract was analysed on a high performance liquid chromatograph (HPLC) attached to a Waters Lambda-Max 481 LC spectrophotometer and Waters LM-45 solvent delivery system. A 30% methanol and 70 % acetone mixture was used as a carrier. The system was calibrated using the chlorophyll-a of red seaweed (*Plocamium corallorhiza*) because it contains no chlorophyll-b to interfere with the chlorophyll-a reading at 665 nm. The seaweed was crushed using a mortar and pestle in 20 ml of 95% ethanol and filtered through a glass fibre filter (Whatman GF/C). The chlorophyll-a concentration was determined from the absorbance reading with the equation used by Nusch (1980). Chlorophyll-a is expressed as average biomass (kilograms) for the entire estuary at each flow rate. These were obtained by multiplying average site chlorophyll-a concentrations (mg m^{-2}) by the approximate area of the intertidal and subtidal regions. Mean values are reported with the standard error of the mean.

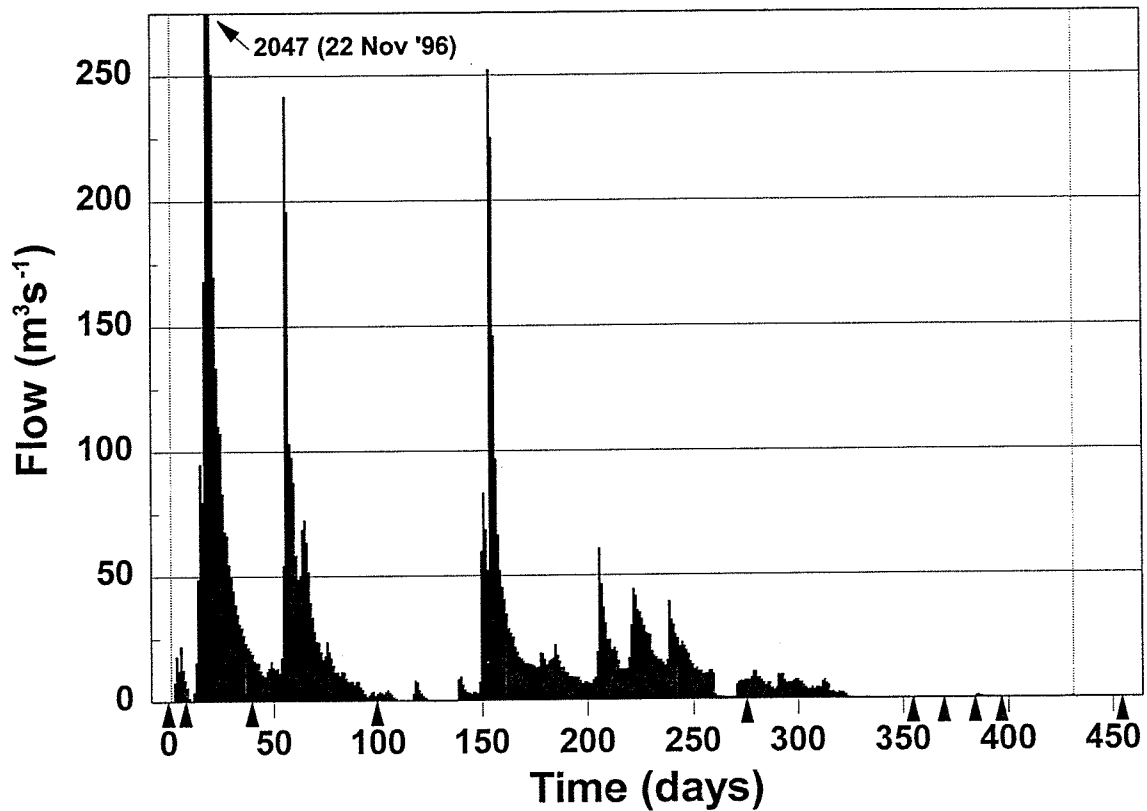


Figure 4.2.2: Combined streamflow recordings (flow recorders L8R001 and L9R001) in the Gamtoos River catchment. Includes flow estimate ($\text{m}^3 \text{s}^{-1}$) for flood of 22 November 1996 and arrows indicate sampling dates starting 4 November 1996 (day 0).

4.2.4 Results

The flow rates determined from the surface salinity data ranged from 30.5 to $0.3 \text{ m}^3 \text{s}^{-1}$ (Table 4.2.1). The sampling, from 25 October 1997 onwards, was restricted to neap tides to reduce the number of variables affecting the REI (e.g. tidal flow). Average river flow for the period 7 November 1996 to 8 November 1998 (Figure 4.2.2) measured at the gauging stations was $25.6 \pm 5.9 \text{ m}^3 \text{s}^{-1}$. This average includes the flood event of November 1996, when flow exceeded $2000 \text{ m}^3 \text{s}^{-1}$. A flow of this magnitude would flush the estuary entirely and change the morphology of the estuary mouth (Reddering and Scarr 1990). Flow thereafter gradually decreased over the following 14 months and the results illustrate a post-flood recovery, particularly with regards to salinity, nitrate and microalgal biomass. The river flow (Figure 4.2.2) showed no discernible seasonal pattern.

There was a strong longitudinal salinity gradient under all flow rates (Figure 4.2.3). In addition, strong vertical salinity stratification was always present at flow rates in excess of $1 \text{ m}^3 \text{ s}^{-1}$. As river water input decreased from 30.5 to $0.3 \text{ m}^3 \text{ s}^{-1}$ after the flood, the saline wedge penetrated from a distance of 8 km to more than 17.5 km from the mouth. In contrast to salinity, nitrate concentrations decreased with depth and towards the mouth of the estuary (Figure 4.2.4). At a flow of $0.3 \text{ m}^3 \text{ s}^{-1}$ all measurable nitrate was localised into the 11 to 16.5 km region of the estuary, a region bordered by the Loerie Flats agricultural land, which contributes nutrients through groundwater seepage and outflow from the agricultural drainage pipe. At flow rates of $0.8 \text{ m}^3 \text{ s}^{-1}$ and above, nitrate concentrations were highest at the head of the estuary, indicating a fluvial origin of nitrate. Average nitrate concentration for the whole estuary decreased as river flow decreased. At a flow rate of $30.5 \text{ m}^3 \text{ s}^{-1}$ the average nitrate concentration was $83.6 \pm 13.8 \text{ } \mu\text{M}$ but decreased by almost two orders of magnitude to $1.94 \pm 0.21 \text{ } \mu\text{M}$ at a flow rate of $0.3 \text{ m}^3 \text{ s}^{-1}$. Under low flow conditions ($0.8 - 0.3 \text{ m}^3 \text{ s}^{-1}$), nitrate concentrations in excess of $20 \text{ } \mu\text{M}$ were confined to the shallow middle and upper reaches ($>10 \text{ km}$ from the mouth) of the estuary.

Phytoplankton chlorophyll-a was unevenly distributed at low flow rates (Figure 4.2.5). The highest levels were within the upper half of the estuary at flow rates of $< 2.3 \text{ m}^3 \text{ s}^{-1}$. At 9.7 and $30.5 \text{ m}^3 \text{ s}^{-1}$ chlorophyll-a levels were low, but well distributed throughout the estuary. Average phytoplankton chlorophyll-a for the whole estuary (Figure 4.2.6) was highest at flow rates of $0.8 \text{ m}^3 \text{ s}^{-1}$ ($47.5 \pm 4.5 \text{ } \mu\text{g l}^{-1}$) and $1.2 \text{ m}^3 \text{ s}^{-1}$ ($49.9 \pm 7.3 \text{ } \mu\text{g l}^{-1}$). Using one way ANOVA on ranks, these maxima were only significantly higher ($n = 69$, $p < 0.05$) than average phytoplankton at a flow of $30.5 \text{ m}^3 \text{ s}^{-1}$ (12 November 1996). At higher flow rates ($> 1.2 \text{ m}^3 \text{ s}^{-1}$) chlorophyll-a decreased to $6.9 \pm 0.6 \text{ } \mu\text{g l}^{-1}$ ($30.5 \text{ m}^3 \text{ s}^{-1}$) which were the lowest levels measured. A Pearson Product Moment Correlation showed that the overall relationship between phytoplankton chlorophyll-a and flow was negative ($n = 221$, Pearson's $r = -0.70$, $P = 0.08$). However, phytoplankton chlorophyll-a increased from $27.1 \pm 6.9 \text{ } \mu\text{g l}^{-1}$ to $47.5 \pm 4.5 \text{ } \mu\text{g l}^{-1}$ as flow increased from 0.3 to $1.2 \text{ m}^3 \text{ s}^{-1}$. This positive relationship ($n = 126$, Pearson's $r = 0.89$, $P = 0.11$) under low flow conditions corresponds well to results obtained from previous Eastern Cape studies (Hilmer and Bate 1990, 1991) where positive correlations between river input and phytoplankton chlorophyll-a were found.

The highest mean vertical chlorophyll-a value ($115 \text{ } \mu\text{g l}^{-1}$), or REI, was situated in the upper reaches of the estuary ($> 12 \text{ km}$ from mouth) at a flow rate of $1 \text{ m}^3 \text{ s}^{-1}$. This maximum occurs in the same region of the estuary as the highest mean vertical nitrate concentration ($145 \text{ } \mu\text{M}$) at the same flow. Mineral nutrients and light are probably made more available to phytoplankton in the upper reaches of the estuary because of the lower volume and shallower water-column. The Student-Newman-Keuls multiple comparisons test showed that mean chlorophyll-a in the upper reaches ($>12 \text{ km}$; $62.4 \pm 11.9 \text{ } \mu\text{g l}^{-1}$) was significantly higher than the mean middle (6 to 12 km ; $28.1 \pm 5.8 \text{ } \mu\text{g l}^{-1}$) and lower reach ($< 6 \text{ km}$; $12.9 \pm 3.5 \text{ } \mu\text{g l}^{-1}$) values ($n = 19$, $P < 0.01$).

Table 4.2.1 Position of the vertically averaged 10 ‰ salinity, tidal state and river flow of the 10 sampling sessions from late 1996 to early 1998. Flow was estimated using CSIR salinity simulations (van Niekerk and Huizinga 1997).

Date	Tide	Estimated flow ($\text{m}^3 \text{s}^{-1}$)
11/12/96	Spring	30.5
13/12/96	Spring	30.5
04/11/96	Neap	9.7
07/08/97	Between tides	2.3
11/02/97	Between tides	1.6
25/10/97	Neap	1.24
08/11/97	Neap	1.16
06/12/97	Neap	1
23/11/97	Neap	0.75
02/02/98	Neap	0.3

The attenuation of light through the water column was always greater near the head of the estuary compared to the mouth (Figure 4.2.7). The region of maximum turbidity for flow rates of $2.3 \text{ m}^3 \text{s}^{-1}$ and lower was also the region of maximum phytoplankton chlorophyll-a, i.e. the REI region of the estuary. As flow increased from 0.3 to $30.5 \text{ m}^3 \text{s}^{-1}$ the REI (Figure 4.2.8) gradually shifted downstream from 16.5 to 6.8 km from the mouth. Chlorophyll-a levels were well distributed at 9.7 and $30.5 \text{ m}^3 \text{s}^{-1}$ making the position of the REI almost indistinguishable at these high flows.

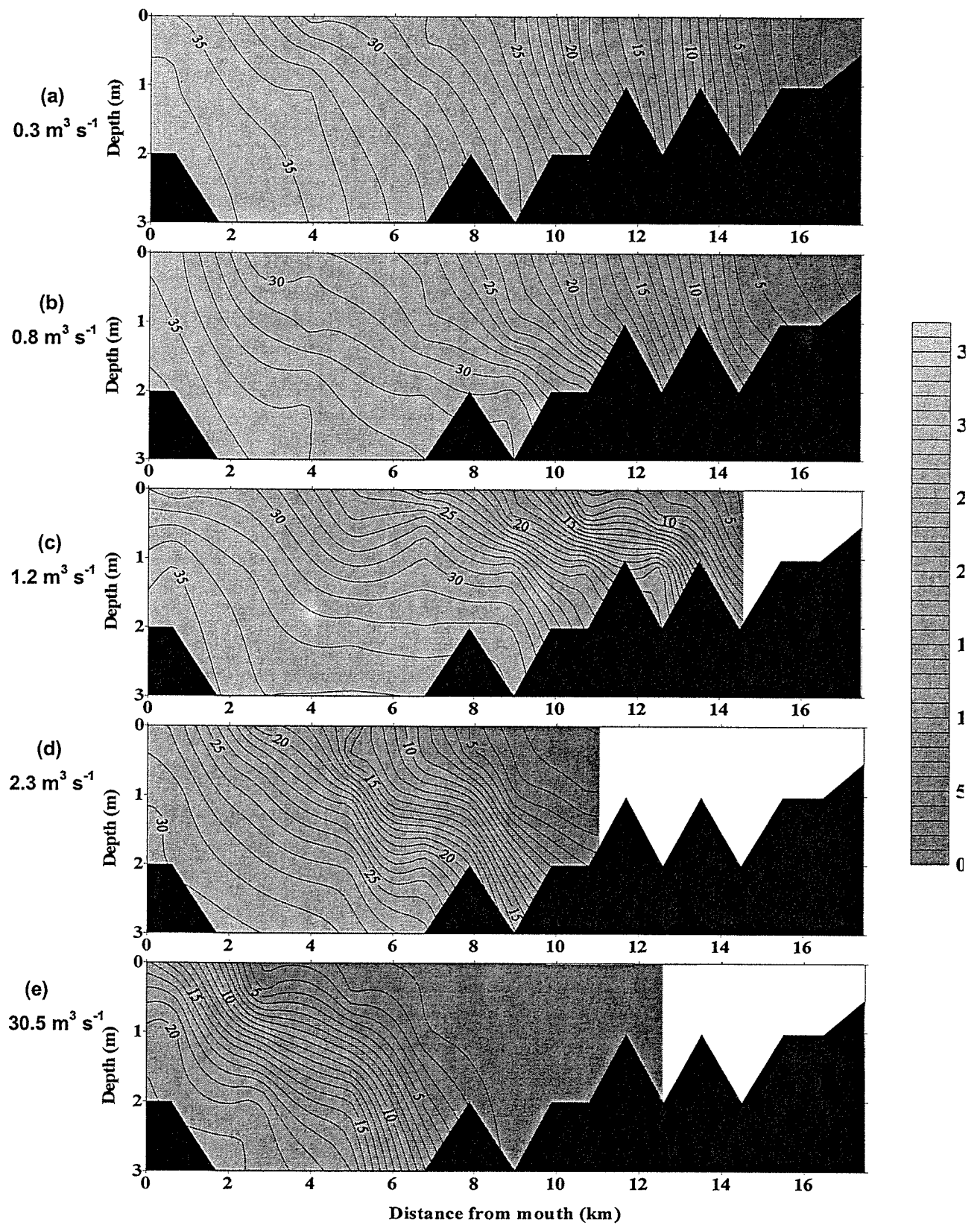


Figure 4.2.3: Salinity profiles (‰) in the Gamtoos Estuary measured at flows from 0.3 to $30.5 \text{ m}^3 \text{ s}^{-1}$.

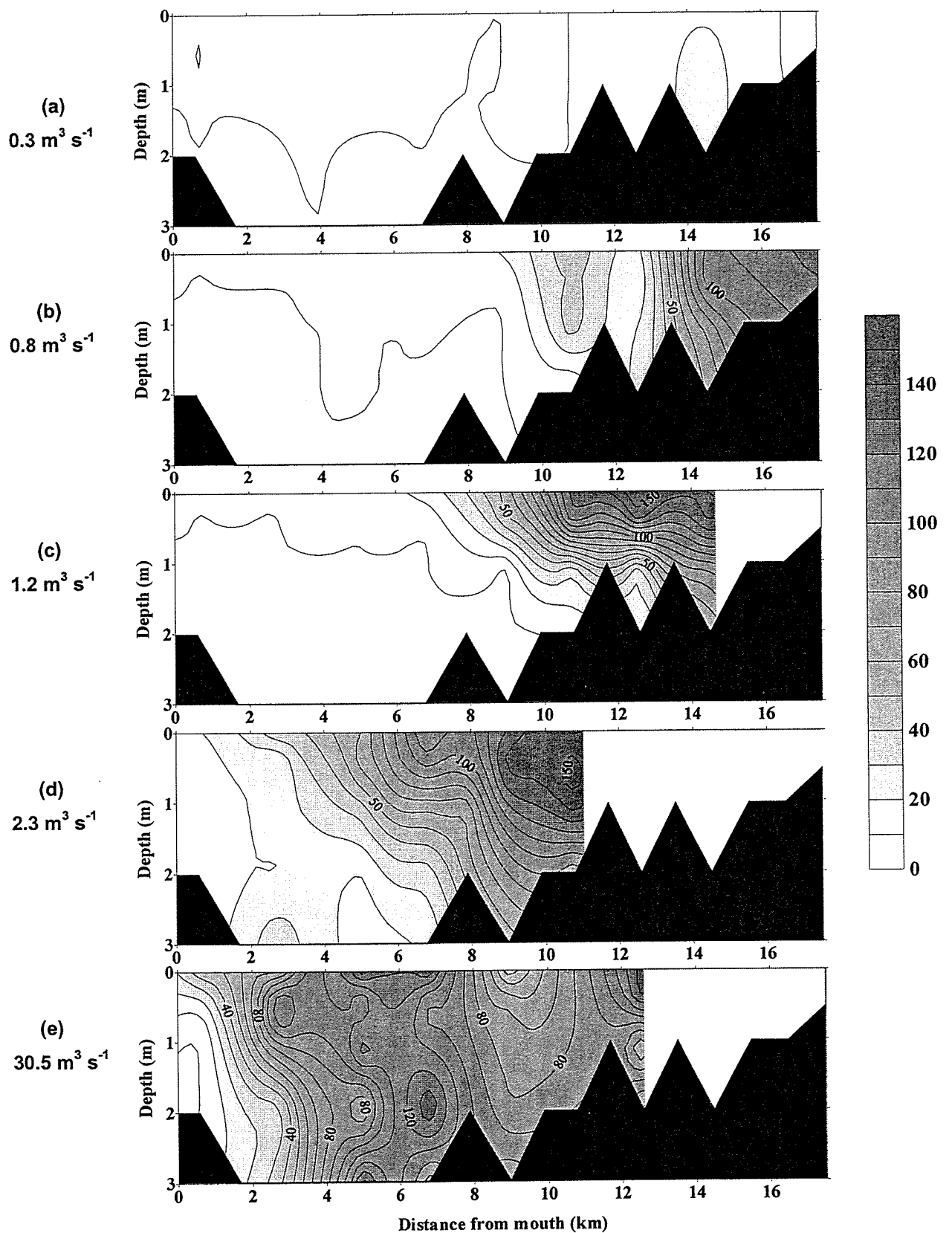


Figure 4.2.4: Nitrate profiles (μM) in the Gamtoos Estuary measured at flows from 0.3 to $30.5 \text{ m}^3 \text{ s}^{-1}$.

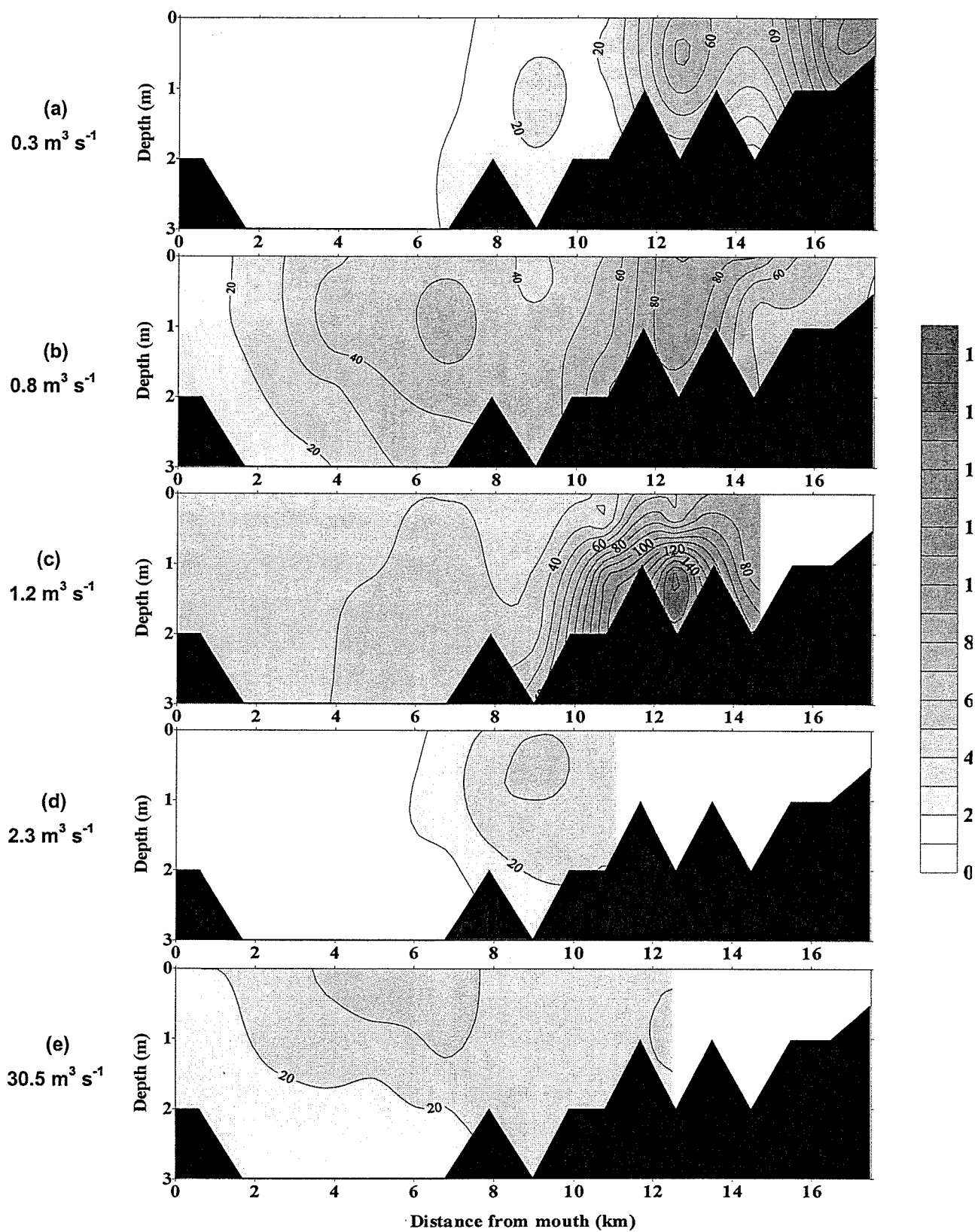


Figure 4.2.5: Water column chlorophyll-a ($\mu\text{g l}^{-1}$) in the Gamtoos Estuary measured at flows from 0.3 to $30.5 \text{ m}^3 \text{ s}^{-1}$.

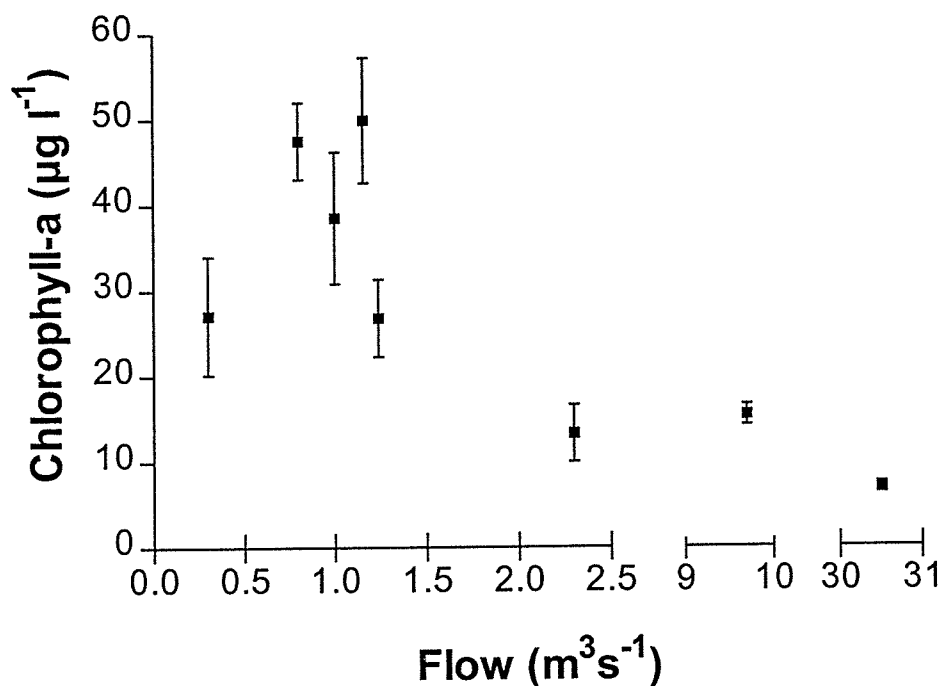


Figure 4.2.6: Average water column chlorophyll-a concentration ($\mu\text{g l}^{-1}$) in the Gamtoos Estuary as a function of flow ($\text{m}^3 \text{s}^{-1}$).

Total benthic chlorophyll-a biomass (Figure 4.2.9) was calculated in relation to the total area for both the subtidal and intertidal regions. The Gamtoos Estuary is steep-sided with a narrow intertidal zone, so the subtidal region makes up the largest area available for benthic microalgal establishment. Total intertidal ($8.7 \pm 0.1 \text{ kg}$, average biomass = $76.3 \pm 8 \text{ mg m}^{-2}$) and subtidal ($57.7 \pm 0.4 \text{ kg}$, average biomass = $43.2 \pm 6.4 \text{ mg m}^{-2}$) benthic chlorophyll-a biomass values for the estuary reached their maximum at a flow of 1.16 and $1 \text{ m}^3 \text{s}^{-1}$ respectively. Using one way ANOVA on ranks, these maxima were significantly higher (intertidal $n = 162$, subtidal $n = 145$, $p < 0.0001$) than the benthic chlorophyll-a at all other flows. There were no trends relating benthic chlorophyll-a and the distance from the mouth of the estuary.

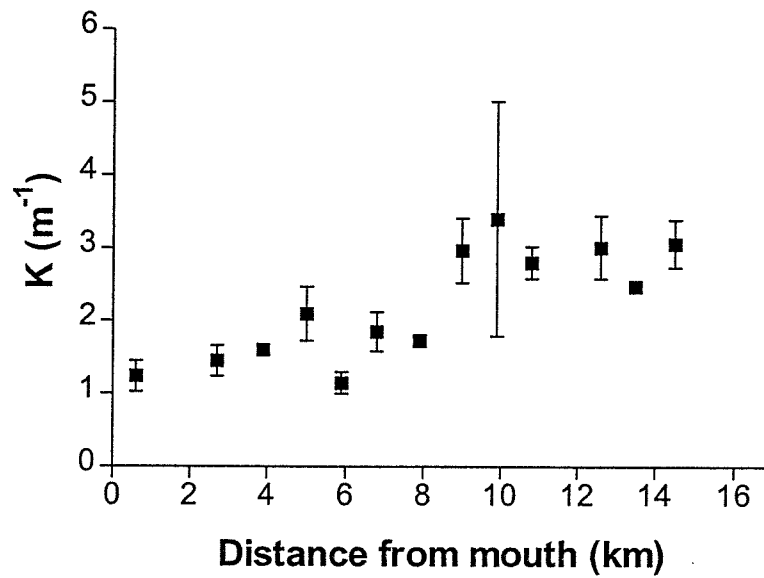


Figure 4.2.7: Average light attenuation (K) through the water column (m^{-1}) as a function of distance from the Gamtoos Estuary mouth ($n = 61$).

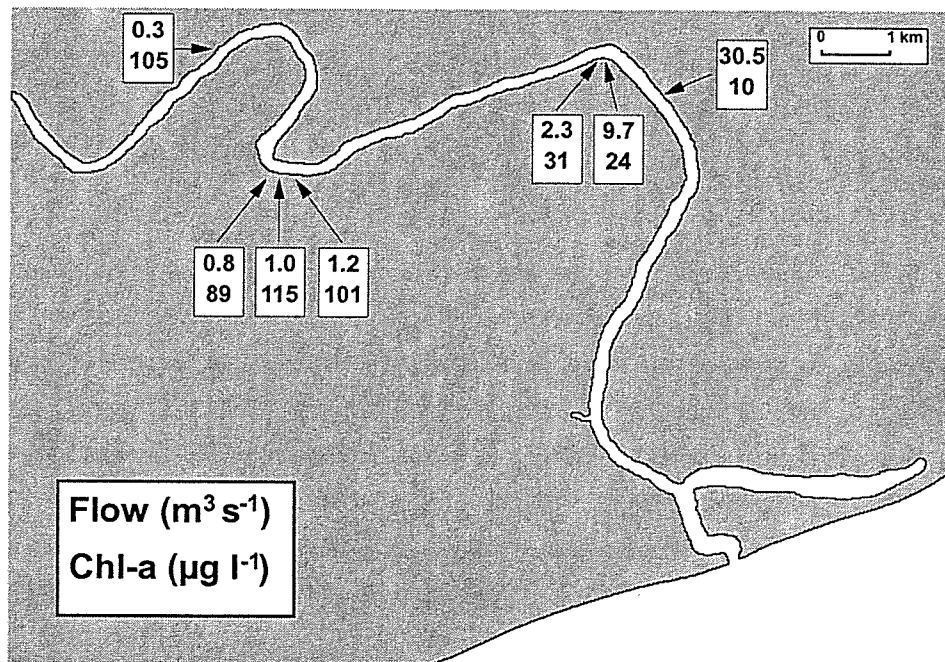


Figure 4.2.8: Positions of the river estuary interface along the length of the Gamtoos Estuary under different river flows. Estimated flow ($\text{m}^3 \text{s}^{-1}$) and vertically averaged chlorophyll-a ($\mu\text{g l}^{-1}$) are shown.

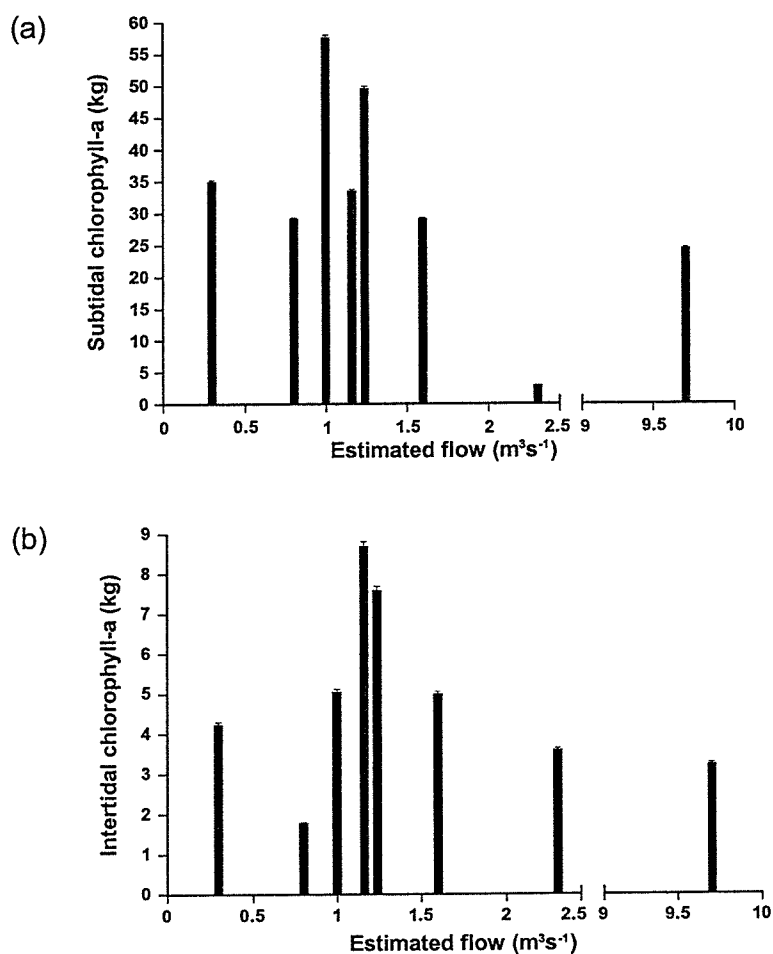


Figure 4.2.9: Total (a) subtidal and (b) intertidal chlorophyll-a biomass (mg m^{-2}) in response to river flow (0.3 to $9.7 \text{ m}^3 \text{ s}^{-1}$).

4.2.5 Discussion

The morphology of the Gamtoos Estuary is very similar to that of the nearby Sundays Estuary ($33^{\circ}43'S$, $25^{\circ}25'E$). Unlike estuaries with broad, exposed middle and lower reaches, very little wind mixing of the water column occurs within the channeled Gamtoos Estuary. Instead, the morphology combined with an average freshwater input of $1 \text{ m}^3 \text{ s}^{-1}$ has resulted in a highly stratified system. High levels of suspended sediment and nutrients are imported into the estuary as a result of the proportionally high runoff coming from a large catchment with little ground cover. Mineral nutrients become trapped in the fresh upper layer of stratified estuaries with very little mixing into the denser saline water (Ahel *et al.* 1996). High velocities of surface flow, pulses in particular, and increased turbulence are required to weaken the stratification and mix the nutrient concentration (Kennish 1992).

River water input is the most important factor controlling loading to the estuary. It also controls the transport of plant mineral nutrients from the upper estuary (where point-source and diffuse-source nutrient inputs are concentrated) to the lower estuary. Nutrient uptake by algal growth, as well as the marine 'dilution-effect', lead to a decrease in nutrient availability within the deeper and broader lower reaches of the estuary. Studies on the Choptank Estuary in the USA (Staver *et al.* 1996) showed that nitrate limitation was extended during periods of below average river discharge. Under low flow conditions, seepage from agricultural land adjacent to the upper reaches of the estuary, becomes the primary source of nitrate (Figure 4.2.4). Under these conditions nitrate is restricted to the shallow upper reaches of the estuary. Chlorophyll-a was high in this region (Figure 4.2.4), suggesting that nitrate was being taken up rapidly in this region. This resulted in there being no measurable nitrate in the middle and lower regions of the estuary.

Nutrient availability as well as the retention time of mineral-rich water control phytoplankton biomass. Nitrate was well distributed along the estuary at flows of 2.3, 9.7 and 30.5 m³ s⁻¹. However, if the total volume of the estuary is taken to be 3.6 x 10⁶ m³, the estuary will only be completely flushed in about 18 days at a flow rate of 2.3 m³ s⁻¹, and 1.4 days at a flow rate of 30.5 m³ s⁻¹. High flows will flush the low volume upper reaches much faster than the deeper and broader middle and lower reaches. Schumann and Pearce (1997) estimated that a flow of 10 m³ s⁻¹ will flush out the upper reaches of the estuary within a tidal period. This implies that the retention time of mineral nutrients suitable for phytoplankton growth increases further down the estuary. This partially explains the gradual shift of the REI down the estuary as flow increases.

Highest phytoplankton biomass occurred at flows of 0.8 and 1.2 m³ s⁻¹. These require at least 56 (> 4 spring tidal cycles) and 36 days (> 2.6 spring tidal cycles) to flush the estuary respectively. If retention time exceeded this, ~140 days (0.3 m³ s⁻¹), nitrate levels become depleted and restricted to the upper reaches. This limits elevated phytoplankton biomass to the upper reaches and total chlorophyll-a for the estuary is reduced. The importance of retention time on algal (chlorophyll-a) biomass has been demonstrated in a number of studies (e.g. Cromar and Fallowfield 1997; Eyre and Twigg 1997; Hilmer and Bate 1990). Phytoplankton chlorophyll-a ranged between 1 and 233 µg l⁻¹ in the Gamtoos Estuary. The maximum is double that of the nearby Sundays Estuary (Hilmer and Bate 1990) which is also influenced by agriculture and has a similar morphology. Maximum chlorophyll-a in the upper Swan Estuary, Australia (Thompson and Hosja 1996), Krka Estuary, Croatia (Ahel *et al.* 1996), San Antonio Bay, Texas (MacIntyre and Cullen 1996) and lower St. Lawrence Estuary, Canada (Roy *et al.* 1996) were all less than a quarter of that recorded in the Gamtoos. River input to the Gamtoos Estuary seldom drops below 0.5 m³ s⁻¹. Phytoplankton biomass is most probably governed by nutrient supply and herbivory mechanisms. Both processes are also important in controlling benthic microalgae, a major component of primary productivity in intertidal and shallow subtidal marine sediments (Duffy *et al.* 1997; Miller *et al.* 1996).

Although there were no trends of microphytobenthos biomass along the longitudinal axis of the estuary, riverwater input was an important controlling factor. Changes in physical factors such as sediment type and turbidity along the length of the estuary, compounded by the effect of nutrients, result in a weak correlation between benthic chlorophyll-a and nitrate. The peaks in benthic chlorophyll-a at flows of 1 (subtidal) and $1.2 \text{ m}^3 \text{ s}^{-1}$ (intertidal) can be attributed to nutrient availability and reduced sediment scouring. Under low flow conditions (0.3 and $0.8 \text{ m}^3 \text{ s}^{-1}$) nutrient concentrations in the water-column are low and not easily available because of a well-developed sediment-water boundary layer. As flow rate increases (1 and $1.2 \text{ m}^3 \text{ s}^{-1}$), nutrient levels increase and the boundary layer dimension becomes reduced, improving sediment-water mineral nutrient fluxes. However, under high flow conditions ($> 1.2 \text{ m}^3 \text{ s}^{-1}$) the biofilms produced by epipellic diatoms (Underwood *et al.* 1998) or the microalgal mats (Miller *et al.* 1996) which reduce sediment erodability are scoured away and a reduction in microphytobenthos biomass is the result.

Miller *et al.* (1996) listed the benthic chlorophyll-a in a number of estuaries from North America, Europe, Australia and Asia. Values ranged between < 1 and 560 mg m^{-2} , and the highest of these were collected from sheltered, muddy habitats. Benthic chlorophyll-a in the Gamtoos was quite low in relation to the international range, being between 2 and 118 mg m^{-2} . The channel-like morphology and the high turbidity of the estuary are the most likely factors limiting microphytobenthos growth. However, values were always lowest in the exposed, sandy mouth regions of the estuary and highest in the muddy middle and upper reaches, thus confirming the trend found in other estuaries.

4.2.6 Conclusions

Average phytoplankton biomass was highest ($> 47.5 \text{ } \mu\text{g l}^{-1}$) at a flow of approximately $1 \text{ m}^3 \text{ s}^{-1}$. Subtidal and intertidal benthic chlorophyll-a values were also highest at this flow, being $57.7 (\pm 0.4) \text{ kg}$ and $8.7 (\pm 0.1) \text{ kg}$ respectively. These results suggest that a river discharge of $1 \text{ m}^3 \text{ s}^{-1}$ is optimal for maximum phytoplankton and benthic microalgal growth. Nutrient concentration combined with estuarine flushing time appeared to be the most important factors controlling microalgal biomass. At a flow $< 1 \text{ m}^3 \text{ s}^{-1}$, measurable nitrate was limited to the upper reaches and average nitrate in the estuary was relatively low ($< 35 \text{ } \mu\text{M}$). At a flow $> 1 \text{ m}^3 \text{ s}^{-1}$, flushing time of the estuary was less than three spring tidal cycles and even faster in the low volume upper estuary. Three tidal cycles was the optimum flushing time in the nearby Sundays Estuary which is similar in morphology to the Gamtoos Estuary and is influenced by agricultural seepage (Hilmer 1990).

Vertically averaged phytoplankton chlorophyll-a was highest at a flow of $1 \text{ m}^3 \text{ s}^{-1}$. Nutrient input from agricultural seepage supported a high phytoplankton biomass in the upper reaches of the estuary, particularly during low flow periods. As a result, the effects of reduced flow did not result in a large phytoplankton biomass decrease in the REI. Flushing time, which is related to freshwater input, had the greatest effect on phytoplankton biomass in the REI. As flow increased above $1 \text{ m}^3 \text{ s}^{-1}$, biomass decreased and shifted towards the middle reaches of the estuary.

4.3 River flow rate and the distribution of phytoplankton groups in the Gamtoos Estuary

4.3.1 Introduction

Estuaries are considered to be the strategic link between the continents and oceans. The biotic structure of an estuary is the result of dynamic interactions between the involved populations and their abiotic environment (Köhler 1997). Sharp gradients in physico-chemical properties occurring in estuaries, particularly at the river estuary interface region (REI), create dynamic biological environments. The REI is hypothesized, in this study, to be the region where phytoplankton cells are present in highest abundance at a vertically averaged salinity of $< 10\text{‰}$. In addition, phytoplankton group structure within the REI is distinguishable from the rest of the estuary. Kinne (1971 cited in Ahel *et al.* 1996) noted that both freshwater and marine organisms have been shown to respond strongly to salinity gradients and a boundary at 4 to 7 ‰ was identified as a critical region of physiological stress for freshwater species.

The Gamtoos Estuary is a stratified and permanently open estuary. Large dams have reduced the normal river input into the estuary to $< 1 \text{ m}^3 \text{ s}^{-1}$ (Schumann and Pearce 1997). Stratified estuaries represent unique ecosystems that are most suitable for studying phytoplankton responses to environmental changes, due to the rather broad range of key physico-chemical properties occurring within a relatively small geographic area (Ahel *et al.* 1996). By examining the distribution of phytoplankton groups, it is possible to investigate the effect of salinity gradients, as well as different mineral element inputs (particularly nitrate), on phytoplankton dynamics in a permanently open and stratified estuarine environment.

The Gamtoos Estuary is a eutrophic system with relatively high levels of mineral elements being imported from agriculture within the Gamtoos River's catchment. As a result, the estuary supports a high biomass of phytoplankton ($\sim 50 \mu\text{g l}^{-1}$). So the question that arises is whether a high phytoplankton biomass indicates that an estuary is functioning optimally. In the case of eutrophic systems where nutrient enrichment is artificial, an elevated phytoplankton biomass could be interpreted as a negative effect; i.e. the system's function has shifted from what is normal. In severe cases, eutrophication can lead to blooms of planktonic organisms and filamentous algae (Boney 1989). Perhaps a more accurate measure of a system's function would be to determine phytoplankton biodiversity.

Evidence, primarily from oligotrophic estuaries in which phytoplankton growth and nutrient recycling are highly coupled, suggests that cells can be near their maximum growth rates and are, therefore, not nutrient limited even at low ambient conditions (Thompson and Hosja 1996). As a result of the high concentrations of mineral elements being introduced into the Gamtoos Estuary from agricultural seepage, mineral elements should not be limiting to microalgal growth, particularly at water flow rates $\geq 1 \text{ m}^3 \text{ s}^{-1}$.

The objective of this study was to determine, using microscopic identification of phytoplankton groups, the effect of reduced river flow into the Gamtoos Estuary on the phytoplankton community structure. A number of hypotheses were devised at the start of this study. The following hypothesis, based on findings during a similar study by Hilmer (1990), was related to phytoplankton group dynamics. The hypothesis states that riverwater entering the estuary is flagellate-dominated. Dinoflagellates are dominant towards the middle reaches if there is strong salinity stratification, but if the water is well mixed, diatoms become the dominant group in the middle reaches.

4.3.2 Materials and methods

Phytoplankton cells were collected using a weighted 0.5 l pop-bottle. Samples were taken at the surface, 0.5, 1 and then at 1 m intervals to the bottom from sites ranging from the mouth to near the head of the estuary. The water samples were fixed with 1.5 ml of 1% (v/v) glutaraldehyde solution (Underwood *et al.* 1998). Glutaraldehyde was preferred to a 10 % neutral formalin solution as formalin can cause flagellates to lose their flagella, thus making identification difficult (Lund *et al.* 1958; Boney 1989). Once preserved, 60 ml of the sample was stirred before being poured into a 26.5 mm internal diameter settling chamber (Figure 4.3.1) (Utermöhl 1958). Cells were left to settle for at least 24 hours before being identified using a Zeiss IM 35 inverted microscope at the maximum magnification of x 630. A minimum of 200 cells was counted for each sample. The number of diatoms, flagellates, dinoflagellates, chlorophytes, cyanophytes and coccolithophorids in 1 ml of estuarine sample was calculated using the formula:

$$N = ((r^2) / A) \times C / V$$

Where: N = Average number of cells ml⁻¹ of each phytoplankton group in a sample; A = area of each frame (mm²); C = number of cells in each frame; V = volume of settling chamber (ml).

The Pearson product-moment correlation was used to determine the strength of association between phytoplankton groups and environmental variables. The Shannon index (H') was used to determine the diversity of the six phytoplankton groups.

The number of phytoplankton cells was correlated to environmental variables using the spreadsheet program Lotus 1-2-3 and the statistical program Stat100. Sampling sites were selected during each sampling session depending on the longitudinal salinity. As a result, the exact sites were not routinely sampled. Hence, to avoid any bias as a result of an uneven spread of site, salinity, cell numbers and nitrate between sampled sites were averaged. Similarly, data were averaged at 1.5 and 2.5 m because samples were taken at the surface, 0.5, 1 and then at 1 m intervals (depths > 3 m were excluded) to the bottom.

Multivariate analysis (DCA and PCA) was used to determine whether there were distinct biotically similar sites for each flow rate. Focus was placed on sites and phytoplankton groups and environmental variables were superimposed onto the ordination diagrams. However, the resulting ordination plots displayed no significant trend so results will only address phytoplankton group and

environmental data. Phytoplankton group dynamics represent changes in a suite of populations. The classification is very broad and this could have led to the failure of multivariate analysis techniques to highlight similar sites.

4.3.3 Results

Schumann and Pearce (1997) estimated that the average annual inflow of riverwater into the Gamtoos Estuary was $< 1 \text{ m}^3 \text{ s}^{-1}$. For the purpose of this study, $1 \text{ m}^3 \text{ s}^{-1}$ will be regarded as the average flow and other flow rates will be referred to as either low or high. A phytoplankton group is considered to be dominant if its abundance contributes more than 10 % to the total phytoplankton.

4.3.3.1 Salinity and nitrate

As river flow entering the Gamtoos Estuary increased following rain in the catchment, the average salinity in the estuary decreased (Figure 4.2.3). At flow rates in excess of $1 \text{ m}^3 \text{ s}^{-1}$ the salinity at the mouth of the estuary became lower than seawater. This resulted in the longitudinal salinity gradient (difference in vertically averaged salinity between the head and mouth of the estuary) becoming weaker. At the highest flow recorded ($30.5 \text{ m}^3 \text{ s}^{-1}$) the longitudinal salinity gradient had become reduced to 15.6‰. Nitrate concentration increased towards the head of the estuary suggesting that riverwater was the source of the mineral nutrient. There was an inversely proportional relationship between nitrate and salinity in the estuary, i.e. as salinity increased towards the mouth of the estuary, nitrate decreased. At flow rates of 0.3 to $1.2 \text{ m}^3 \text{ s}^{-1}$, no measurable nitrate was present near to the mouth of the estuary. The average nitrate in the estuary at the lowest flow ($0.3 \text{ m}^3 \text{ s}^{-1}$) was $1.8 \pm 3.4 \text{ } \mu\text{M}$ increasing to $89.2 \pm 34.6 \text{ } \mu\text{M}$ when the flow rate was $30.5 \text{ m}^3 \text{ s}^{-1}$.

4.3.3.2 Phytoplankton group structure

4.3.3.2.1 Average flow: $1 \text{ m}^3 \text{ s}^{-1}$

Average phytoplankton abundance was $11748 \pm 1629 \text{ cells ml}^{-1}$ (Table 4.3.1). Total phytoplankton abundance decreased towards the mouth of the estuary and was negatively correlated (Table 4.3.2) with the increase in salinity towards the mouth (Figure 4.3.2A). The maximum vertical average of $15854 \pm 1668 \text{ cells ml}^{-1}$ was between sites 7 and 8 where vertically averaged salinity was $6.1 \pm 1.4\text{‰}$. The maximum phytoplankton chlorophyll-a zone was located at site 7 at a flow of $1 \text{ m}^3 \text{ s}^{-1}$. As the vertically averaged salinity increased to $35.7 \pm 0.5 \text{ ‰}$ near to the mouth of the estuary, the concentration of cells decreased to a vertically averaged minimum of $4241 \pm 504 \text{ cells ml}^{-1}$. Total phytoplankton abundance decreased rapidly at a salinity value greater than 30‰. Flagellates (43%), diatoms (38%) and chlorophytes (14%) dominated the phytoplankton. Phytoplankton group diversity was highest 10 to 14 km from the mouth of the estuary. The phytoplankton group diversity (H') was 0.50 (Table 4.3.1). The maximum possible diversity (H'_{max}) of the five phytoplankton groups was 0.70.

Phytoplankton cell numbers increased significantly (Table 4.3.2) as water column nitrate concentration increased and salinity decreased from the mouth to the estuary head (Figure 4.3.2B). There was a significant difference ($t = 277$, $P < 0.0001$) between the number of phytoplankton cells where the ambient nitrate concentration was $< 1.7 \mu\text{M}$ and those where it was greater.

Longitudinally averaged flagellate numbers were most abundant at the surface. Vertically averaged flagellate numbers were highest in the middle and upper reaches of the estuary. Their numbers decreased from $7395 \pm 1055 \text{ cells ml}^{-1}$ at site 9, where the average salinity was low (0.5 ‰), to $1864 \pm 274 \text{ cells ml}^{-1}$ at site 1. Flagellate numbers decreased sharply near the mouth of the estuary where salinity was $> 30 \text{ ‰}$ (Figure 4.3.2C). This was the region where nitrate was lowest. The number of flagellates were significantly lower ($T = 261$, $P < 0.0001$) at an ambient nitrate concentration of $< 1.7 \mu\text{M}$ (Figure 4.3.2D) and increased significantly as nitrate increased (Table 4.3.2).

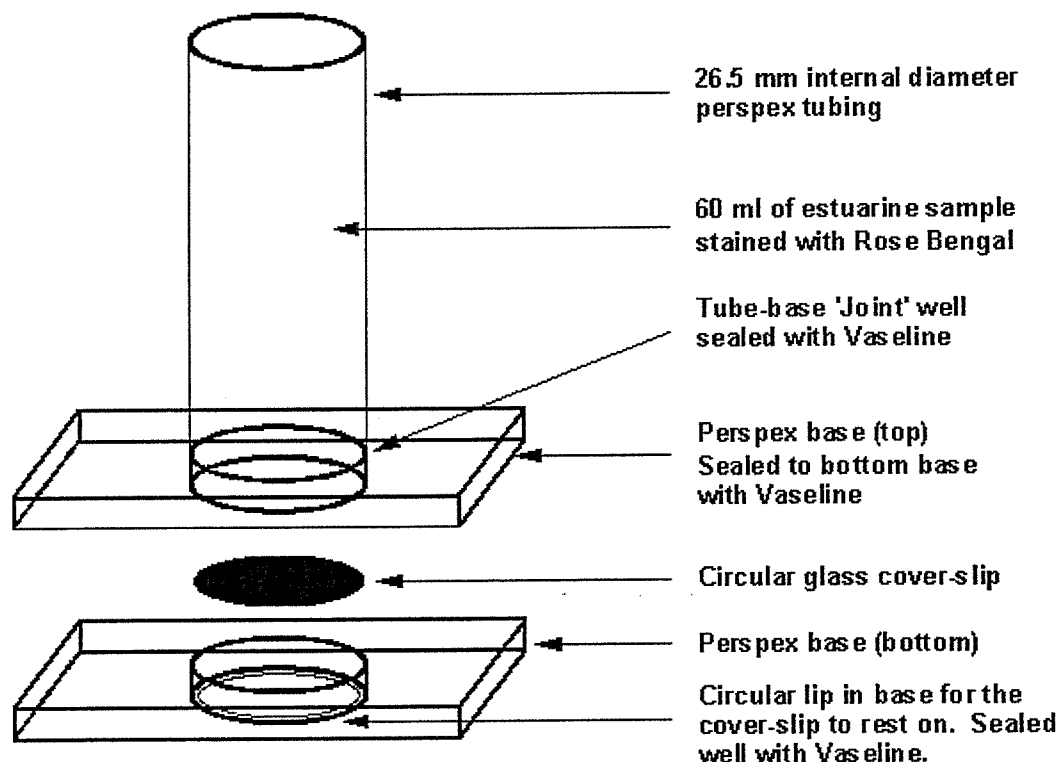


Figure 4.3.1: A diagrammatic representation of a 26.5 mm internal diameter settling chamber used for phytoplankton counts and identification (Utermöhl 1958).

Longitudinally averaged diatom numbers were most abundant 0.5 m from the surface. Numbers were highest in the middle and upper reaches of the estuary, decreasing from a maximum of 6633 ± 532 cells ml^{-1} at site 5, to a minimum of 2158 ± 279 cells ml^{-1} near to the mouth of the estuary. Diatom numbers decreased towards the mouth of the estuary where salinity was greater than 30 ‰ (Figure 4.3.2E) and where nitrate was low (Figure 4.3.2F). The number of diatoms was significantly lower ($t = -3.83$, $P = 0.0003$) at an ambient nitrate concentration of $< 1.7 \mu\text{M}$. There was a significant but weak correlation between diatoms and salinity (Table 4.3.2). There was no significant relationship between diatom numbers and nitrate along the length of the estuary.

Longitudinally averaged chlorophyte cell numbers were greatest 0.5 m from the surface. Numbers were only in excess of 1000 cells ml^{-1} in the middle and upper reaches of the estuary (sites 5 to 9). The maximum vertically averaged number of chlorophytes, 4998 ± 1260 cells ml^{-1} , was between sites 7 and 8. Chlorophyte numbers decreased rapidly and significantly (Table 4.3.2) as salinity exceeded 28 ‰ (Figure 4.3.2G). Chlorophyte numbers decreased with a decrease in nitrate (Table 4.3.2), and were lowest (20 ± 6 cells ml^{-1}) at the mouth of the estuary where nitrate was $< 1.7 \mu\text{M}$ (Figure 4.3.2H). Maximum vertically averaged phytoplankton chlorophyll-a, measured at site 7, was measured in the same vicinity as the maximum phytoplankton cell and chlorophyte cell abundance.

The most dominant group, the coccolithophorids, bloomed and exceeded 10000 cells ml^{-1} at a salinity ranging between 18 and 34 ‰ (Figure 4.3.3C). There was no significant trend between the number of coccolithophorid cells and salinity (Table 4.3.2). However, their numbers increased significantly (Table 4.3.2) as nitrate decreased (Figure 4.3.3D). The inverse correlation was weak as a result of a sharp decrease in cell numbers towards the mouth of the estuary where no measurable nitrate was recorded. The highest vertically averaged number of coccolithophorids was measured at site 6.

Longitudinally averaged flagellate numbers were most abundant at the surface. Vertically averaged flagellate numbers were highest in the upper reaches of the estuary. Numbers decreased from 7518 ± 49 cells ml^{-1} between sites 8 and 9, where the average salinity was low (< 6 ‰), to 1689 ± 124 cells ml^{-1} at site 1. Flagellates decreased significantly with an increase in salinity (Table 4.3.2). The greatest decrease in flagellates occurred as salinity exceeded 32 ‰ (Figure 4.3.3E). Nitrate levels were very low ($< 1 \mu\text{M}$) in the lower reaches of the estuary where salinity was > 28 ‰. Flagellate numbers were strongly and positively correlated (Table 4.3.2) to nitrate concentration in the estuary (Figure 4.3.3F). However, this may be correlative rather than cause and effect because the nitrate was brought in by the freshwater.

Table 4.3.1: Distance of the vertically averaged 10 % and saline water (≥ 1 %) penetration from the estuary mouth (km), average nitrate $\pm$ standard error (SE), average phytoplankton abundance (cells ml^{-1}) and Shannon phytoplankton group diversity indices (H' and H'_{max}), in relation to river discharge ($\text{m}^3 \text{s}^{-1}$).

Flow ($\text{m}^3 \text{s}^{-1}$)	0.3	0.8	1	1.2	2.3	9.7	30.5
Position of 10 % & < 1%	13 & 16.5	14 & 17.5	11 & 16.5	13.5 & 14.5	8.5 & >10.8	9.5 & 12.6	5.5 & 7.9
Nitrate Avg. $\pm$ SE (μM)	1.8 $\pm$ 0.4	14.8 $\pm$ 3.3	39.3 $\pm$ 5.5	48.4 $\pm$ 6.9	55.4 $\pm$ 7	Not measured	89.2 $\pm$ 4.7
Diatoms Avg. $\pm$ SE (cells ml^{-1})	2679 $\pm$ 188	4376 $\pm$ 203	4510 $\pm$ 620	7199 $\pm$ 2387	1128 $\pm$ 345	4776 $\pm$ 732	1857 $\pm$ 129
Flagellates Avg. $\pm$ SE (cells ml^{-1})	8497 $\pm$ 773	4050 $\pm$ 240	5112 $\pm$ 715	3250 $\pm$ 698	2859 $\pm$ 233	1788 $\pm$ 204	1700 $\pm$ 142
Dinoflagellates Avg. $\pm$ SE (cells ml^{-1})	1937 $\pm$ 257	46 $\pm$ 14	108 $\pm$ 51	248 $\pm$ 102	33 $\pm$ 16	40 $\pm$ 15	13 $\pm$ 5
Chlorophytes Avg. $\pm$ SE (cells ml^{-1})	393 $\pm$ 70	1607 $\pm$ 120	1639 $\pm$ 501	152 $\pm$ 58	61 $\pm$ 31	126 $\pm$ 52	99 $\pm$ 32
Cyanophytes Avg. $\pm$ SE (cells ml^{-1})	457 $\pm$ 22	134 $\pm$ 36	378 $\pm$ 111	1016 $\pm$ 314	64 $\pm$ 61	156 $\pm$ 50	23 $\pm$ 11
Coccolithophorids Avg. $\pm$ SE (cells ml^{-1})	None recorded	10744 $\pm$ 1504	None recorded	None recorded	None recorded	None recorded	None recorded
Total cells Avg. $\pm$ SE (cells ml^{-1})	13963 $\pm$ 944	20958 $\pm$ 1750	11748 $\pm$ 1629	11864 $\pm$ 3026	4145 $\pm$ 465	6887 $\pm$ 764	3618 $\pm$ 221
Diversity H'/H'_{max}	0.48/0.70	0.53/0.78	0.50/0.70	0.43/0.70	0.34/0.70	0.34/0.70	0.37/0.70

Table 4.3.2: Pearson correlation coefficients for multiple correlations of phytoplankton group abundance, salinity and nitrate at different river inputs. There were insufficient nitrate data for correlations at flows of $9.7 \text{ m}^3 \text{ s}^{-1}$ and $30.5 \text{ m}^3 \text{ s}^{-1}$. Data include longitudinal and vertical measurements, and 'n' is the number of observations.

	Diatoms	Flag's	Dino's	Chloro's	Cyano's	Cocco's	Total cells	n
$0.3 \text{ m}^3 \text{ s}^{-1}$								
Salinity	0.206	-0.717***	-0.509**	-0.699***	-0.569**	None recorded	-0.845***	83
Nitrate	0.058	0.454***	-0.042	0.415***	0.075		0.426***	
$0.8 \text{ m}^3 \text{ s}^{-1}$								
Salinity	-0.058***	-0.825***	-0.298**	-0.735***	-0.433**	0.0781	-0.395***	87
Nitrate	0.275**	0.701***	0.254	0.701***	0.288**	-0.331**	0.107	
$1 \text{ m}^3 \text{ s}^{-1}$								
Salinity	-0.298**	-0.609***	0.165	-0.569***	-0.335**	None recorded	-0.689***	81
Nitrate	0.149	0.476***	-0.252*	0.496***	0.242*		0.527***	
$1.2 \text{ m}^3 \text{ s}^{-1}$								
Salinity	-0.751***	-0.777***	-0.864**	-0.697***	-0.499**	None recorded	-0.830***	53
Nitrate	0.713***	0.758***	0.861***	0.758***	0.554***		0.804***	
$2.3 \text{ m}^3 \text{ s}^{-1}$								
Salinity	-0.817***	-0.494***	0.288*	0.188	0.179	None recorded	-0.783***	51
Nitrate	0.861***	0.500***	0.233	0.186	-0.192		0.818***	
$9.7 \text{ m}^3 \text{ s}^{-1}$								
Salinity	0.044	-0.221***	0.406***	-0.253*	-0.105	None recorded	-0.034	51
Nitrate	N/A	N/A	N/A	N/A	N/A		N/A	
$30.5 \text{ m}^3 \text{ s}^{-1}$								
Salinity	0.321*	0.333*	0.42	-0.293*	0.034	None recorded	0.336*	45
Nitrate	N/A	N/A	N/A	N/A	N/A		N/A	

*p < 0.05, **p < 0.01 and ***p < 0.001

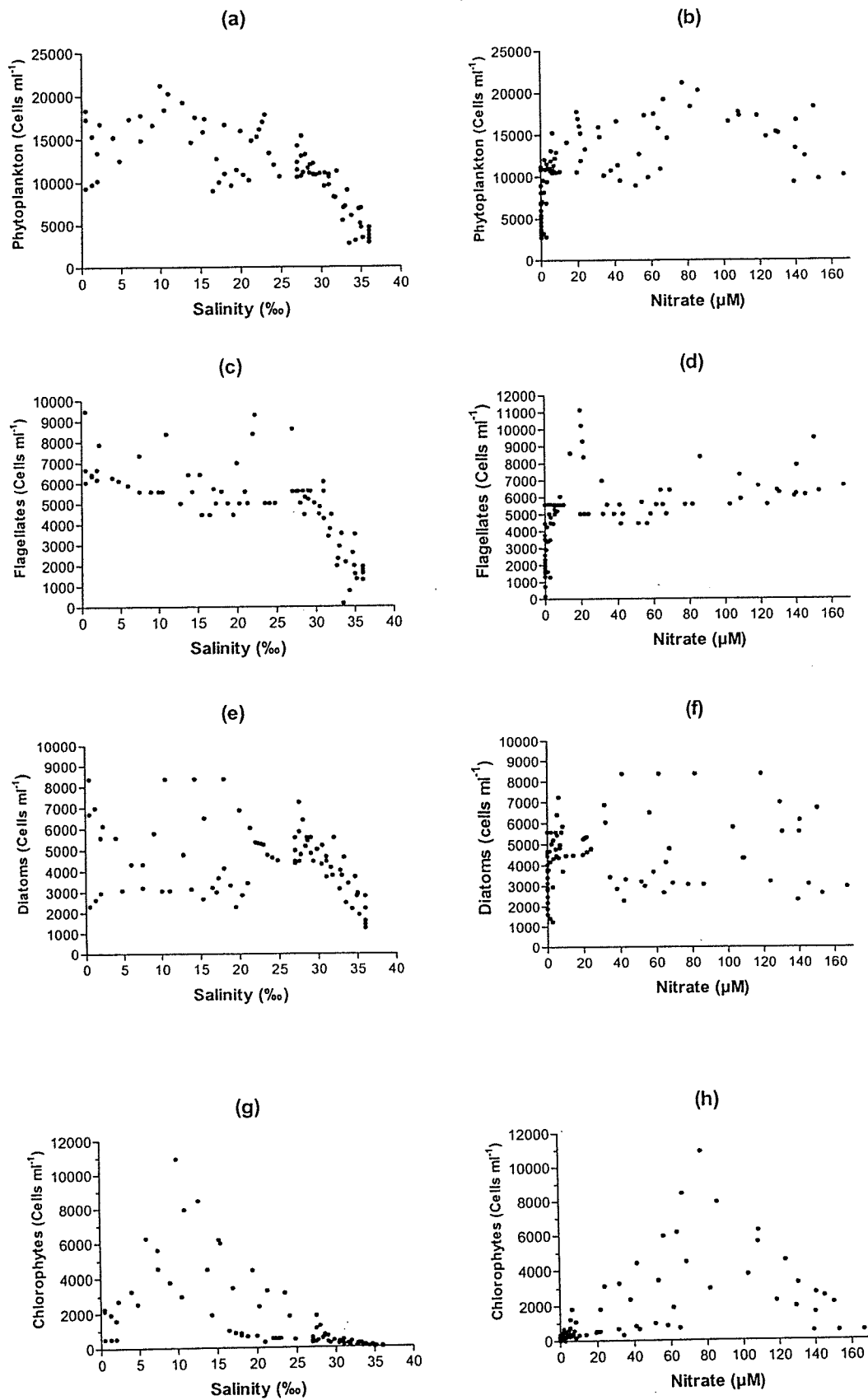


Figure 4.3.2: Total phytoplankton (a and b), flagellate (c and d), diatom (e and f) and chlorophyte (g and h) abundance in relation to salinity and nitrate respectively (flow = 1 m³ s⁻¹).

Longitudinally averaged diatom numbers were most abundant within the top 0.5 m of the water column. Numbers were highest in the middle reaches of the estuary, decreasing from a maximum of 5128 ± 71 cells l^{-1} between sites 6 and 7, to a minimum of 3710 ± 221 cells l^{-1} at site 4. Diatom numbers decreased slightly to < 4000 cells l^{-1} in the lower reaches of the estuary where salinity was > 32 ‰ (Figure 4.3.3G) and where nitrate was low (< 1 μM) (Figure 4.3.3H). The number of diatoms increased significantly with nitrate but decreased with an increase in salinity (Table 4.3.2).

The highest vertically averaged chlorophyll-a was recorded at site 7. Highest numbers in phytoplankton cells ranged between sites 6 and 8, with no distinct peak from any of the phytoplankton groups at site 7.

4.3.3.2.2 Flow: $0.3 m^3 s^{-1}$

The average number of phytoplankton cells at the lowest water flow recorded during the study was 13963 ± 944 cells ml^{-1} . Flagellates (61 %) were the most dominant group at this flow. Diatoms (19 %) and dinoflagellates (14 %) were the only other major groups. Average phytoplankton group diversity (H') decreased to 0.48 out of a maximum of 0.70 (Table 4.3.1). Phytoplankton numbers increased significantly as salinity decreased (Table 4.3.2). The lowest number of phytoplankton cells (< 10000 cells ml^{-1}) was recorded at salinities in excess of 27 ‰ (Figure 4.3.4A). Phytoplankton abundance was significantly higher (Table 4.3.2) in higher nitrate concentration found towards the head of the estuary (Figure 4.3.4B). The highest vertical averages of phytoplankton, 23227 ± 1077 and 23782 ± 4307 cells ml^{-1} , were recorded at sites 7 and 9 respectively.

Longitudinally averaged flagellate numbers were most abundant at the surface. Vertically averaged flagellate numbers were highest in the middle and upper reaches (> 10000 cells ml^{-1} between sites 5 and 8) of the estuary. Their numbers decreased from 17837 ± 1115 cells ml^{-1} at site 7, where vertically averaged salinity was 13.5 ± 0.6 ‰, to a minimum of 1310 ± 105 cells ml^{-1} at site 1 (Figure 4.3.4C). Nitrate levels were low throughout the estuary, with the only measurable nitrate being sampled between sites 6 and 9. The maximum vertically averaged nitrate concentration was 11.4 ± 2 μM at site 8, just downstream of the agricultural drainage pipe. Flagellate numbers were significantly correlated to nitrate concentration (Table 4.3.2), increasing as nitrate increased (Figure 4.3.4D).

Using Kruskal-Wallis One Way Analysis of Variance on Ranks, no significant difference ($P = 0.47$, $n = 83$) could be found between longitudinally averaged diatom numbers. Their abundance was well mixed throughout the water column. Vertically averaged diatom numbers remained relatively consistent throughout the estuary. The only significant difference between sites, using All Pairwise Multiple Comparison Procedures (Dunn's Method), was between sites 2 and 7 ($P < 0.05$). Diatom cells were well distributed at this flow, and showed no significant correlations between cell abundance and nitrate or salinity (Table 4.3.2).

Longitudinally averaged dinoflagellate numbers were highest at 0.5 m depth. Vertically averaged numbers decreased, from a maximum of 3827 ± 353 cells ml^{-1} at site 9, to 164 ± 14 cells ml^{-1} at the mouth of the estuary. Dinoflagellate numbers were significantly higher (Table 4.3.2) in low

salinity water at the head of the estuary (Figure 4.3.4G). There was no significant correlation (Table 4.3.2) between dinoflagellate abundance and nitrate concentration (Figure 4.3.4H).

Vertically averaged phytoplankton chlorophyll-a was highest at site 9. This was the site where total phytoplankton and dinoflagellates were highest. However, there was a distinct peak in chlorophyll-a at site 7, where the highest number of flagellate cells was recorded.

4.3.3.2.3 Flow: $1.2 \text{ m}^3 \text{ s}^{-1}$

River flow into the estuary was the same on the 25 October 1997 and 8 November 1997 so salinity, nitrate and phytoplankton cell numbers were averaged. There was a linear decrease in total phytoplankton abundance (Table 4.3.2) as salinity increased (Figure 4.3.5A). The maximum vertical average of phytoplankton, $20244 \pm 6318 \text{ cells ml}^{-1}$, was sampled between sites 5 and 6 where vertically averaged salinity was $< 15 \text{ ‰}$. The mouth, where vertically averaged salinity was $30.5 \pm 2.5 \text{ ‰}$, was the site of the lowest vertically averaged phytoplankton numbers ($5741 \pm 461 \text{ cells ml}^{-1}$). Phytoplankton abundance was lowest where salinity was $> 25 \text{ ‰}$. Diatoms (61 %) and flagellates (27 %) were the dominant phytoplankton groups. Average phytoplankton group diversity (H') was 0.43 out of a possible maximum of 0.70 (Table 4.3.1).

The relationship between phytoplankton and nitrate was the opposite of that with salinity (Figure 4.3.5B). There was a significant linear increase in phytoplankton numbers (Table 4.3.2) towards the head of the estuary where vertically averaged nitrate was the highest ($138 \pm 26.1 \text{ } \mu\text{M}$). Phytoplankton cells were lowest in areas of the estuary, particularly near the mouth, where nitrate was $< 25 \text{ } \mu\text{M}$.

Longitudinally averaged diatom numbers were most abundant at the surface. Vertically averaged diatom numbers were highest in the middle and upper reaches of the estuary, decreasing from a maximum of $13525 \pm 4938 \text{ cells ml}^{-1}$ between sites 5 and 6, to a minimum of $3265 \pm 241 \text{ cells ml}^{-1}$ near to the mouth of the estuary. Diatom numbers were significantly lower (Table 4.3.2) in low salinity water (Figure 4.3.5C) and increased significantly where nitrate concentration was highest, towards the head of the estuary (Figure 4.3.5D). Diatom numbers were lowest in water with a nitrate concentration of $< 25 \text{ } \mu\text{M}$.

The trend in flagellate numbers was similar to diatoms. Longitudinally averaged flagellate numbers were most abundant at the surface. Vertically averaged flagellate numbers were highest in the middle and upper reaches of the estuary. Their numbers decreased from $5176 \pm 1491 \text{ cells ml}^{-1}$ between sites 5 and 6, to a minimum of $1945 \pm 350 \text{ cells ml}^{-1}$ at site 1 (Figure 4.3.5E). Flagellate numbers were significantly correlated (Table 4.3.2) to nitrate concentration in the estuary (Figure 4.3.5F). The lowest number of flagellates were recorded where the nitrate was $< 25 \text{ } \mu\text{M}$.

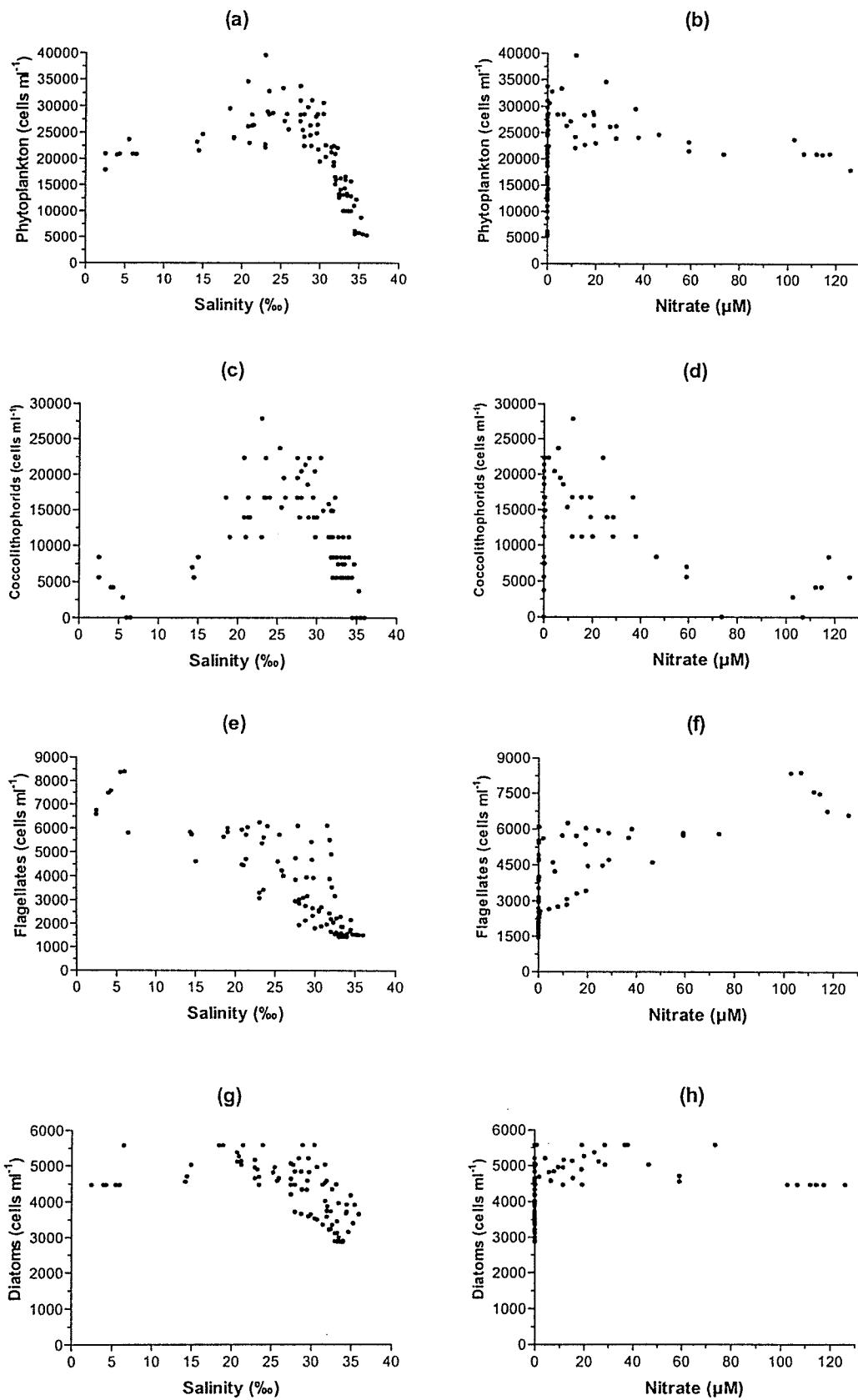


Figure 4.3.3: Total phytoplankton (a and b), coccolithophorid (c and d), flagellate (e and f) and diatom abundance (g and h) in relation to salinity and nitrate respectively (flow = $0.8 \text{ m}^3 \text{ s}^{-1}$).

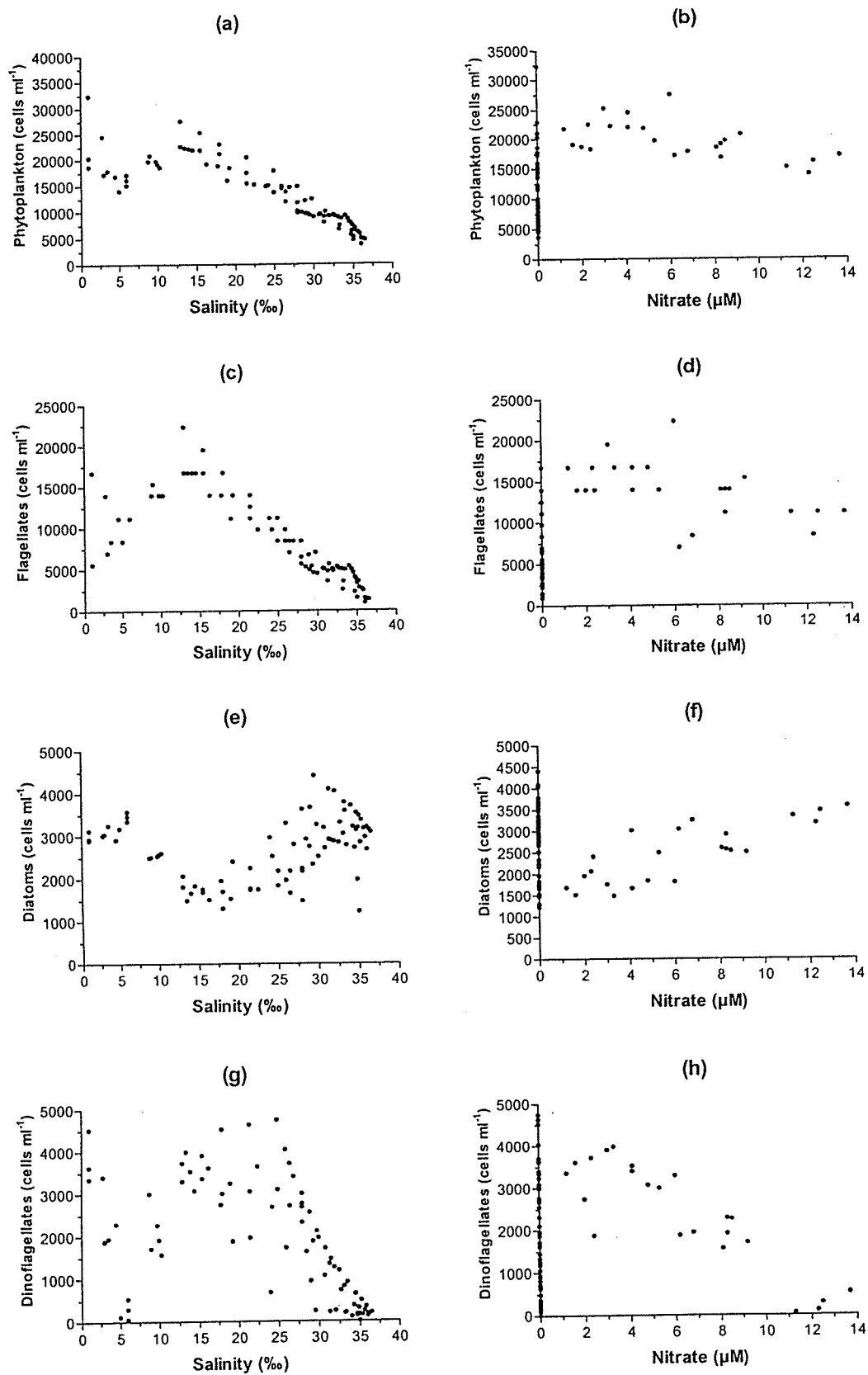


Figure 4.3.4: Total phytoplankton (a and b), flagellate (c and d), diatom (e and f) and dinoflagellate (g and h) abundance in relation to salinity and nitrate respectively (flow = $0.3 \text{ m}^3 \text{ s}^{-1}$).

Phytoplankton chlorophyll-a was highest at sites 7 and 8 for the two sampling sessions. Vertical averages of diatoms, flagellates and total phytoplankton were all between sites 5 and 6, so no single phytoplankton group is likely to have resulted in the peak in chlorophyll-a.

4.3.3.2.4 Flow: $2.3 \text{ m}^3 \text{ s}^{-1}$

Total phytoplankton abundance decreased significantly (Table 4.3.2) towards the mouth of the Gamtoos Estuary (Figure 4.3.6A) where salinity was highest and nitrate lowest (Figure 4.3.6B). The maximum vertical average of phytoplankton, $6461 \pm 368 \text{ cells ml}^{-1}$, was sampled at the site 6. Site 1, where vertically averaged salinity was $29.2 \pm 0.9 \text{ ‰}$, was the site of the lowest vertically averaged phytoplankton ($2382 \pm 392 \text{ cells ml}^{-1}$). Phytoplankton cell numbers were only $> 5000 \text{ cells ml}^{-1}$ at a vertically averaged salinity of 5.5 ‰ . Diatoms (27 %) and flagellates (69 %) were the dominant phytoplankton groups but the average number of phytoplankton in the estuary was lower ($< 4200 \text{ cells ml}^{-1}$) than was counted during lower water flow rates. Average phytoplankton group diversity was lower than recorded at lower flow rates ($H' = 0.34$) (Table 4.3.1).

Longitudinally averaged flagellate numbers were most abundant at 0.5 m. Numbers decreased significantly as salinity increased (Table 4.3.2) from a maximum of $4557 \pm 163 \text{ cells ml}^{-1}$ between sites 4 and 5 (vertically averaged salinity was $10.2 \pm 6.2 \text{ ‰}$), to a minimum of $1778 \pm 86 \text{ cells ml}^{-1}$ at site 1 (Figure 4.3.6C) where the salinity was $29 \pm 0.9 \text{ ‰}$. Flagellate numbers were significantly correlated (Table 4.3.2) to nitrate concentration in the estuary (Figure 4.3.6D). Flagellate numbers only exceeded $4000 \text{ cells ml}^{-1}$ when the nitrate concentration was $> 50 \text{ } \mu\text{M}$.

Vertically averaged diatom numbers were most abundant at the surface. Vertically averaged diatom numbers were highest at site 6 ($3239 \pm 765 \text{ cells ml}^{-1}$) but lowest ($170 \pm 37 \text{ cells ml}^{-1}$) near to the mouth of the estuary. Diatom numbers decreased (Table 4.3.2) as salinity increased (Figure 4.3.6E) and were significantly highest at the head of the estuary, where nitrate concentration was greatest (Figure 4.3.6F). Diatom numbers were highest ($> 2000 \text{ cells ml}^{-1}$) where nitrate was $> 100 \text{ } \mu\text{M}$. Vertically averaged phytoplankton chlorophyll-a was highest at site 5. Phytoplankton numbers were high in this region of the estuary but no particular group peaked in abundance at this particular site.

4.3.3.2.5 Flow $9.7 \text{ m}^3 \text{ s}^{-1}$

Nitrate was not measured during sampling at this flow rate. However it is noteworthy that 17 days later (21 November 1996), the Gamtoos Estuary flooded ($> 2000 \text{ m}^3 \text{ s}^{-1}$) resulting in the area being declared a disaster area. At a flow of $9.7 \text{ m}^3 \text{ s}^{-1}$, the average number of phytoplankton cells in the estuary was $6887 \pm 764 \text{ cells ml}^{-1}$. Diatoms (69 %) and flagellates (26 %) dominated the phytoplankton. Average phytoplankton group diversity was 0.34 (Table 4.3.2). Phytoplankton were evenly spread throughout the estuary, showing no distinct relationship to salinity (Figure 4.3.7A). The highest vertical averages of phytoplankton were at site 2 ($10866 \pm 1858 \text{ cells ml}^{-1}$) and site 4 ($10453 \pm 1522 \text{ cells ml}^{-1}$).

Longitudinally averaged diatom numbers were highest at 0.5 m depth. Vertically averaged diatom numbers were not correlated to salinity and the highest vertically averaged abundance, $9640 \pm 1834 \text{ cells ml}^{-1}$, was measured at site 2 (Figure 4.3.7B). The highest average number of flagellates was at a depth of 3 m. Like diatoms at this flow, vertically averaged flagellate numbers showed no correlation with salinity (Figure 4.3.7C). The highest vertically averaged abundance, $3512 \pm 197 \text{ cells ml}^{-1}$, was sampled at site 7.

Vertically averaged phytoplankton chlorophyll-a was highest at site 7, the same site as the peak in flagellate numbers.

4.3.3.2.6 Flow: $30.5 \text{ m}^3 \text{ s}^{-1}$

A similar flow of $30.5 \text{ m}^3 \text{ s}^{-1}$ was estimated for sampling sessions on 12 November 1996 and 13 December 1996. A very large flood at $> 2000 \text{ m}^3 \text{ s}^{-1}$ occurred between these sampling dates. Nitrate was only measured on the earlier sampling session, 12 November 1996. Salinity and phytoplankton cell counts measured on the two sampling sessions were averaged. Vertically averaged nitrate concentration ranged from $19.7 \pm 26.3 \text{ } \mu\text{M}$ at the mouth to $106.7 \pm 46.3 \text{ } \mu\text{M}$ at site 7. Nitrate was well mixed throughout the water column with little stratification.

Average number of phytoplankton cells in the estuary was $3618 \pm 221 \text{ cells ml}^{-1}$. Vertically averaged phytoplankton abundance increased significantly (Table 4.3.2) with salinity and were highest, $5062 \pm 223 \text{ cells ml}^{-1}$, at site 2 (Figure 4.3.8A). There was no trend linking phytoplankton abundance with nitrate concentration (Figure 4.3.8B). The lowest vertical average was $1871 \pm 118 \text{ cells ml}^{-1}$ between sites 5 and 6. Diatoms (49 %) and flagellates (47 %) dominated the phytoplankton. Phytoplankton group diversity (H') was 0.37 (Table 4.3.2).

Diatoms were well mixed throughout the water column. Vertically averaged diatom numbers increased significantly as salinity increased (Table 4.3.2) (Figure 4.3.8C) and no correlation with nitrate was evident (Figure 4.3.8D). Vertically averaged numbers were highest, $2803 \pm 141 \text{ cells ml}^{-1}$, at site 2 and decreased to a minimum of $939 \pm 38 \text{ cells ml}^{-1}$ between sites 5 and 6.

Flagellates were well mixed throughout the water column. Vertically averaged flagellate numbers were significantly higher (Table 4.3.2) at high salinity (Figure 4.3.8E) but there was no correlation with nitrate (Figure 4.3.8F). The highest vertically averaged number ($2228 \pm 158 \text{ cells ml}^{-1}$) was between sites 2 and 3, and the lowest ($861 \pm 96 \text{ cells ml}^{-1}$) was between sites 5 and 6.

Phytoplankton chlorophyll-a was well mixed throughout the Gamtoos Estuary. Vertically averaged maxima were located at site 4 (12 November 1996, $9.7 \text{ } \mu\text{g l}^{-1}$) and site 7 (13 December 1996, $29.5 \text{ } \mu\text{g l}^{-1}$). Maxima in abundance of the dominant phytoplankton groups and total phytoplankton did not correspond to the position of the REI at this flow.

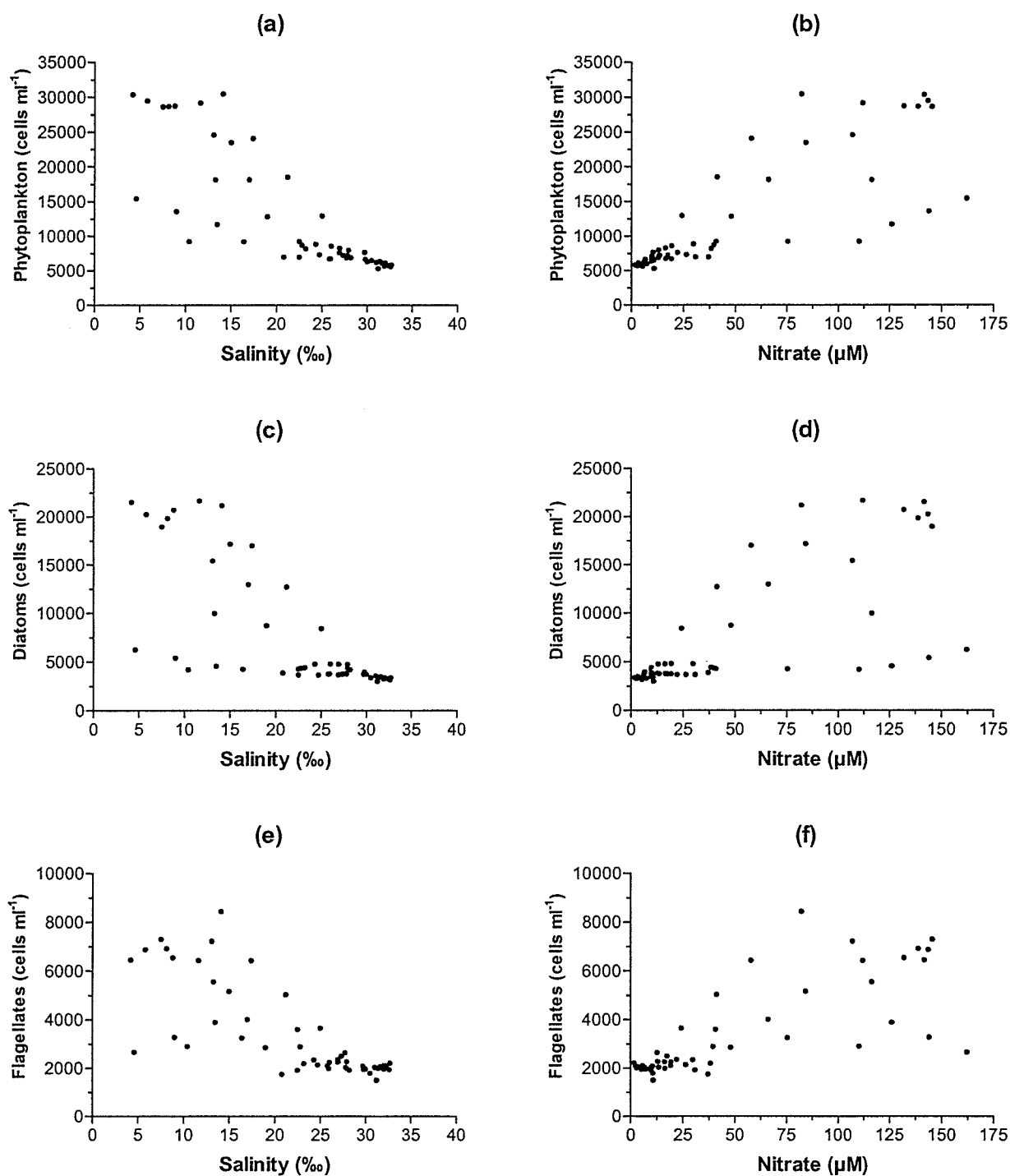


Figure 4.3.5: Total phytoplankton and dominant phytoplankton group numbers in relation to salinity (‰) and nitrate (μM) at a flow of 1.2 m³ s⁻¹. Total phytoplankton (a and b), diatom (c and d) and flagellate (e and f) abundance in relation to salinity and nitrate respectively.

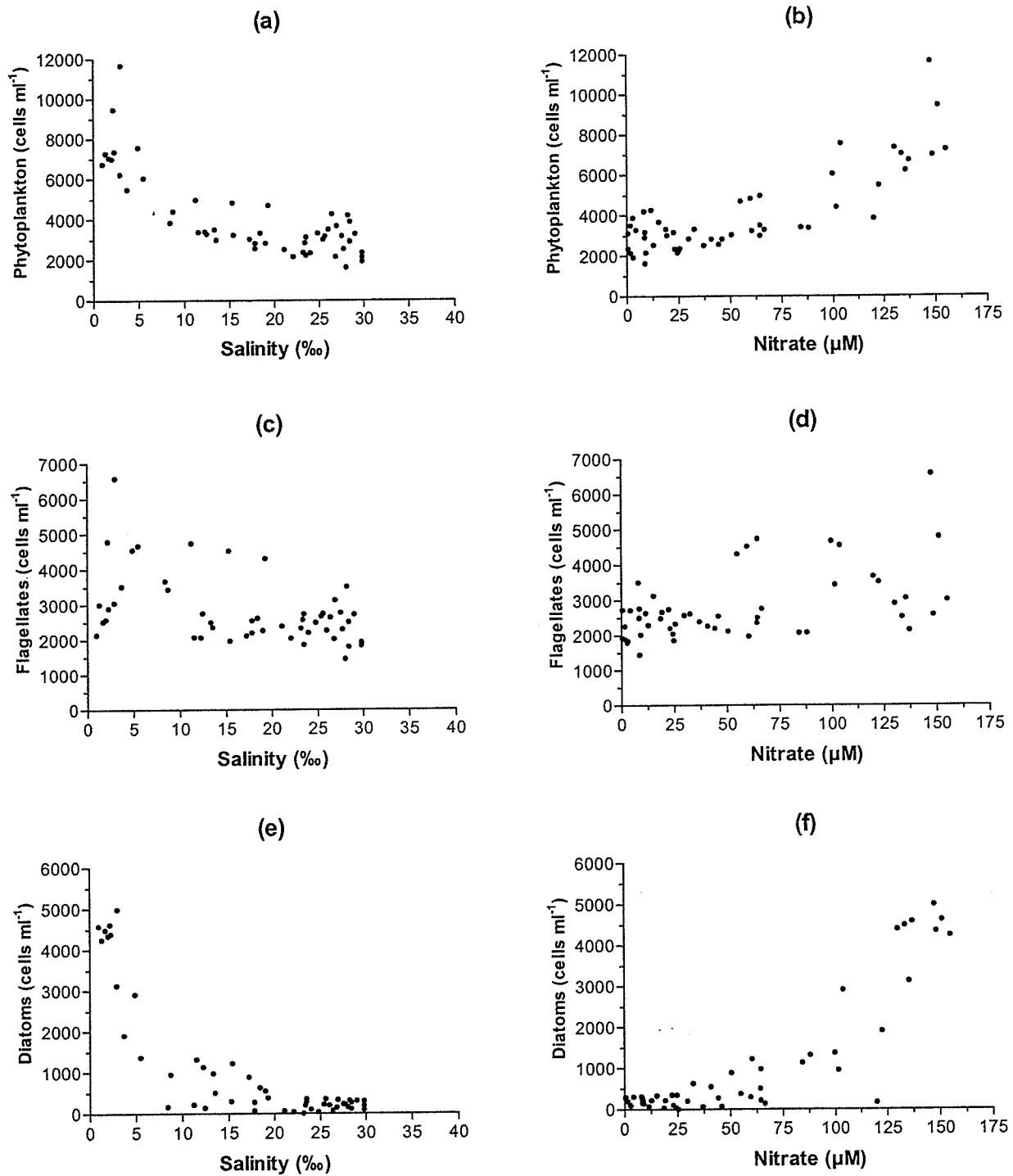


Figure 4.3.6: Total phytoplankton and dominant phytoplankton group numbers in relation to salinity (‰) and nitrate (μM) at a flow of $2.3 \text{ m}^3 \text{ s}^{-1}$. Total phytoplankton (a and b), flagellate (c and d) and diatom (e and f) abundance in relation to salinity and nitrate respectively.

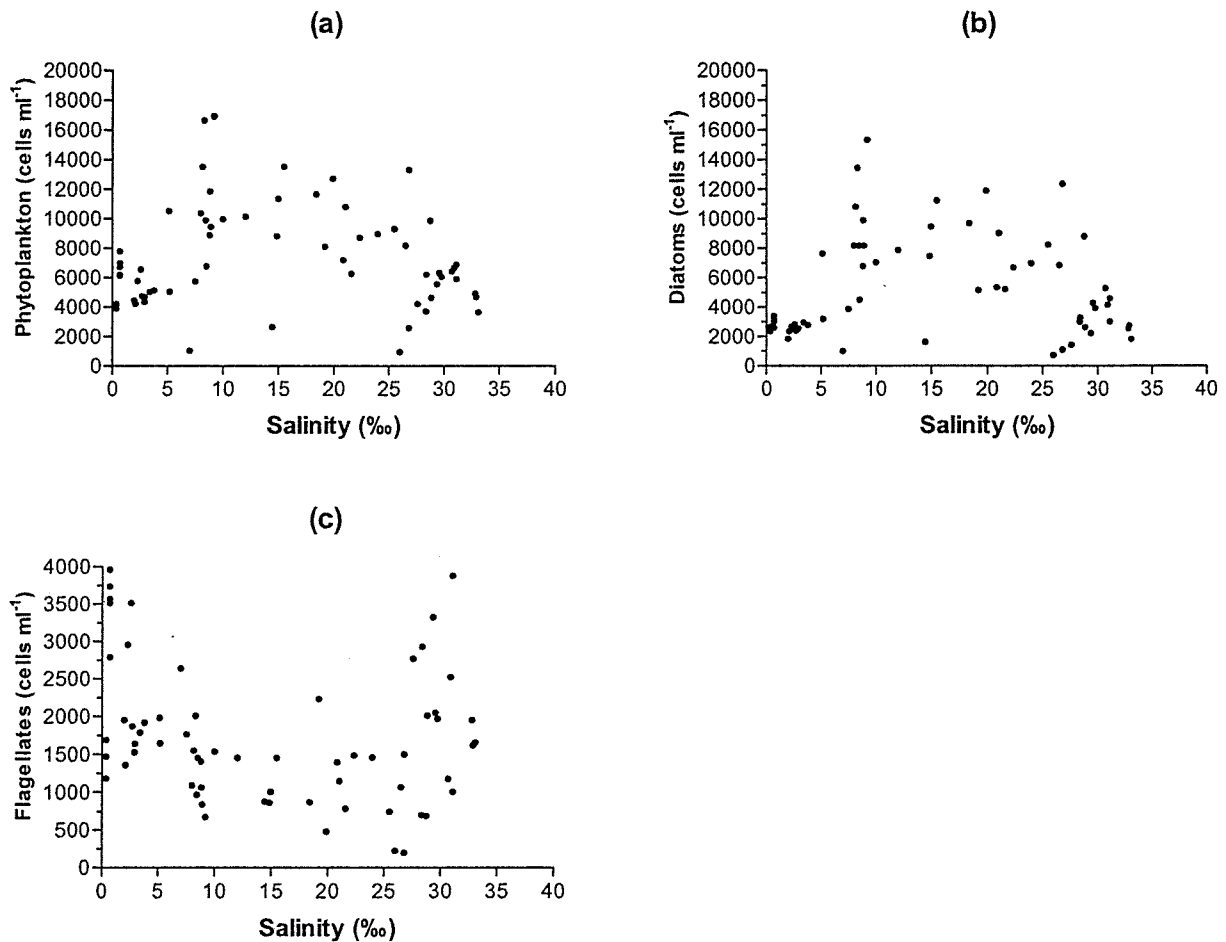


Figure 4.3.7: Total phytoplankton (a) and dominant phytoplankton group numbers (diatoms (b) and flagellates (c) in relation to salinity (‰) at a flow of $9.7 \text{ m}^3 \text{ s}^{-1}$. Nitrate was not measured.

4.3.4 Discussion

The Gamtoos Estuary becomes shallower and narrower with distance from the estuary mouth. Pearce (1996) described the area of the estuary above site 7 as aggraded (i.e. the upper estuary becomes more shallow towards the head as a result of accumulated fluvium), limiting the extent of the tidal influence since the more dense saline water remains at the bottom of the water-column. With the onset of river flooding the upper estuary becomes completely dominated by riverine processes, with tidal influences limited to the lower estuary.

The agricultural drainage pipe at site 9, which discharges runoff from an area of approximately 50 ha, gives an indication of the contribution of nutrients from surface agricultural runoff into the estuary. A mixing zone exists around the discharge point, 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 mineral elements, measured at certain times in the estuary, to within acceptable limits.

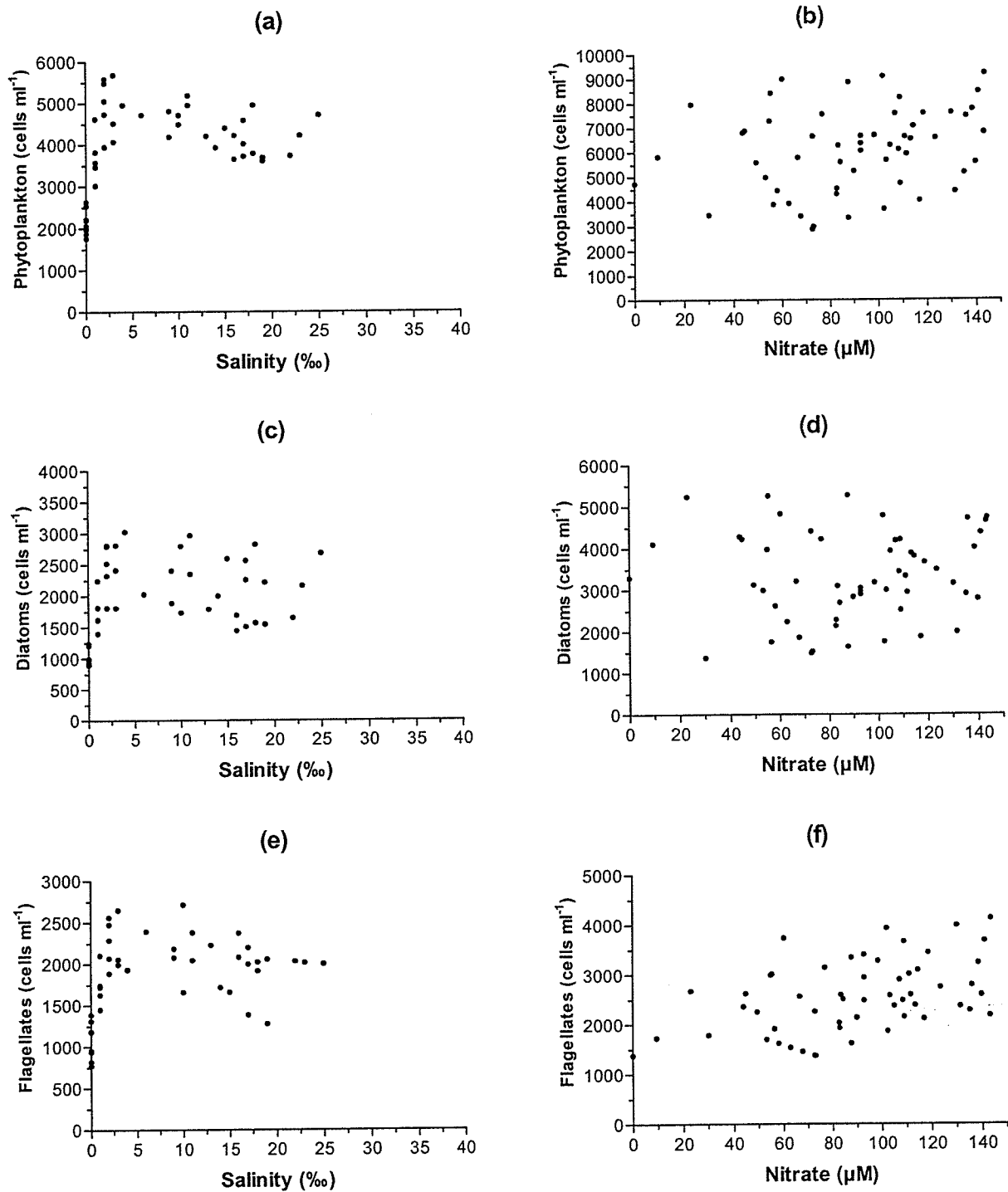


Figure 4.3.8: Total phytoplankton and dominant phytoplankton group numbers in relation to salinity (‰) and nitrate (μM) at a flow of 30.5 m³ s⁻¹. Total phytoplankton (a and b), diatom (c and d) and flagellate (e and f) abundance in relation to salinity and nitrate respectively.

During this study, under conditions of low river input, nitrate in excess of 20 μM was confined to the shallow upper and middle reaches of the estuary. At higher flow rates, nitrate concentrations increased throughout the estuary and particularly near to the mouth. As a result of river input, a strong longitudinal nitrate gradient, contrary to the salinity gradient, was always present. A strong longitudinal salinity gradient was present at all flow rates and a strong vertical salinity gradient was present at flow rates $> 1 \text{ m}^3 \text{ s}^{-1}$.

There was a general increase in phytoplankton abundance as salinity increased, or as nitrate decreased. It was not possible to determine whether phytoplankton abundance responded to salinity or nitrate. Recent estuarine studies have provided evidence for phosphorous being the limiting nutrient to phytoplankton growth in low salinity pools and during periods of high river input (Malone *et al.* 1996; Thompson and Hosja 1996). Even though phosphate was never measured during this study, phytoplankton numbers were always highest in the middle and upper reaches of the estuary, where salinity was lowest, so it is unlikely that phosphorous limited phytoplankton growth during this study. In addition, it is likely that nutrient input by agricultural seepage would have contributed significant levels of phosphorous. Groundwater phosphorous measured in the Gamtoos floodplain by Pearce (1996) ranged between 1.2 and 2.9 m . This is considerably higher than the general concentration of phosphorous measured in natural groundwater, which contains less than 0.1 m (Bouwer 1978, cited in Pearce 1996).

Phytoplankton biomass, group abundance and group diversity only appeared to fall slightly as a result of nitrate limitation at the low flow of $0.3 \text{ m}^3 \text{ s}^{-1}$. This decrease, which was not significant, could have been as a result of the high residence time of nutrients at low flow rates. Phytoplankton was most abundant near site 9 during low flows. This site is influenced by the seepage of nutrients from the Loerie Flats agricultural land, which was described by Pearce (1996). Flagellates and dinoflagellates in particular were most abundant in low salinity water at the head of the estuary. It is unclear whether they are imported in riverwater or if they are responding to the high nutrients in the region. The decrease in average phytoplankton numbers as river flow increased (Figure 4.3.9) was more likely to be the result of the rapid flushing of water from the estuary and by the dilution effects of river discharge. At an average flow of $1 \text{ m}^3 \text{ s}^{-1}$ the $3.6 \times 10^6 \text{ m}^3$ estuary will be flushed in 42 days but this falls to 1.4 days at a flow of $30.5 \text{ m}^3 \text{ s}^{-1}$. Average phytoplankton numbers reached a maximum of $20958 \pm 1750 \text{ cells ml}^{-1}$ at a flow of $0.8 \text{ m}^3 \text{ s}^{-1}$. A study of phytoplankton communities in the Ohio River (Wehr and Thorp 1996) revealed that current velocity was an important regulator of the density of major phytoplankton groups. Higher flow rates led to a decrease in the abundance of phytoplankton in the Ohio River.

Phytoplankton group diversity (H') increased as flow decreased and reached a maximum of 0.50, out of a possible maximum of 0.70, at a flow of $1 \text{ m}^3 \text{ s}^{-1}$. Under average flow conditions, phytoplankton groups other than flagellates and diatoms become more abundant resulting in a greater evenness of numbers between the phytoplankton groups. Diatoms and flagellates dominated the phytoplankton community during the entire period of this study. The average number of flagellates

in the estuary increased as flow decreased and reached a maximum of 8497 ± 773 cells ml^{-1} at a flow of $0.3 \text{ m}^3 \text{ s}^{-1}$ (Figure 4.3.9). These results are supported by a similar study of the Kromme Estuary, which has been deprived of all natural river flow other than large floods. This has resulted in the water in the estuary becoming oligotrophic and having a similar salinity to seawater. These conditions have resulted in a lower phytoplankton group diversity (maximum recorded $H' = 0.22$) dominated by small flagellates. In the Gamtoos system, diatom abundance increased as flow increased, reaching a maximum of 7199 ± 2387 cells ml^{-1} at a flow of $1.2 \text{ m}^3 \text{ s}^{-1}$ (Table 4.3.1).

Strong vertical and longitudinal salinity stratification was present at flow rates in excess of $1 \text{ m}^3 \text{ s}^{-1}$. In addition, average nitrate concentration in the estuary as a result of the increased river input exceeded $40 \text{ } \mu\text{M}$ at these higher flow rates. At the lowest flow during the study, $0.3 \text{ m}^3 \text{ s}^{-1}$, diatom abundance was highest near the head of the estuary and the mouth. As flow increased, diatoms were lowest in water with salinity in excess of 30 ‰. At a flow of $9.7 \text{ m}^3 \text{ s}^{-1}$, diatom numbers were lowest near to the mouth (salinity > 30 ‰) and near to the head of the estuary. At a flow of $30.5 \text{ m}^3 \text{ s}^{-1}$, diatoms were evenly spread throughout the estuary except that their numbers decreased at the head of the estuary. At high flow rates, it appears that riverwater contained low numbers of diatoms relative to the estuary. However, phytoplankton cells were not counted in the riverwater and this would be necessary to confirm this observation.

Other groups which were significant in the phytoplankton during the study included the dinoflagellates (14 % at a salinity of 0.3 ‰) and coccolithophorids (51 % at a salinity of 0.8 ‰). Dinoflagellates became a significant group under stable, low flow conditions and the highest vertically averaged abundance was recorded in the middle reaches of the estuary. The low freshwater input still resulted in a strong longitudinal salinity gradient but the vertical salinity gradient was weak (< 5 ‰). Coccolithophorids also bloomed during low flow conditions and were most abundant in the middle and lower reaches of the estuary where salinity was 18 to 34 ‰. Coccolithophorids were not present in any other samples during any other period of the study.

It was hypothesised at the start of the study that phytoplankton abundance is greatest in the REI region and that group structure would be distinguishable from the rest of the estuary. This was seldom the case although the REI was distinguished by the highest vertically averaged phytoplankton chlorophyll-a. When cell abundance and group structure of the REI was compared to the rest of the estuary at all flows, it was seldom the case that cell abundance was highest and group structure differed from the rest of the estuary. Instead, the only general changes in phytoplankton group structure and cell abundance was between high salinity water near to the mouth of the estuary (> 30 ‰) and the rest of the estuary. Phytoplankton abundance (flagellates, dinoflagellates, chlorophytes and cyanophytes in particular) was generally lowest in the lower reaches of the estuary, with diatom abundance being well distributed throughout the estuary.

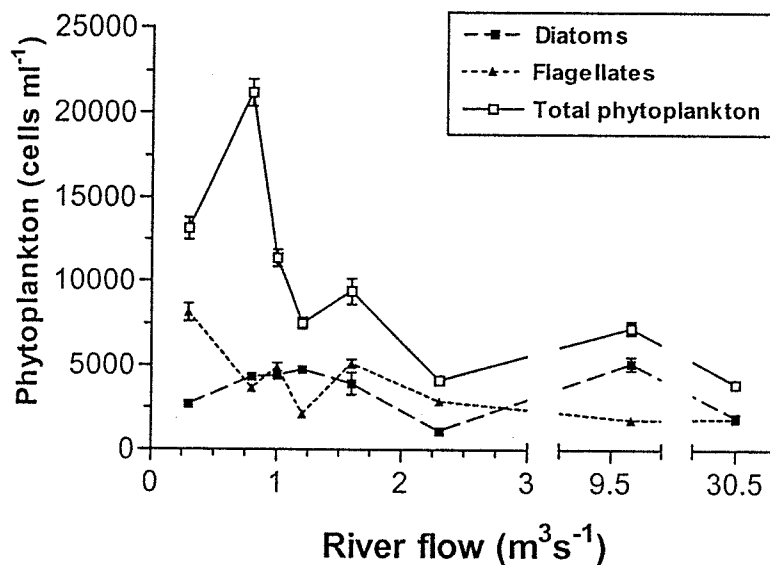


Figure 4.3.9: Average total phytoplankton, diatom and flagellate abundance (cells ml^{-1}) in relation to river input to the Gamtoos Estuary. Vertical bars represent SE mean.

4.3.5 Conclusions

Phytoplankton abundance, group diversity and evenness increased as flow decreased and reached a maximum at a flow of approximately $1 \text{ m}^3 \text{s}^{-1}$. Diatoms and flagellates dominated the phytoplankton during the study period. Flagellates increased as flow and average nitrate concentration decreased. Diatom abundance was lowest during periods of low riverine input ($<1 \text{ m}^3 \text{s}^{-1}$) and low average nitrate concentrations. There was no distinct relationship between the position of the REI and the position of maximum phytoplankton cell abundance and phytoplankton group structure.

The hypothesis set at the beginning of this study indicated that riverwater entering the estuary would be dominated by flagellates. Dinoflagellates become dominant towards the middle reaches if there is strong salinity stratification but if the water is well mixed, diatoms are dominant in the middle reaches. Flagellates were always highest in the upper reaches of the estuary and decreased towards the mouth, except at the highest flow of $30.5 \text{ m}^3 \text{s}^{-1}$. This would suggest that flagellates were highest in riverwater entering the estuary except under very high flow conditions. Dinoflagellates were dominant in the upper and middle reaches of the estuary, particularly during periods of low flow when the water column was stable but vertical salinity stratification was weak. Diatoms were dominant during the entire study, but numbers were greatest when flows were in excess of $1 \text{ m}^3 \text{s}^{-1}$ and salinity stratification was well developed. Results from this study do not support the entire hypothesis. Dinoflagellates, in particular, were most abundant during stable low flow conditions when salinity stratification was weakest. Occasional dinoflagellate blooms were recorded in the middle and lower regions of the estuary.

4.4. Influence of the Gamtoos REI on intertidal and subtidal microphytobenthic communities

4.4.1 Introduction

The microphytobenthos have long been considered a significant component of primary production in shallow estuarine systems (Lukatelich and McComb 1986; Duffy *et al.* 1997), particularly in estuarine regions composed of fine sediments. They also provide an energy source for many deposit feeders and play a major role in benthic processes, influencing the deposition and erosion of sediments (Underwood 1994; Miller *et al.* 1996), and sediment-water nutrient fluxes (Thornton *et al.* 1999).

The presence of spatial and temporal patterns within the estuarine microphytobenthos has been found in many studies (McIntire 1978; Whiting and McIntire 1985; Underwood and Kromkamp 1999). Spatial gradients within microphytobenthos communities have been attributed to several environmental factors, including salinity, temperature, sediment type and organic content, and dissolved nutrient concentrations, particularly porewater nutrients (Underwood 1994; Underwood *et al.* 1998). Often, gradients within microphytobenthos community composition and biomass may be found in relation to nearby human activities which cause changes in the ambient conditions of such environmental parameters (e.g. Underwood *et al.* 1998).

In South Africa, the ambient physico-chemical state of many estuarine systems has been altered due to freshwater abstraction activities (Snow *et al.* 2000). The primary ecological impact is an increase in marine water penetration, and a subsequent reduction in size and position of the region where fresh and marine waters meet and mix within the estuary (the REI). With salinity recognised as an important environmental variable affecting estuarine microphytobenthos communities, it is predicted estuarine freshwater abstraction will ultimately affect the distribution and potentially composition of microphytobenthos within the system. However, as there are few previous ecological studies investigating microphytobenthos communities in South African estuaries, and the influence of environmental parameters on them, it is extremely difficult to predict by how much and in what way communities will be affected by freshwater abstraction schemes. Therefore, the aim of this study was to investigate the influence of several environmental parameters on subtidal and intertidal microphytobenthos communities in the Gamtoos Estuary.

4.4.2 Materials and methods

4.4.2.1 Experimental design

The hypothesis of this research was: "Variation in subtidal and intertidal benthic microalgal biomass at the river-estuary interface (REI) region of the Gamtoos Estuary is attributable to environmental conditions at the sediment-water interface, including sediment characteristics and conditions in the interstitial waters of the sediments."

To test this hypothesis, the following sampling design was implemented:

	ESTUARY		REI	RIVER	
SALINITY	20	15		10	5
SUBTIDAL	S1	S2	⋮	S3	S4
INTERTIDAL	I1	I2	⋮	I3	I4

Where: the REI is defined as the point at which the average salinity of the water column is 10 ‰; S1, S2, S3 and S4 are the subtidal sites, and I1, I2, I3 and I4 are the intertidal sites. Sites are distanced according to a salinity gradient, separated by intervals of 5 ‰. The statistical design was a nested 2-way ANOVA (Table 4.4.1), testing the independent factors region and salinity (or distance upstream from the estuary mouth).

Table 4.4.1: Two-way Repeated Measures ANOVA design applied to field experiment, investigating the impacts of salinity and tidal region on microphytobenthos communities.

Factor	Site									
	Intertidal					Subtidal				
Region										
Salinity	20	15	10	5	1	20	15	10	5	1
Replication	4	4	4	4	4	4	4	4	4	4

4.4.2.2 Sampling and analysis

The following measurements of microphytobenthos, sediment characteristics and the water column were made in both the intertidal and subtidal regions at four sampling points, in March 2000, in the Gamtoos Estuary. All measurements were conducted in replicates of four, from an area of approximately 50 m² at each point (Figure 4.4.1). Sites were selected according to salinity along the horizontal estuarine axis.

4.4.2.3 Microphytobenthos

Sediment samples were collected with a 6 cm (internal diameter) Perspex corer, to a depth of 2 cm. Each sample was preserved with 2 % solution of neutralised formalin. These were then analysed for diatom species in the laboratory. Additional sediment samples were collected with a 1.4 cm (internal diameter) Perspex corer, to a depth of 2 cm, and preserved in 95 % ethanol. These were later analysed for chlorophyll a concentration using a Waters Millipore HPLC with a Waters M-45 pump at an operating pressure of 2000psi, a Waters model 481 LC spectrophotometer, and a Beckman Control System Gold.

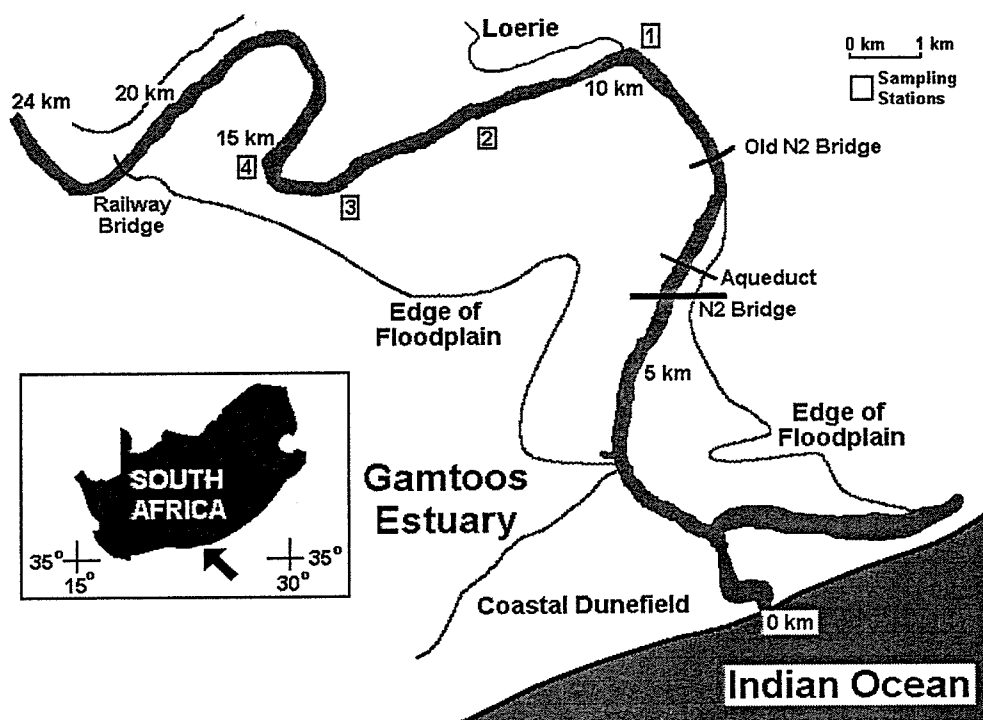


Figure 4.4.1: Location of sampling sites used for microphytobenthos community studies in the Gamtoos Estuary.

4.4.2.4 Sediment characteristics

Sediment samples were collected with a 6 cm (internal diameter) perspex corer, to a depth of 2 cm. Three samples were bulked and squeezed for porewater extraction using a sediment squeezer modeled on the design of the Wildco™ 2214-c30 sediment squeezer. This process was performed four times per tidal region, per site, to obtain replicate samples. Porewater was filtered using Schleicher and Schuell glass fibre filter papers, preserved with mercuric chloride (Gearing 1991), and analysed for nitrate, nitrite, ammonium and phosphate using standard methods (APHA 1992). Porewater salinity, temperature and pH were measured using a hand-held refractometer, a mercury thermometer, and an EDT Microprocessor pH meter Series 3 RE 357, respectively. Additional sediment samples were collected for later analyses of redox potential, using a platinum electrode (Metrohm model 6.0451.100LL), sediment composition and particle size (Gee and Bauder 1986), and organic and calcium carbonate content. Sediment organic content was determined by drying sediment at 100°C for 2 days, then combusting at 450°C for 12h, after which organic content was measured as loss on ignition (%LOI; the difference in weight before and after combustion at 450°C) (Craft *et al.* 1991). Sediment samples were then combusted for a further 12h at 950°C to determine calcium carbonate content (%LOI, the difference in weight before and after combustion at 950°C).

4.4.2.5 Water column

Measurements of the physico-chemical parameters, salinity, temperature, pH, and light were measured at the sediment surface in the subtidal region, using a YSI Inc. model 30M/10FT, an EDT Microprocessor pH meter Series 3 RE 357, and a Li-Cor LI-1000 meter, respectively. 250 ml water samples were collected at the sediment surface in the subtidal region, filtered in the field using Schleicher and Schuell glass fibre filter papers, and preserved immediately using mercuric chloride (Gearing 1991). Filtered water samples were analysed manually for the inorganic nutrients nitrate, nitrite, ammonium and phosphate using standard methods (APHA 1992).

4.4.3 Results

4.4.3.1 Physico-chemical

Sediment porewater salinity and pH varied significantly with distance upstream of the estuary mouth, and the interaction value of space and time was also significant (Figures 4.4.2 and 4.4.3, Table 4.4.2). Post-hoc analyses showed that porewater salinity decreased significantly from sites sampled in the lower estuarine region (10 to 11.5 km from estuary mouth), to those in the upper estuarine region (13.5 to 15 km; Table 4.4.3). Mean porewater salinities ranged from 16.2 to 7.5 ‰, decreasing with increasing distance from the estuary mouth. A number of physico-chemical variables were found to significantly correlate negatively with salinity, including sediment redox potential, and porewater pH, nitrate and phosphate concentrations (Table 4.4.5). pH was found to be significantly lower at the sampling site 10 km from the estuary mouth, relative to all other sites. Mean porewater pH values were between 8.2 and 8.5 units.

Temperatures in the sediment porewater differed significantly with distance upstream of the estuary mouth, between intertidal and subtidal regions, and the interaction of the factors distance and region was also significant (Figure 4.4.4, Table 4.4.2). Mean porewater temperatures were between 24 and 25°C at all sites and regions, with the exception of the subtidal region at the sampling site 10 km from the estuary mouth, where the mean temperature was 20°C.

Porewater nitrate and phosphate concentrations varied significantly with distance upstream of the estuary mouth, between intertidal and subtidal regions, and the interaction of these factors was also significant (Figures 4.4.5 and 4.4.6, Table 4.4.2). *Post-hoc* analyses found significant differences in porewater nitrate concentrations with distance upstream from the estuary mouth and was attributable to relatively lower concentrations recorded at the sampling site 10 km from the mouth (Table 4.4.3). Mean porewater nitrate concentrations in intertidal and subtidal regions were between 6 and 140 µM, and 13 and 42 µM, respectively. Sediment porewater nitrate concentrations were found to correlate positively with sediment redox potential, and porewater pH and phosphate concentrations (Table 4.4.4). Porewater phosphate concentrations differed significantly with distance upstream from the estuary mouth as a result of lower concentrations at sampling sites 10 and 11.5 km upstream relative to sites 13.5 and 15 km upstream of the mouth (Table 4.4.2). Mean porewater phosphate

concentrations in intertidal and subtidal regions of the estuary were between 0.3 and 0.5 μM , and 0.3 and 1 μM , respectively. Mean sediment porewater ammonium and nitrite concentrations were between 0.75 and 1.5 μM , and 5.5 and 8.5 μM , respectively, in both intertidal and subtidal regions (Figures 4.4.7 and 4.4.8, respectively).

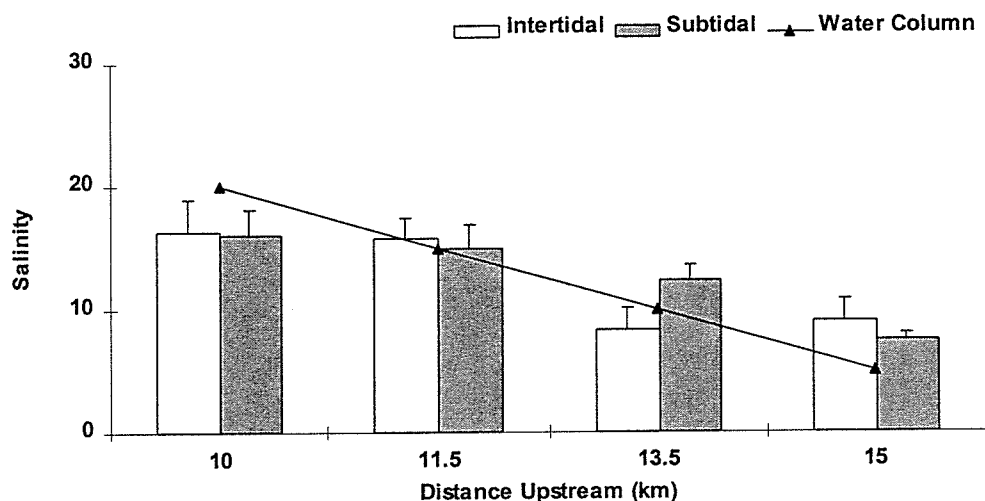


Figure 4.4.2: Mean porewater salinity of intertidal and subtidal sediments, and of bottom waters, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE

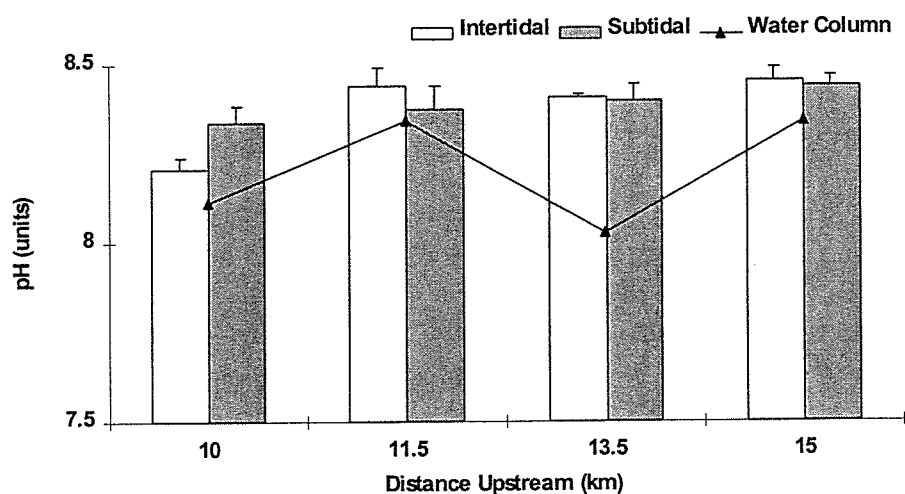


Figure 4.4.3: Mean porewater pH (in units) of intertidal and subtidal sediments, and of bottom waters, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

Table 4.4.2: Results of ANOVAs using sediment, porewater and other environmental variables.

Variable	Factors	d.f.	F	MS	p-level
Salinity	Distance upstream	3	35.625	118.75	***
	Tidal region	1	0.3375	1.125	NS
	Interaction	3	3.6625	12.2083	*
Temperature	Distance upstream	3	1235	12.8646	***
	Tidal region	1	675	7.0312	***
	Interaction	3	1011	10.5312	***
PH	Distance upstream	3	24.2457	0.0455	***
	Tidal region	1	0.4809	0.0009	NS
	Interaction	3	7.6866	0.0144	***
Porewater					
Nitrate	Distance upstream	3	276.0635	1279.048	***
	Tidal region	1	33.5377	155.386	***
	Interaction	3	35.2468	163.304	***
Phosphate	Distance upstream	3	9.4418	0.1766	***
	Tidal region	1	19.9207	0.3727	***
	Interaction	3	6.7138	0.1256	**
Ammonium	Distance upstream	3	0.3571	0.1421	NS
	Tidal region	1	1.0117	0.4026	NS
	Interaction	3	2.275	0.9054	NS
Nitrite	Distance upstream	3	0.5036	29.7426	NS
	Tidal region	1	0.9751	57.5845	NS
	Interaction	3	0.6064	35.8094	NS
Redox	Distance upstream	3	4.0215	35325.6	*
	Tidal region	1	0.5681	4990	NS
	Interaction	3	7.663	67313.23	***
Chlorophyll a	Distance upstream	3	3.6146	4.021	*
	Tidal region	1	5.5166	6.1367	*
	Interaction	3	3.4384	3.8249	*
Sediment Organic Content	Distance upstream	3	11.9101	1.9063	***
	Tidal region	1	28.8251	4.6136	***
	Interaction	3	6.7327	1.0776	**
Sediment Particle Size					
% Sand	Distance upstream	3	30.85386	3484.255	***
	Tidal region	1	1.26145	142.453	NS
	Interaction	3	12.93286	1460.478	***
% Clay	Distance upstream	3	1.41603	7.3729	NS
	Tidal region	1	16.55386	86.19183	***
	Interaction	3	8.34362	43.44313	***
% Silt	Distance upstream	3	26.66561	3561.318	***
	Tidal region	1	0.05264	7.03	NS
	Interaction	3	12.9736	1732.686	***

Table 4.4.3: Results of Post-hoc Tukey tests of variables measured for which the factor site was significant in ANOVA analyses, where 1,2,...etc represent sampling sites at 10, 11.5,...km etc upstream of the mouth (* = < 0.05, ** = < 0.01, *** = < 0.001, NS = not significant) (n=4).

Variable		p-level			
		1	2	3	4
Salinity	2	NS			
	3	***	***		
	4	***	***	NS	
Temperature	2	***			
	3	***	***		
	4	***	***	***	
pH	2	***			
	3	***	NS		
	4	***	NS	NS	
Porewater					
Nitrate	2	NS			
	3	***	***		
	4	***	***	***	
Phosphate	2	NS			
	3	*	NS		
	4	***	**	NS	
Redox	2	NS			
	3	NS	NS		
	4	NS	*	NS	
Chlorophyll a	2	NS			
	3	NS	NS		
	4	NS	*	NS	
Sediment Organic Content	2	**			
	3	***	NS		
	4	NS	*	**	
Sediment Particle Size					
% Sand	2	***			
	3	***	NS		
	4	NS	***	***	
% Silt	2	***			
	3	***	NS		
	4	NS	***	***	

Table 4.4.4: Results of Post-hoc Tukey tests of variables measured for which the factor tidal region was significant in ANOVA analyses, where 1,2,...etc represent sampling sites at 100, 11.5,...km etc upstream of the mouth (* = < 0.05, ** = < 0.01, *** = < 0.001, NS = not significant) (n=4).

Variable		p-level	
		1	2
Temperature	2	***	
Porewater			
Nitrate	2	***	
Phosphate	2	***	
Chlorophyll a	2	*	
Sediment organic content	2	***	
Sediment particle size	2	***	

Table 4.4.5: Results of significant correlations between variables measured (* = < 0.05, ** = < 0.01, *** = < 0.001, NS = not significant) (n=32).

Factors	Salinity	Nitrate	Redox	pH	PO ₄ -P	Chl a	Temp	% Sand	% Clay	% Silt
Salinity										
Nitrate	-0.8 (**)									
Redox	-0.4 (*)	0.5 (**)								
PH	-0.5 (**)	0.5 (**)	0.38 (*)							
PO ₄ -P	-0.42 (*)	0.38 (*)	NS	NS						
Chlorophyll a	NS	NS	NS	NS	-0.48 (**)					
Temperature	NS	NS	-0.45 (*)	NS	NS	NS				
% Sand	NS	NS	NS	NS	NS	NS	NS			
% Clay	-0.38 (*)	0.42 (*)	0.38 (*)	NS	NS	NS	NS	NS		
% Silt	NS	NS	NS	NS	NS	NS	NS	-0.99 (***)	NS	
SOC	NS	NS	NS	NS	NS	0.51 (**)	NS	0.48 (**)	0.5 (**)	-0.54 (**)

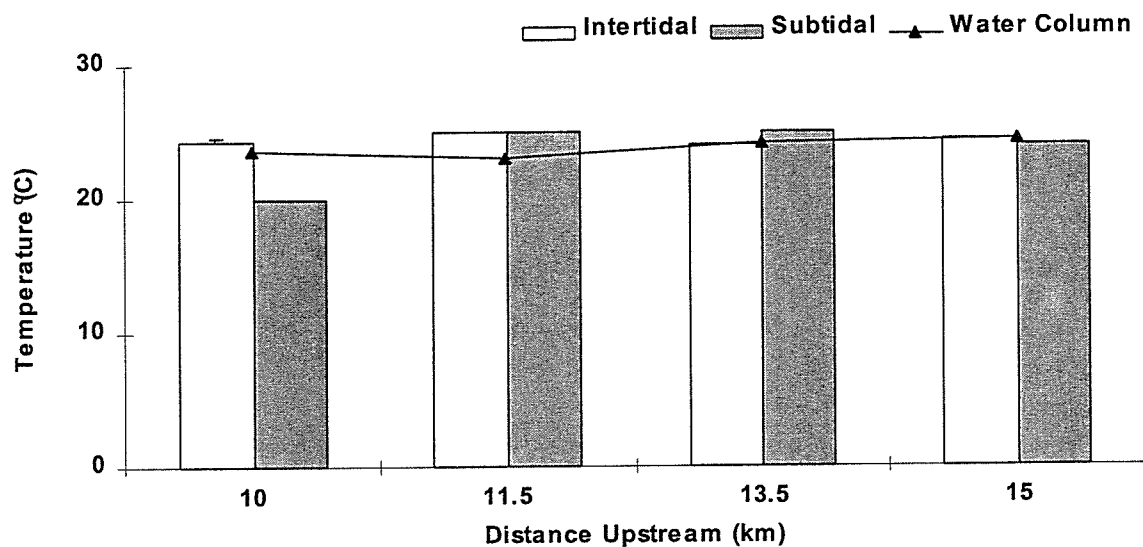


Figure 4.4.4: Mean porewater temperature (in °C) of intertidal and subtidal sediments, and of bottom waters, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

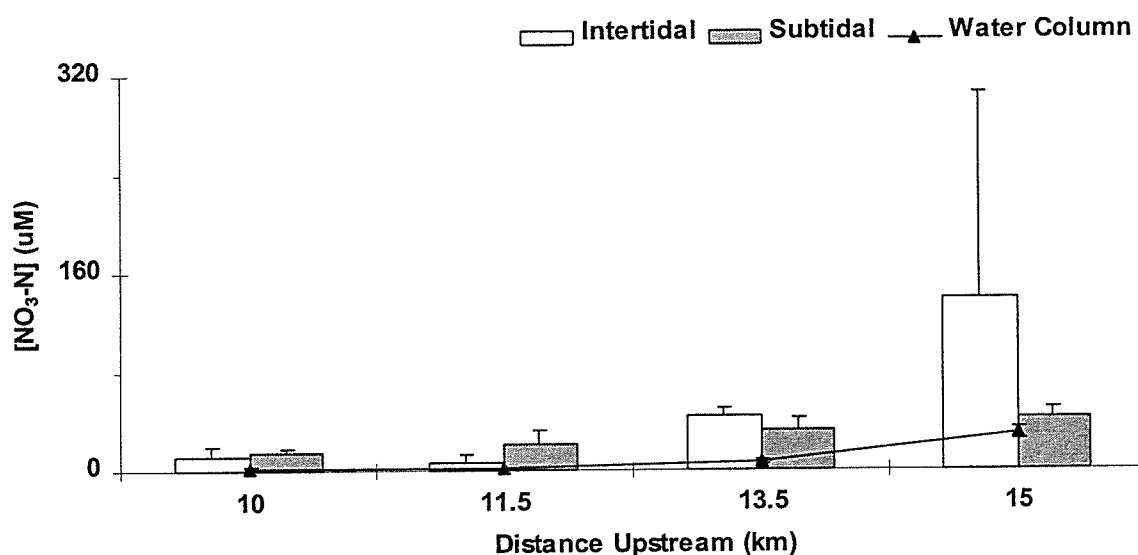


Figure 4.4.5: Mean porewater nitrate concentrations (in μM) of intertidal and subtidal sediments, and bottom waters, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

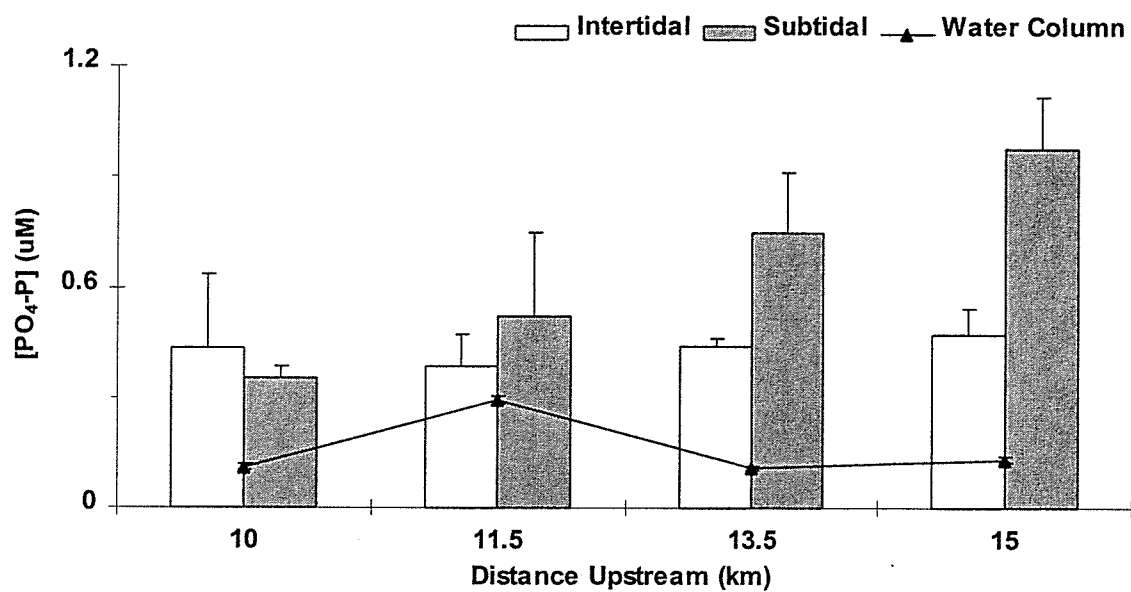


Figure 4.4.6: Mean porewater phosphate concentrations (in μM) of intertidal and subtidal sediments, and bottom waters, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

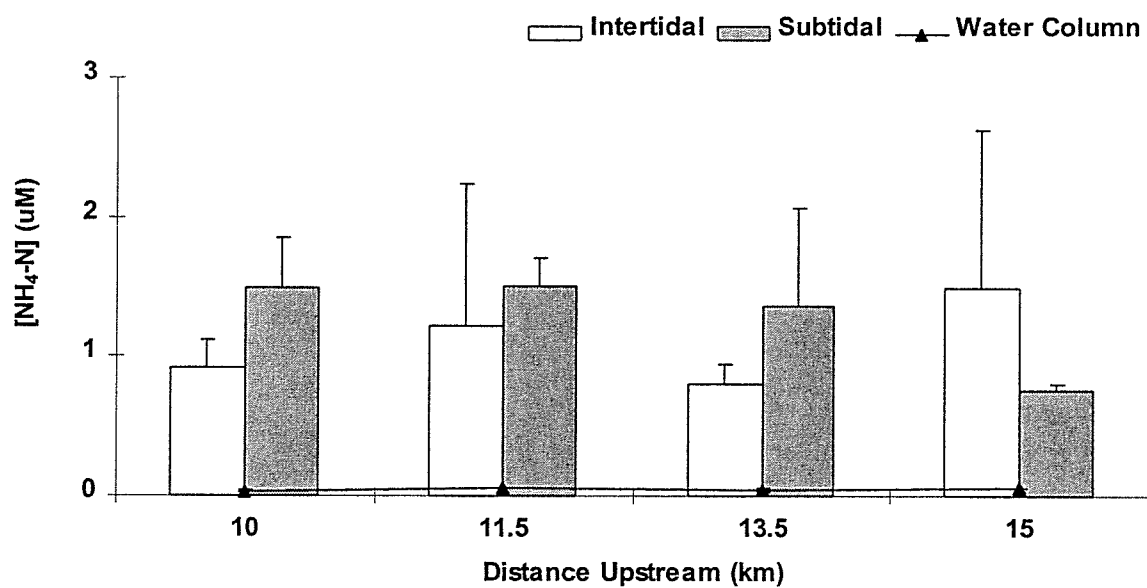


Figure 4.4.7: Mean porewater ammonium concentrations (in μM) of intertidal and subtidal sediments, and bottom waters, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

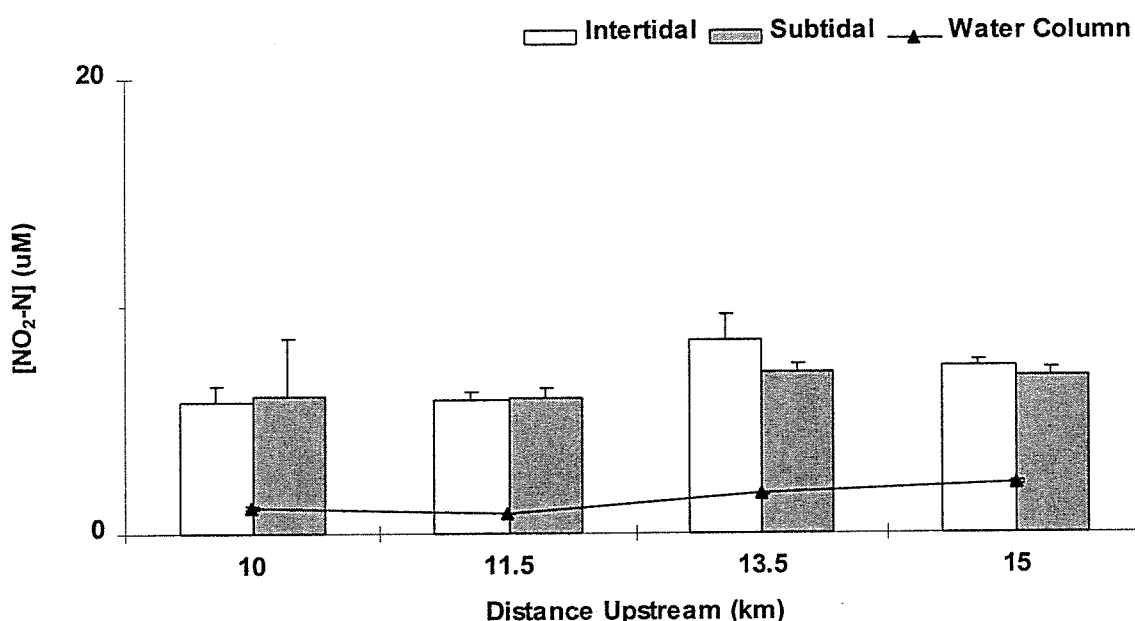


Figure 4.4.8: Mean porewater nitrite concentrations (in μM) of intertidal and subtidal sediments, and bottom waters, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

4.4.3.2 Sediment characteristics

Sediment redox potential varied significantly with distance from the estuary mouth, and the interaction of distance upstream and tidal region was also significant (Figure 4.4.9, Table 4.4.2). *Post-hoc* analyses found variation with distance upstream was the result of significant differences in sediment redox potentials between sites at 11.5 and 15 km upstream of the mouth (Table 4.4.3). Mean redox potentials for intertidal and subtidal sediments were between 12 and -175 mV, and 97 and -152 mV, respectively. Sediment redox potential was found to have a significant, positive correlation with pH, and a significant negative correlation with temperature (Table 4.4.5).

Sediment organic content varied significantly with distance upstream from the estuary mouth, between inter- and sub-tidal regions, and the interaction value of these factors was also significant (Figure 4.4.10, Table 4.4.2). *Post-hoc* analyses revealed that significant differences with distance upstream were due to higher organic contents recorded at sites 11.5 and 13.5 km upstream from the mouth, relative to organic contents measured at sites 10 and 15 km from the mouth (Table 4.4.3). Intertidal sediment organic contents were significantly higher (means between 0.4 and 2.3 %), than organic contents measured in subtidal sediments (means between 0.4 and 0.8 %) (Table 4.4.4).

Percent sand, clay and silt composition was determined for sediments at all sampling sites (Figure 4.4.11). Percentage sand and silt compositions of sediments varied significantly with distance upstream from the estuary mouth, percentage clay varied significantly between tidal regions, and the interaction of distance upstream and tidal region was significant for all three composition types (Table

4.4.2). Post-hoc analyses showed silt and sand compositions at sites 10 and 15 km upstream of the estuary mouth were significantly higher and lower, respectively, than at sites 11.5 and 13.5 km upstream of the mouth (Table 4.4.3). Further analyses revealed that sediment clay composition was significantly higher in intertidal regions than in subtidal regions (Table 4.4.4). Mean percent sand and silt composition of sediments ranged between 14 and 79 %, and 11 and 69 %, respectively. Mean sediment clay content was much less variable, between 8 and 18 %. Sediment clay composition was found to correlate significantly with several environmental factors including salinity, sediment redox potential and organic content, and porewater nitrate concentrations (Table 4.4.5). Percent sand and silt compositions of sediment were also found to correlate significantly with sediment organic content.

4.4.3.3 Biological characteristics

Sediment chlorophyll *a* concentrations were measured as an indicator of microalgal biomass (Figure 4.4.12). Benthic chlorophyll *a* concentrations varied significantly with distance upstream of the estuary mouth, between intertidal and subtidal regions, and the interaction of these two factors was also significant (Table 4.4.2). Sediment chlorophyll *a* concentrations at sites 11.5 and 13.5 km upstream from the estuary mouth were higher than concentrations measured at sites 10 and 15 km from the estuary mouth (Table 4.4.3). Intertidal chlorophyll *a* concentrations (means between 1.15 and 3.6 $\mu\text{g cm}^{-3}$) were significantly higher than those of subtidal regions (means between 0.6 and 2.2 $\mu\text{g cm}^{-3}$) (Table 4.4.4). Benthic chlorophyll *a* concentrations were found to significantly correlate with sediment porewater phosphate concentrations and organic content (Table 4.4.5).

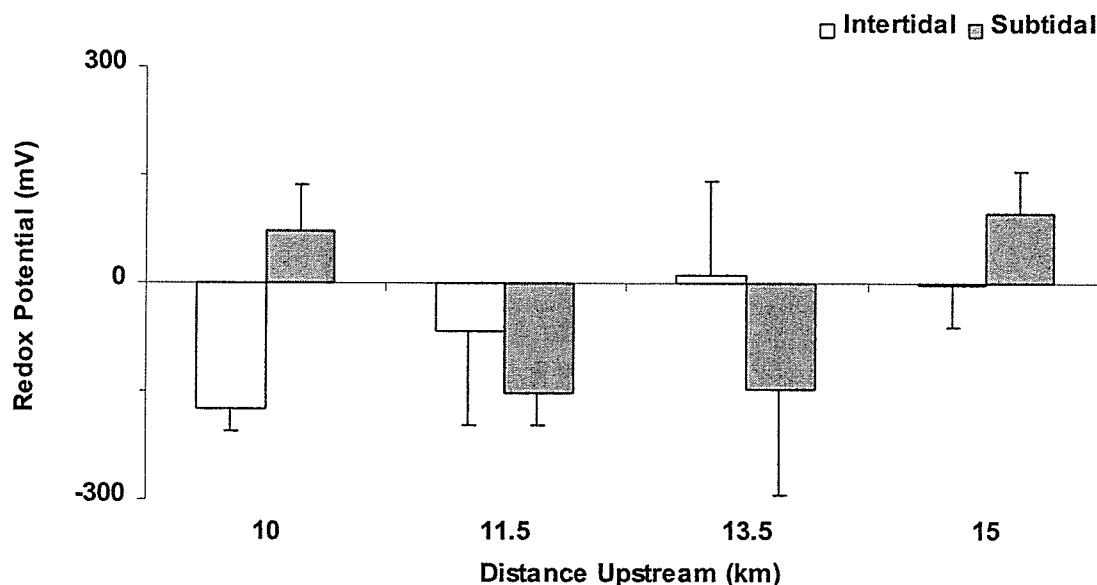


Figure 4.4.9: Mean sediment redox potential (in mV) of intertidal and subtidal sediments, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

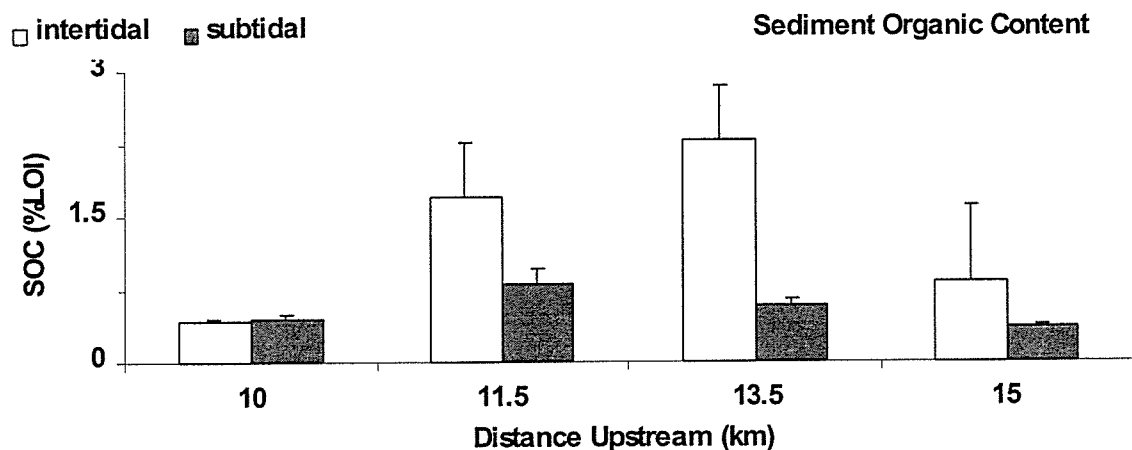


Figure 4.4.10: Mean sediment organic content (in %) of intertidal and subtidal sediments, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

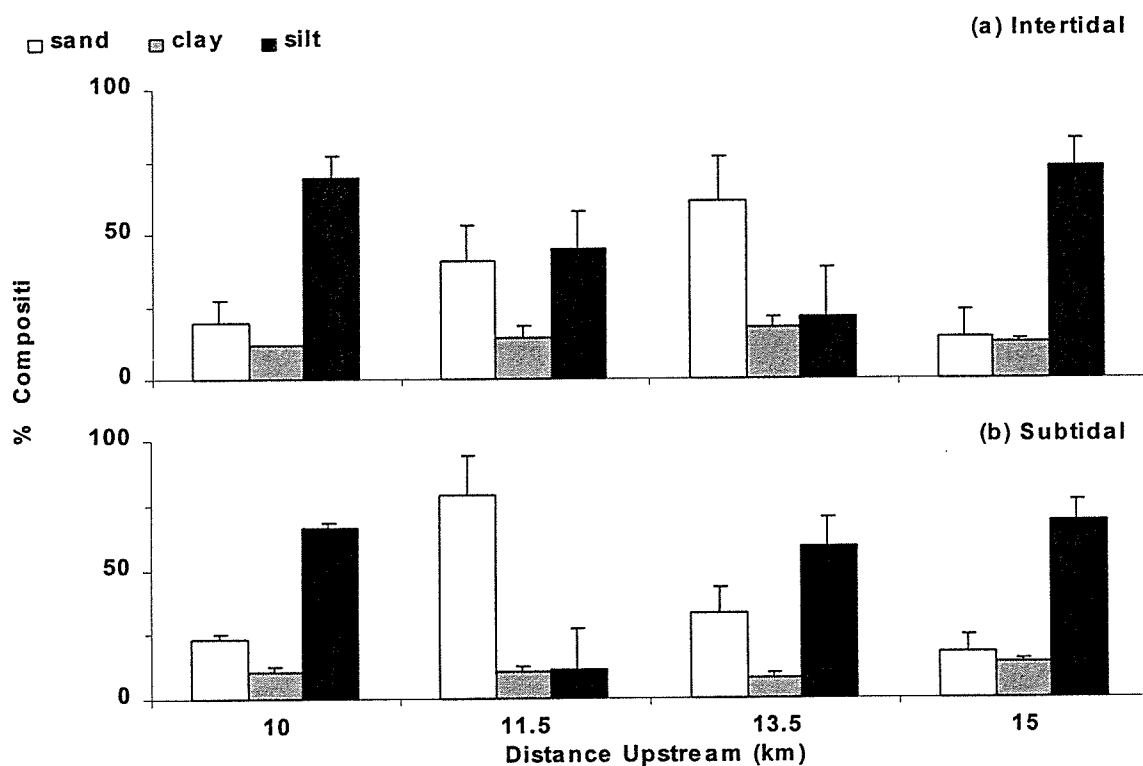


Figure 4.4.11: Mean sediment sand, clay and silt composition (in %) of intertidal (a) and subtidal (b) sediments, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

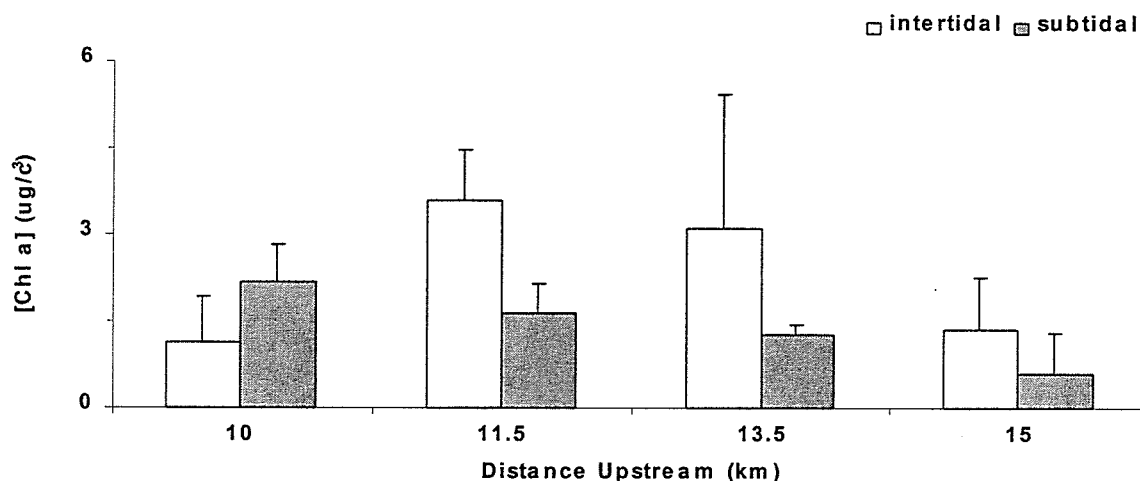


Figure 4.4.12: Mean sediment chlorophyll a concentration (in $\mu\text{g cm}^{-3} = 0.1 \text{ mg m}^{-2}$) of intertidal and subtidal sediments, with increasing distance upstream from the estuary mouth ($n=4$) $\pm$ SE.

4.4.4 Discussion

These preliminary results infer some extremely important management considerations for the Gamtoos Estuary. Firstly, benthic microalgal biomass was found to be significantly higher in the intertidal region than the subtidal region. The intertidal region is a depositional area within an estuary, commonly having relatively high sediment organic content and small sediment particle size (Underwood 1994). With high sediment organic content, high levels of bacterial mineralisation and inorganic nitrogen concentrations are found. Sediment composed of smaller particle size is more stable and easier to colonise than sediments of coarser particle size (Colijn and Dijkema 1981). In addition to sediment characteristics, the intertidal area is an environment with relatively low light attenuation, a factor which often limits benthic microalgal productivity and biomass (Lukatelich and McComb 1986). As a result of these influential environmental variables a relatively high estuarine microphytobenthic biomass is commonly found in the intertidal zone (Colijn and Dijkema 1981; Underwood and Kromkamp 1999). Consequently, the intertidal area may be recognised as an important microalgal environment and region of primary productivity within an estuary. Therefore, it is imperative to maintain this environment in estuarine management activities. Thus, when evaluating the viability of further freshwater abstraction in the Gamtoos Estuary, predicted impacts on intertidal regions need to be considered carefully, as impacts on the environment will directly affect microphytobenthic communities, and therefore estuarine productivity. For example, excessive freshwater abstraction from an estuary may alter the status of the mouth (i.e. open to closed), and in turn alter tidal influences within the estuary, potentially decreasing or removing intertidal habitats.

Secondly, microalgal biomass was highest in areas, including the REI region, where salinities were between 10 and 15 ‰. Previous studies have found benthic microalgae to be tolerant of wide salinity ranges (e.g. 2 to 45 ‰), however, marked changes in community composition and biomass occur at salinities less than 8 ‰ (Underwood 1994). Salinities of between 10 and 15 ‰ are frequently encountered in the middle and upper reaches of permanently open estuaries, and it is likely that this region is the most physiological tolerable section of an estuary for estuarine organisms. Thus, it is not surprising that the highest microalgal biomass is generally found within this region. With this being the region of highest benthic microalgal biomass, it should be recognised as an important 'habitat' within the estuary. It is predicted that the extent of this region will decrease with freshwater abstraction activities, as a decrease in freshwater volume will also decrease the expanse of oligohaline and mesohaline waters. In addition, the longitudinal position of the REI region will be altered, moving further upstream from the estuary mouth. These are all impacts that will influence benthic microalgal communities within the Gamtoos Estuary.

Finally, microalgal biomass was found to positively correlate with porewater phosphate concentrations. No relationships were found with any of the inorganic nitrogen porewater nutrients measured. Previous studies have found benthic microalgal communities were not directly influenced by the overlying water column characteristics (Gruendling 1971; Lukatelich and McComb 1986). However, it is acknowledged that benthic microalgae acquire inorganic nutrients from sediment porewaters (Underwood and Kromkamp 1999). Underwood *et al.* (1998) have reported strong relationships between benthic porewater ammonium concentrations and microalgal biomass. However, literature documenting relationships between porewater phosphate concentrations and benthic microalgal biomass were not found. The results presented here highlight the severity of phosphate limitation in benthic primary producer communities in the Gamtoos Estuary, and that it is this inorganic nutrient influencing benthic microalgae. This observation is from a single season in one estuarine system, however, and should be considered cautiously at this point. It would indicate though that managers should be aware of future changes in sources influencing porewater phosphate concentrations in the Gamtoos Estuary (surface run-off, groundwater, and possibly riverine inputs), as changes in porewater phosphate concentrations will influence benthic microalgal biomass and possibly community structure and composition.

In conclusion, preliminary results indicate the major factors influencing benthic microalgal communities in the Gamtoos Estuary are the intertidal habitat in mesohaline regions, and porewater phosphate concentrations. Therefore, future activities and management considerations for the Gamtoos Estuary should endeavour to maintain these two factors at present levels to avoid direct or indirect impacts on benthic microalgal communities.

4.5 General conclusions

Conclusions reached regarding the original hypotheses (given between quotation marks):

1. "Phytoplankton $\text{m}^3 \text{s}^{-1}$ biomass would peak at a flow rate of between 1 to $8 \text{ m}^3 \text{s}^{-1}$ in the Gamtoos Estuary". The highest phytoplankton biomass was recorded at flows of 0.8 and $1.2 \text{ m}^3 \text{s}^{-1}$. The hypothesis is accepted as a flow of $\sim 1 \text{ m}^3 \text{s}^{-1}$ resulted in the maximum estuarine phytoplankton biomass.
2. "Three tidal cycles are required for maximum phytoplankton production". If the mean volume of the Gamtoos Estuary is taken to be $3.6 \times 10^6 \text{ m}^3$, the estuary will only be completely flushed in about 42 days at a flow rate of $1 \text{ m}^3 \text{s}^{-1}$. This translates into 3 tidal cycles and the hypothesis is thus accepted.
3. "Benthic chlorophyll-a is high when water flow is low because sediment erosion is minimal". Subtidal and intertidal benthic chlorophyll-a was highest at flows of 1 ($57.7 \text{ kg} \pm 0.4 \text{ SE}$) and $1.16 \text{ m}^3 \text{s}^{-1}$ ($8.7 \text{ kg} \pm 0.1 \text{ SE}$) respectively. Although sediment erosion was not measured in these experiments, subtidal and intertidal benthic microalgal biomass was lower at flow rates greater than 1 and $1.16 \text{ m}^3 \text{s}^{-1}$ respectively. However, mineral nutrient availability becomes a limiting factor at flow rates lower than those mentioned previously. In this series of experiments, the details of the hypothesis were not tested but the numerical results were not different to the ideas contained in the hypothesis. It will be very difficult to test the details of this hypothesis because at flow rates $> 2 \text{ m}^3 \text{s}^{-1}$ sub-surface observations and measurements are both difficult and dangerous.
4. "As the flow of water entering the estuary increases, so the average measurable plant mineral nutrient concentration of the water increases". The average nitrate level for the estuary increased from $1.94 \mu\text{M} \pm 0.21 \text{ SE}$ at a flow of $0.3 \text{ m}^3 \text{s}^{-1}$ to $83.6 \mu\text{M} \pm 13.8 \text{ SE}$ at a flow of $30.5 \text{ m}^3 \text{s}^{-1}$. This increase in nitrate concentration with flow rate supports the hypothesis.
5. "Variation in subtidal and intertidal benthic microalgal biomass at the river-estuary interface region is attributable to environmental conditions at the sediment-water interface, including sediment characteristics and conditions in the interstitial waters of the sediments". Preliminary results showed that the major factors influencing benthic microalgal communities in the Gamtoos Estuary were intertidal habitat in mesohaline regions, and porewater phosphate concentrations.

The global hypothesis for microalgae was as follows: "The structure and dynamics of microalgae in the REI region in the Gamtoos Estuary is governed by the quality and quantity of riverwater entering the estuary." The vertical average of phytoplankton chlorophyll-a was used to determine the position of peak phytoplankton biomass. The high biomass of phytoplankton in the REI plays an

important role in removing excessive mineral nutrients from the estuary and as a food source supporting higher trophic levels. Mineral nutrients are imported into the estuary from the catchment in river water. As river flow increases, the concentration of mineral nutrients in the estuary increases, thus supporting a greater biomass of phytoplankton in the REI. However, if river flow increases above $1 \text{ m}^3 \text{ s}^{-1}$, then the estuary is flushed too quickly (> 3 spring tidal cycles) and phytoplankton biomass decreases. There was no evidence that the phytoplankton structure in the REI differed from the rest of the estuary. The only change was a general decrease in phytoplankton abundance in the lower reaches of the estuary where salinity exceeded 30 ‰. To maintain a maximum biomass of phytoplankton in the REI, an average annual flow of $1 \text{ m}^3 \text{ s}^{-1}$ is required. At this flow, the REI is positioned at site 7 and the vertically averaged salinity was 12.5 ‰.

Results from the study indicate that a base flow of $1 \text{ m}^3 \text{ s}^{-1}$ is optimal for pelagic and benthic microalgal growth and phytoplankton group diversity in the Gamtoos Estuary. It is essential that the base flow does not become interrupted for long periods or severely reduced. This will result in a weakening of salinity stratification, reducing microalgal biomass and phytoplankton group diversity, particularly within the REI.

Floods are essential to scour accumulated sediment and organic matter from estuaries but further research is required to determine their importance in relation to both pelagic and benthic associated microalgae. In addition to floods, a study of microalgae over a daily or spring tidal cycle, particularly a phytoplankton bloom, could contribute to our understanding of microalgal dynamics in the estuary. Unfortunately, benthic microalgal species identification and counts were unsuccessful but are essential for trophic status studies and should still be considered for investigation. River flow estimates into the Gamtoos Estuary have been made difficult by the distance of flow recorders from the estuary ($> 100 \text{ km}$). To achieve more accurate results in relation to flow, it should be considered a priority for a flow recorder to be situated above the head of the estuary.

According to the Mike 11 model (see Chapter 2), the 10 ‰ salinity level will be located 8 to 9 km from the mouth of the Gamtoos Estuary at a flow of $1 \text{ m}^3 \text{ s}^{-1}$. From this information, it will be easier for dam managers to ensure that the optimal amount of water is released to maintain the ecological function of the estuary and REI (based on this microalgal sub-project). A flow of $1 \text{ m}^3 \text{ s}^{-1}$ will flush the estuary in 3 spring tidal cycles (42 days) assuming the estuary has a volume of $3.6 \times 10^6 \text{ m}^3$. Three tidal cycles was also optimal for phytoplankton in the nearby Sundays Estuary (Hilmer 1990). If this is the optimal residence time for a permanently open estuary, then the required base flow to maintain an optimal biomass and group diversity of estuarine microalgae should be able to be estimated from the volume of the estuary being studied. This does not allow for local conditions such as trophic status and temperature but does allow for a rough flow estimate to be made, particularly for poorly studied estuarine systems.

4.6 Acknowledgements

The Water Research Commission is thanked for funding this study. The National Research Foundation provided a bursary for Mr G. Snow and funded a post-doc fellowship for Dr H. Astill. Bernadette Snow and Brian Colloty are thanked for assistance in the field.

4.7 References

- Ahel, M., Barlow, R.G. and Mantoura, R.F.C. 1996. Effect of salinity gradients on the distribution of phytoplankton pigments in a stratified estuary. *Marine Ecology Progress Series* 143: 289-295.
- Allanson, B.R. and Read, G.H.L.. 1995. Further comment on the response of Eastern Cape Province estuaries to variable freshwater inflows. *South African Journal of Aquatic Science* 21(1/2): 56-70.
- APHA. 1992. Standard Methods of the examination of water and wastewater. American Public Health Association, Washington.
- Bate, G.C. and Heelas, B.V. 1975. Studies on the nitrate nutrition of two indigenous Rhodesian grasses. *Journal of Applied Ecology* 12: 941-952.
- Boney, A.D. 1989. Phytoplankton. Edward Arnold, London.
- Colijn, F. and Dijkema, K.S. 1981. Species composition of benthic diatoms and distribution of chlorophyll-a on an intertidal flat in the Dutch Wadden Sea. *Marine Ecology Progress Series* 4: 9-21.
- Craft, C.B., Seneca, E.D. and Broome, S.W. 1991. Loss on ignition and Kjeldahl digestion estimating organic carbon and total nitrogen in estuarine marsh soils: calibration with dry combustion. *Estuaries* 14: 175-179.
- Cromar, N.J. and Fallowfield, H.J. 1997. Effect of nutrient loading and retention time on performance of high rate algal ponds. *Journal of Applied Phycology* 9: 301-309.
- Dawes, C.J. 1981. Marine botany. John Wiley & Sons, New York: pp. 306-309.
- Duffy, E.J., Kinlan, B.P. and Valiela, I. 1997. Influence of grazing and nitrogen loading on benthic microalgal biomass in estuaries of Waquoit Bay, Massachusetts. Boston University Marine Program, Massachusetts, Unpublished Report.
- Eyre, B. and Twigg, C. 1997. Nutrient behaviour during post-flood recovery of the Richmond River estuary Northern NSW, Australia. *Estuarine, Coastal and Shelf Science* 44: 311-326.
- Gee, G. W. and. Bauder, J.W. 1986. Particle-size analysis. In: Methods of soil analysis. Part I. Physical and mineralogical methods (ed. A. Kluten). American Society of Agronomy, Wisconsin: pp. 383-411.
- Gruendling, G. K. 1971. Ecology of the epipelagic algal communities in Marion Lake, British Columbia. *Journal of Phycology* 7: 239-249.

- Gearing, J. N. 1991. The study of diet and trophic relationships through natural abundance ^{13}C . In: Carbon isotope techniques (ed. D.C. Coleman and B.F. Coleman). Academic Press, New York: pp. 201-218.
- Hilmer, T. 1990. Factors influencing the estimation of primary production in small estuaries. Ph.D. Thesis, Department of Botany, University of Port Elizabeth.
- Hilmer, T. and Bate G.C. 1990. Covariance analysis of chlorophyll distribution in the Sundays River estuary, Eastern Cape. *South African Journal of Aquatic Science* 16(1/2): 37-59.
- Hilmer, T. and Bate, G.C. 1991. Vertical migration of a flagellate-dominated bloom in a shallow South African estuary. *Botanica Marina* 34: 113-121.
- Heineken, T.J.E. 1981. Estuaries of the Cape. Part II: Synopses of available information on individual systems. CSIR Report No. 7: Gamtoos (CMS48). CSIR Research Report 406: 40 pp.
- Kennish, M.J. 1992. Ecology of estuaries: anthropogenic effects. *Marine Science Series* 1: 33-61.
- Köhler, J. 1997. Measurements of *in situ* growth rates of phytoplankton under conditions of simulated turbulence. *Journal of Plankton Research* 19(7): 849-862.
- Lukatelich, R.J. and McComb, A.J. 1986. Distribution and abundance of benthic microalgae in a shallow southwestern Australian estuarine system. *Marine Ecology Progress Series* 27: 287-297.
- Lund, J.W.G, Kipling, C. and Lecren, E.D. 1958. The inverted microscope method of estimating algal numbers and the statistical basis of estimations by counting. *Hydrobiologia* 11: 143-170.
- Macintyre, H.L. and Cullen, J.J. 1996. Primary production by suspended and benthic microalgae in a turbid estuary: time-scales of variability in San Antonio Bay, Texas. *Marine Ecology Progress Series* 145: 245-268.
- Malone, T.C., Conley, D.J., Fisher, T.R., Gilbert, P.M., Harding, L.W. and Sellner, K.G. 1996. Scales of nutrient-limited phytoplankton productivity in Chesapeake Bay. *Estuaries* 19(2B): 371-385.
- McIntire, C.D. 1978. The distribution of estuarine diatoms along environmental gradients: a canonical correlation. *Estuarine and Coastal Marine Science* 6: 447-457.
- Midgley, D.C., Pitman, W.V. and Middleton, B.J. 1994. Surface-water resources of South Africa 1990, Water Research Commission Report No. 298/1/94.
- Miller, D.C., Geider, R.J. and Macintyre, H.L. 1996. Microphytobenthos: The ecological role of the "secret garden" of unvegetated, shallow-water marine habitats. II. Role in sediment stability and shallow-water food webs. *Estuaries* 19(2A): 202-212.
- Nusch, E.A. 1980. Comparison of different methods for chlorophyll and phaeopigment determination. *Archiv für Hydrobiologie, Beiheft Ergebnisse der Limnologie* 14: 14-36.
- Pearce, M.W. 1996. The Effect of Land Use on Gamtoos Estuary Water Quality. Ph.D. Thesis, University of Port Elizabeth, Port Elizabeth.
- 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 Scarr, N.G. 1990. The physical characteristics of the Gamtoos estuary. Institute for Coastal Research Report 23. University of Port Elizabeth: 39 pp.
- Rodriguez, F.D.G. 1994. The determination and distribution of microbenthic chlorophyll-a in selected South Cape estuaries. M.Sc. Thesis, University of Port Elizabeth, Port Elizabeth.
- Roy, S., Chanut, J.P., Gosselin, M. and Sime-Ngando, T. 1996. Characterization of phytoplankton communities in the lower St. Lawrence Estuary using HPLC-detected pigments and cell microscopy. *Marine Ecology Progress Series* 142: 55-73.
- Schlacher, T.A. and Wooldridge, T.H. 1996a. Axial zonation patterns of subtidal macrozoobenthos in the Gamtoos estuary, South Africa. *Estuaries* 19(3): 680-696.
- Schumann, E.H. and Pearce, M.W. 1997. Freshwater inflow and estuarine variability in the Gamtoos estuary: Implications for watershed management. *Estuaries* 20(1): 124-133.
- Snow, G.C., Bate, G.C. and Adams, J.B. 2000. The effects of a single artificial freshwater release into the Kromme Estuary. 2: Microalgal response. *Water SA*: 301-310.
- 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.
- Thompson, P.A. and Hosja, W. 1996. Nutrient limitation of phytoplankton in the upper Swan River estuary, Western Australia. *Australian Journal of Marine and Freshwater Research* 47:659-667.
- Thornton, D.C.O., Underwood, G.J.C. and Nedwell, D.B. 1999. Effect of illumination and emersion period on the exchange of ammonium across the estuarine sediment-water interface. *Marine Ecology Progress Series* 184: 11-20.
- Underwood, G.J.C. 1994. Seasonal and spatial variation in epipellic diatom assemblages in the Severn Estuary. *Diatom Research* 9: 451-472.
- Underwood, G.J.C. and Kromkamp, J. 1999. Primary production of phytoplankton and microphytobenthos in estuaries. *Advances in Ecological Research* 29: 93-153.
- Underwood, G.J.C., Phillips, J. and Saunders, K. 1998. Distribution of estuarine benthic diatom species along salinity and nutrient gradients. *European Journal of Phycology* 33: 173-183.
- Wehr, J.D. and Thorp, J.H. 1996. Effects of navigation dams, tributaries, and littoral zones on phytoplankton communities in the Ohio River. *Canadian Journal of Fisheries and Aquatic Sciences*. 54: 378-395.
- Whiting, M.C. and McIntire, C.D. 1985. An investigation of distributional patterns in the diatom flora of Netarts Bay, Oregon, by correspondence analysis. *Journal of Phycology* 21: 655-661.

Chapter 5: Structure and dynamics of the invertebrate component within the river-estuary interface (REI) region of the Gamtoos Estuary

T. H. Wooldridge and T.A. Schlacher
Department of Zoology, University of Port Elizabeth

General summary

1. Patterns of invertebrate species distribution across the estuarine salinity gradient generally show marked discontinuities. Pelagic and benthic assemblages of the river-estuary interface (REI) region differ significantly in their species composition and abundance from more saline estuarine reaches. Ecological communities of low salinity regions are therefore distinct in their biological structure.
2. Benthic communities of the REI are not only unique in their biological structure but display a different trophic organisation compared with more saline waters: filter-feeding forms dominate the assemblages of low salinity regions, possibly as a response to the high phytoplankton production in this region.
3. Salinity strongly influences the carbon composition of estuarine organisms. Carbon signatures (i.e. stable isotope ratios) of estuarine plants such as phytoplankton increase with increasing salinity. Similarly, carbon signatures of invertebrate consumers mirror the dependence of carbon fingerprints on habitat salinity seen in primary producers. Food web components in the REI carry distinct signatures compared with the rest of the estuary.
4. The similar and systematic variation in carbon composition across the salinity spectrum in both producers and consumer groups, suggests that production and consumption of organic matter are coupled on a local scale. Localised processing of energy through the food web within a salinity zone may override transport and mixing of organic matter between salinity zones.
5. Within the estuarine ecosystem, the REI region displays a dualism of unique species assemblages (i.e. biological structure) and largely internal energetic pathways (i.e. biological processes). This dualism clearly illustrates that low salinity regions of estuaries are unique ecological units and contribute substantially to overall ecological diversity of estuaries.

Contributions of the invertebrate component to the global hypothesis of the REI project

Hypothesis: "The structure and function of the REI region is governed primarily by the quality, quantity and supply pattern of fresh water received. Furthermore the interface region has strong influences on the physic-chemical as well as biological structure and functioning of the entire estuary".

Focus on aspect of global hypothesis	Major thrusts and sub-objectives of invertebrate module	Relevant findings and major conclusions of the invertebrate module
Structure of REI	Define REI based on species composition and abundance of benthic and pelagic invertebrates. Determine boundary conditions of the REI-communities with respect to water salinity.	Both zooplankton (pelagic) and macrofauna (benthic) assemblages have distinct structures compared with more saline reaches. The identification of boundary salinity values between communities of the REI and the estuarine reaches needs further refinement.
Function of REI	Identify the trophic organisation of assemblages of the REI. Test whether unique species composition of the REI matches differences in feeding groups.	Benthic assemblages of the REI comprise distinct functional groups compared with regions further downstream. Filter-feeding species are particularly prevalent in the REI.
Influence of REI on biological structure of the estuary	Assess structural contribution of REI to estuarine-wide assemblages	Invertebrate species groups of the REI present discrete ecological communities that contribute significantly to overall estuarine biodiversity.
Influence of REI on biological functioning of the estuary	Trace material and energy pathways within and between estuarine reaches.	Carbon signals in all compartments of the food web shift systematically from freshwater to marine habitats. Production and consumption of organic matter appear to be localised. Food web processes may be largely confined to a particular salinity zone

Contributions of the invertebrate component within the REI project to predictive capabilities

In order to improve confidence and accuracy of predictive capabilities in estuarine management, a major programme objective was to extend our understanding of estuarine dynamics and behaviour to specific estuarine processes. The REI was identified as a region with strong biological and physico-chemical links to both the freshwater and estuarine domain. However, key processes in this region, particularly those related to freshwater inflow and its connection with estuarine reaches downstream, were poorly known and impeded the development of more accurate predictive capabilities in estuarine management. Consequently, a set of five key-questions - linked to predictive capabilities' needs - was developed for the original proposal. The contribution of the invertebrate component is as follows:

Key question	Contribution by this component
1. What are the spatial and temporal ranges and patterns of the interface region in relation to inflow amounts and quality of freshwater?	Because the seaward boundary of animal assemblages typical of the interface region is closely linked to low salinity (< 10 ‰) conditions, variations in freshwater inflow shift the range of these communities along the main axis of an estuary.
2. What is the composition of the freshwater-estuarine interface community and does it differ from the rest of the estuary?	Pelagic and benthic animal assemblages of the interface region differ significantly in species composition and abundance from other estuarine regions.
3. What is the response of the community to freshettes and what is the duration of this response?	Estuarine zooplankton responds rapidly to freshettes by marked increases in population densities, but this response is generally of short duration (i.e. weeks).
4. Does the interface community contribute significantly to vital estuarine processes?	Food-web processes appear to be largely confined to specific salinity zones. In conjunction with the distinct nature of the interface community (in terms of species composition), the localised pattern of energy flow indicates that the REI region is not a major exporter to the rest of the estuary.
5. Is the interface community ephemeral or permanent?	Under normal flow conditions, the distinct nature of the interface community is always evident. Extreme habitat changes during above-average flow events may, however, temporarily reduce the diversity of estuarine assemblages, including the interface community.

Contributions of the invertebrate component within the REI project to decision support for the conservation and management of estuaries

In keeping with the overall programme objective of "...further development of decision support systems for estuaries...", the invertebrate component contributed significantly to three key-elements of estuarine decision support systems as outlined below.

The emphasis on advancing our understanding of principles that underpin ecological structure and function in estuaries is particularly evident from findings that stress the unique nature of the habitats and communities in the interface region. In addition, specific new process information (e.g. close coupling of organic matter production and consumption on a local scale, and energetic importance of phytoplankton over macrophytes) can now be linked to structural data to reinforce the concept that the REI region is an integral part of overall estuarine system diversity.

5.1 Faunistic attributes and determinants of the REI region - macrobenthos and zooplankton

5.1.1 Summary

1. The Gamtoos Estuary has considerable interannual variability in river inflow, but a common trend of increasingly lower salinity values was evident for the last six to seven years, resulting in a progressively seaward expanding zone of freshwater at the head of the estuary.
2. Both zooplankton and macrobenthos showed definite density clines from the estuary mouth to the head, but in opposite directions. While the planktonic component decreased in abundance towards the upper reaches, highest densities of macrobenthos were attained in the interface region.
3. Similarly, clines in biotic diversity were pronounced for both pelagic and benthic invertebrates but in this case were symmetrical, with an appreciable decline in species richness and diversity from the mouth to the freshwater regions for both groups.
4. The distribution of subtidal macrobenthos across the salinity gradient was strongly disparate with marked faunistic boundaries giving rise to definite zones along the main axis of the system. The most landward-located faunal zone was always found in salinities < 10.0 ‰.
5. Longitudinal macrofaunal assemblages were relatively invariant over time with the distinct community structure of the REI region always being apparent. The fauna of this region was on the one hand dominated by species normally affiliated with riverine habitats (e.g. chironomids) and on the other hand by a few extremely euryhaline estuarine taxa.
6. Zooplankton distribution followed an essentially similar pattern to that of the benthos with:
 - a. Distinct faunal discontinuities giving rise to marked zonation patterns across the salinity gradient.
 - b. A strong consistency to this pattern over time, with a distinct community structure being evident for the waters of the interface region.
7. Besides a strong underlying gradient of faunal distribution and diversity across the salinity spectrum, community structure remained distinct for the REI region for at least two biota (zooplankton and zoobenthos) - it also appears to be relatively invariant over time.

5.1.2 Introduction

On a landscape scale, estuaries are among the globe's most striking ecotones, forming the major transition zones between fresh water biomes and the oceans. Scaling spatial analysis down from the landscape level to individual estuaries reveals the prominence of a further two salient transition zones: on the one hand, the river ecosystems grade into the estuary itself, while on the other hand the most seaward sections of the estuary form a dynamic link with adjacent marine systems. Dissecting patterns of physico-chemical parameters within an estuarine ecosystem even further eventually exposes more distinct habitat zones and associated interface regions. Each of these zones may be

characterised by a typical subset of species, and the identification of such distinct assemblages may indeed be used to advance an approach where estuarine zonation schemes are based on biological grounds (Bulger *et al.* 1993; Schlacher and Wooldridge 1996b). Such discontinuous zones of different plant and animal assemblages, each occupying a particular stretch of the estuary, are well developed in South Africa's estuaries (Schlacher and Wooldridge 1996a).

Among the various levels of spatial patterning evident for biological communities within estuaries (Bulger *et al.* 1993), the transition zone between fresh- and saltwater, forms one of the most conspicuous interfaces. The salient features of this section are very steep gradients of salinity, with the amount of freshwater inflow structuring the REI most decisively. In the southern African subregion fresh water is a commodity in extremely short supply and this has already resulted in a severe curtailment of freshwater inputs to local estuaries thus resulting in a number of negative ecological consequences (Schlacher and Wooldridge 1996b).

In this study we set out to:

- a. Determine the biological boundaries of the upper reaches of estuaries where fresh- and saltwater interface most strongly.
- b. Determine the broad structural properties of the invertebrate communities inhabiting these transition zones.
- c. Estimate temporal and spatial variability in community structure for assemblages typical of the REI.

5.1.3. Methods

5.1.3.1 Study area

All field work for the present study was conducted in the permanently open Gamtoos Estuary. The mouth region is shallow (< 1.5 m) and narrow (< 50.0 m), and for the first 1.5 km upstream an extensive flood tide delta has developed. Beyond the flood tide delta, the main channel deepens to ~ 4.0 m in the middle reaches, but upstream of 8.0 km the estuary becomes progressively narrower (< 100.0 m) and shallower (< 2.5 m). This trend continues all the way to the tidal head (20.0 km), where water depths are usually < 1.0 m. A 3.0 km long, shallow channel, which represents an abandoned tidal inlet, joins the main estuary 1.4 km from the mouth. This arm is hereafter referred to as "Gamtoos lagoon", and is shallow (< 1.8 m) with a maximum width of ~ 200.0 m (Heineken 1981; Reddering and Scarr 1990).

The sediment deposits in the lower estuary are predominantly marine, receiving sand from tidal action, longshore currents, riverine floods and aeolian transport from a coastal dune field. By contrast, sediments in the middle and upper reaches are of purely fluvial origin, with flocculation of the freshwater mud suspension occurring mainly in the first 8.0 km upstream from the mouth area. Consequently, upstream of the flood tidal delta sediment grain size decreases rapidly. These lower reaches are dominated by fine to very fine fluvial muds, after which there is a gradual coarsening of the sand fraction towards the upper estuary (Reddering and Esterhuysen 1984).

Three major dams are situated in the catchment (34 450 km²) of the Gamtoos Estuary, and the total storage capacity of these reservoirs (249 x 10⁶ m³) is equivalent to half the mean annual runoff (501 x 10⁶ m³; Jezewski and Roberts 1986). The tidal reaches of the estuary extend over approximately 20 km, though the exact position of the tidal head shifts several kilometers, depending on river scouring and the amount of fresh water entering the estuary (see Chapter 2).

5.1.3.2 Invertebrate sampling

5.1.3.2.1 Macrobenthos

Benthic macrofauna was sampled at 12 sites positioned at approximately 2.0 km intervals along the axis of the estuary. Sampling spanned the entire estuary from the mouth to the tidal head (for a map showing sampling localities see Schlacher and Wooldridge 1996a). The site closest to the mouth represents the furthest point safely navigable in a seaward direction, and similarly, the furthest upstream site represents the landward limit of navigable waters. Numbering of sites follows a previous study by T.H. Wooldridge on plankton distribution. In addition to Wooldridge's eight plankton sites, five additional sites were sampled for macrofauna in this study. Site 1.5 was situated in the muddy, blind arm (i.e. lagoon) adjacent to the sandy flood tide delta, sites 5.5 and 6.5 were located in the transition zone between the muddy and the sandy sediments of the middle reaches and sites 9 and 10 covered the tidal reaches of the upper estuary close to the tidal head.

For the purpose of this programme a benthos survey was conducted at the sites described above on 4 November 1996. Data obtained in this sampling series are then compared to earlier studies (same sampling design) conducted on 23/24 June 1993 and 25 February 1994. Macrobenthos sampling in this programme focused on the subtidal fauna for three main reasons: first, since the intertidal area is poorly developed in the Gamtoos (Reddering and Scarr 1990), most of the benthic habitat lies subtidally. Secondly, significant anthropogenic disturbances in the intertidal zone (e.g. intensive bait-collecting) impact on benthic assemblages and may confound true longitudinal zonation patterns. Thirdly, short-term environmental disturbances (e.g. rainfall and temperature extremes) are higher in intertidal habitats. Since our primary objective was to assess the broad, prevailing zonation of the macrofauna, such small-scale disturbances can be expected to affect the distribution of subtidal organisms less when compared to that of the intertidal fauna.

At each site three random benthic samples were collected with a van Veen grab (211 cm² sampling area), operated by hand from a small boat. To reduce effects on macrofauna density related to water depth, at each site the cross-channel position of the sampling station was adjusted so that grab samples were taken between a depth of 1.5 to 2.0 m. Whole grab samples were immediately preserved in 10.0 % formalin containing Rose Bengal dye. Thalassinid prawns, which burrow up to 1.0 m into the sediment were not effectively sampled with the van Veen grab, and this group is therefore omitted from the numerical analysis. At each site temperature and salinity profiles of the water column were measured at 0.5 m depth intervals with a Valeport CTD meter.

In the laboratory, macrobenthos was extracted from sediment samples by repeated decantation and all individuals retained on a mesh of 0.25 mm aperture size were enumerated to the nearest identifiable taxon. A sieve mesh size of 0.25mm is the coarsest mesh size that can be used in

this particular habitat if faunal density is to be determined with any reasonable level of accuracy (Schlacher and Wooldridge 1996c).

5.1.3.2.2 Zooplankton

On 4 November 1996 zooplankton was sampled two hours after sunset over sites 1 to 8, where macrobenthos grabs were taken during daylight hours. The sampling protocol is detailed in Schlacher and Wooldridge (1995). In brief, WP2-nets with a mesh size of 0.2mm and fitted with Khalisco flow-metres were used to sample the water-column. Two nets were attached to booms protruding laterally from the bow region of a small ski-boat; one net sampled just below the water surface, while the bottom-clearance of the second net was adjusted to approximately 20 cm above the substrate. In the laboratory the number of plankters per tow was determined from 3 to 6 subsamples and final counts then converted to individuals per m^3 using the flow-metre data. As was the case for macrobenthos, these data were compared against a medium-term data series recorded at the same sites at monthly intervals from February 1989 to March 1991, thus yielding 26 density estimates per site and species.

5.1.3.3 Data analysis

Univariate measures of community structure centred around species abundance relationships in the samples and included a) Hill's numbers N_0 and N_1 which are intrinsic indices of richness (N_0) and diversity (N_1) (Hill 1973), and b) total abundance of individuals per site. Hill's numbers N_0 and N_1 coincide with two of the most commonly employed diversity indices, i.e. $N_0 = S$, where S is species richness and $N_1 = e^{H'}$, where H' is the popular Shannon-Wiener index. Thus, N_0 gives the total number of species in the sample and N_1 the number of "abundant" species. When compared to the classical indices (e.g. I and H') they have the advantage of being in units of species and thus are generally easier to interpret (Ludwig and Reynolds 1988).

Multivariate analyses of community structure followed the procedure outlined by Field *et al.* (1982) and detailed in Clarke (1993). In brief, ranked similarity matrices were constructed using the Bray Curtis resemblance function. To prevent "swamping" of the community composition by the most abundant species, data were fourth-root transformed to increase the contribution of the less common species to inter-site similarity. For both grouping of biotically similar sites (normal or q-type analysis) and for grouping of co-occurring species (inverse or r-type analysis) two complementary methods were used, namely group average clustering, applied to both sites and species data as the classification technique of choice, and non-metric multidimensional scaling (Clarke and Green 1988) which was employed to ordinate sites and species.

The ANOSIM procedure (Clarke and Green 1988) was used to test for multivariate differences between *a priori* defined groups. In the present study this was applied to compare sampling years for macrobenthos and to test for differences in zooplankton communities between waters split at a salinity of 10.0 ‰. Prior to inverse analysis of species, rare taxa (< 1.0 % of relative density at any one site) were omitted to avoid spurious groupings of species (Field *et al.* 1982).

5.1.4 Results

5.1.4.1 Physico-chemical characteristics - salinity regimes

A steep salinity gradient along the main axis of the estuary was developed during all sampling series, ranging from near full-strength seawater near the mouth to freshwater at the most landward sites (Figure 5.1.1). A significant feature was, however, that the freshwater wedge moved progressively seawards over time: in 1993 the landward limit of seawater intrusion was located 16.7 km (site 8) from the tidal inlet (Figure 5.1.1 A), but in subsequent years fresh water penetrated a further 2.5 km downstream, now also encompassing site 7 (Figure 5.1.1 B and C). Thus, there appears to be a definite trend towards a greater spatial range of freshwater habitat at the head of the estuary, as well as an overall decrease in salinity for the estuary as a whole. Interestingly, there was very little gradation between the saline and fresh waters across the REI region. A sharp longitudinal halocline delineated the brackish from the saline reaches, with bottom salinities dropping from 12.5 to 24.6 ‰ to 0.1 to 0.5 ‰ over a distance of less than 2.5 kms.

5.1.4.2 Longitudinal gradients in invertebrate density and diversity

Distinct clines of abundance were evident for both invertebrate components studied, but the direction of density change was in opposite directions for zooplankton and macrobenthos. Whereas pelagic invertebrates tended to decline from the tidal inlet to the freshwater section ($r_s = -0.31$, $p = 0.25$), density of benthic macrofauna was elevated with increasing distance from the sea ($r_s = 0.49$, $p = 0.09$), attaining the highest abundance in waters of near zero salinity (Figure 5.1.2 A).

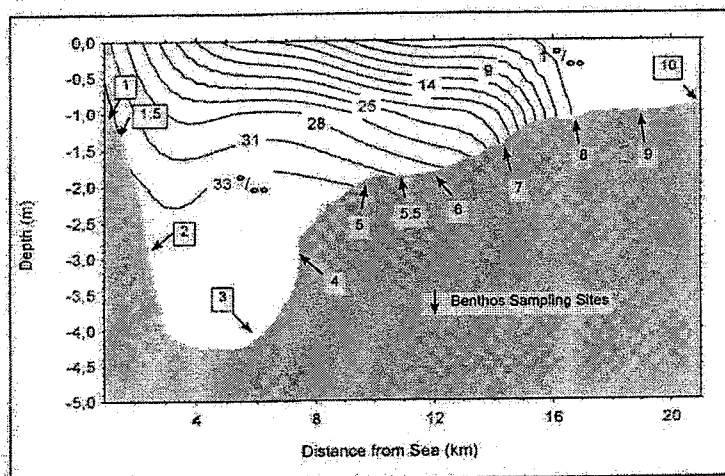
By contrast, species richness showed a similar longitudinal gradient for both groups, with the number of taxa significantly decreasing towards the landward limit of tidal influence (plankton $r_s = -0.74$, $p < 0.01$; benthos $r_s = -0.70$, $p = 0.02$; Figure 5.1.2 B). No such trend was evident for diversity, with no clear relationship between distance from the sea and any univariate measure of diversity (max. $r_s = 0.38$, $p = 0.20$; Figure 5.1.2 C).

5.1.4.3 Longitudinal patterns of invertebrate community structure

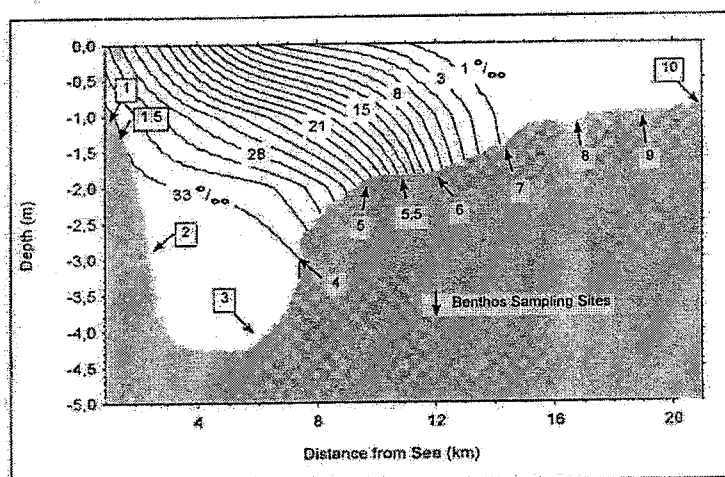
Although community structure of benthic macrofauna showed an overall gradual change from the regions bordering the ocean to substrates being dominated by freshwater inflow, two patterns of zonation stand out (Figure 5.1.3):

1. Sediments near the tidal inlet (sites 1 and 1.5) harbour assemblages distinctly different from the fauna of the lower estuarine reaches. The boundary of this tidal inlet region is located approximately 1.5 km from the estuary mouth.
2. On the opposite end of the salinity range, another distinct assemblage is restricted to the REI region (sites 7 to 10), the seaward boundary of this zone shifting between 14.0 and 17.0 km, with salinities always < 0.5 ‰. In the mesohaline middle reaches of the estuary (sites 2 to 6.5) the macrofaunal zonation is more variable, but clear faunistic discontinuities delineate it (on both ends) from both the tidal inlet section and the river-estuary interface region.

1993



1994



1996

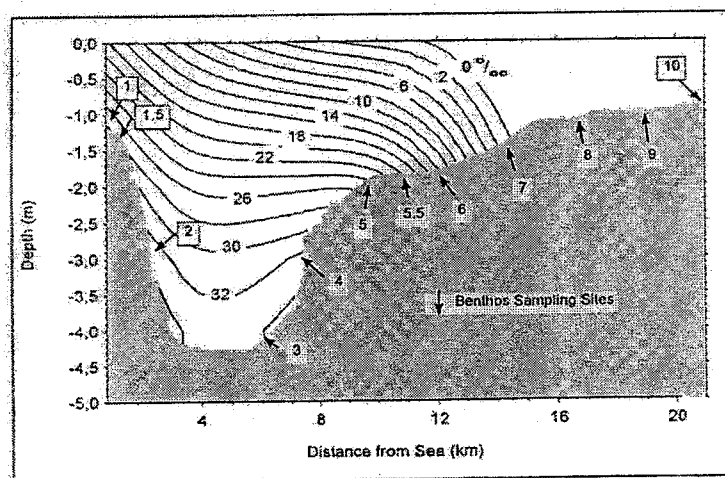


Figure 5.1.1: Profiles of salinity distribution in the water column along the main axis of the Gamtoos Estuary recorded during macrobenthos surveys on 23/25 June 1993 (top), 25 February 1994 (middle), and 4 November 1996 (bottom). Arrows and numbers indicate longitudinal position of sampling sites.

No single species was found to be a perfect discriminator between the assemblage typical of the river-estuary interface region and the rest of the estuary (Table 5.1.1), but the amphipod *Corophium trianonyx*, the polychaete *Desdemona ornata* and Oligochaeta each contribute more than 8.0 % to average dissimilarity between these two broad zones.

Similarly, the same taxa (i.e. *Desdemona ornata*, *Corophium trianonyx* and Oligochaeta) contribute most to intra-group similarity (Table 5.1.2) and can therefore be regarded as being typical of this assemblage. Other species indicative of the REI community include the polychaete *Ceratonereis keiskama*, and Chironomids (Table 5.1.2). The fauna of the REI region appears therefore to be comprised of taxa normally affiliated with fluvial habitats (e.g. chironomids and oligochaetes) and a few estuarine taxa of exceptionally high physiological tolerance to low salinities (e.g. corophioid amphipods, some polychaetes).

Having extracted the existence of a sufficiently unique macrobenthos assemblage inhabiting the REI region, the question arises whether this community is of an ephemeral or permanent nature. Irrespective of sampling period, the same broad pattern of macrofaunal zonation along the salinity gradients is developed (minimum r_s between similarity matrices among years = 0,70***; see insert in Figure 5.1.4), the distinct assemblages of the REI region being evident not only in each yearly plot (Figure 5.1.3) but also clearly clustering out as a definite group in an ordination of the combined data set (Figure 5.1.4). Thus, the basic nature of the macrobenthos assemblage in the REI zone is fairly invariant over time. The most marked changes of this particular subset of the estuarine fauna affect the spatial range over which it extends, the seaward boundary clearly depending on the amount of freshwater inflow.

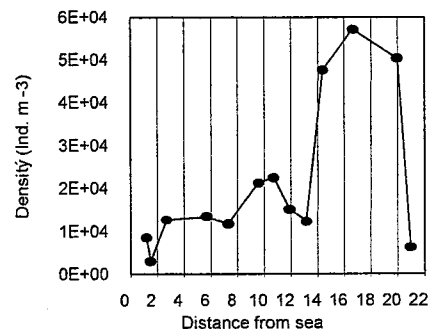
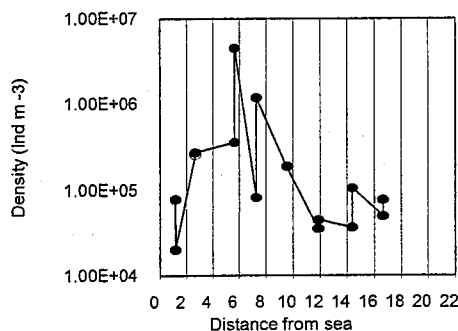
Another line of evidence for the persistence of distinct changes in community structure with salinity can be derived from the degree to which such changes follow a linear sequence (here taken as a progression from the mouth region to the head of the estuary). Although salinity values in the estuary tended to decrease appreciably from 1993 to 1996, the essential sequence of community changes across the salinity spectrum for each particular sampling period persisted over time (Figure 5.1.5). A gradual drop in the index of multivariate seriation (1993 - 0.85; 1994 - 0.81; 1996 - 0.66) may indicate that increased freshwater inflow over longer time periods acts as a disturbance agent, which eventually leads to some breakdown in seriation (*sensu* Clarke *et al.* 1993).

Zooplankton community structure across the estuarine salinity gradient follows an essentially similar pattern to that of the macrobenthos (Figure 5.1.6). Again, zooplankton assemblages at both the tidal inlet region (site 1) and the REI region (sites 7 and 8) are distinct compared with the communities in the polyhaline waters of the middle section.

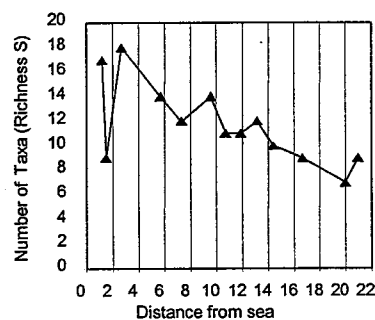
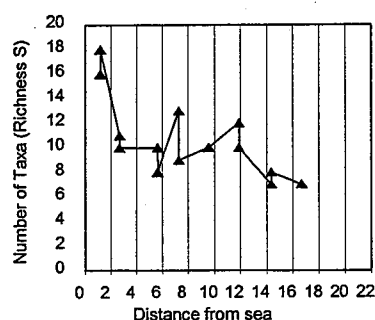
Zooplankton

Macrobenthos

Total Density



Species Richness



Species Diversity

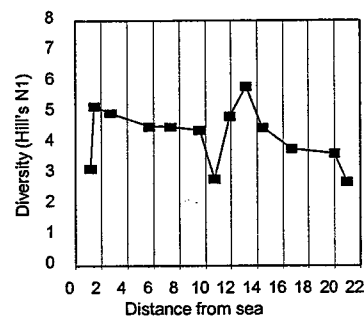
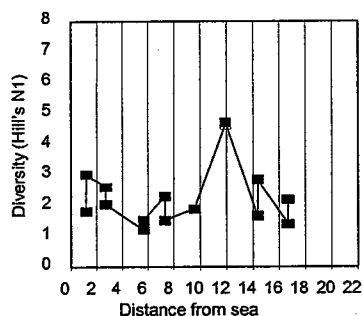


Figure 5.1.2: Longitudinal trends in total abundance (A), richness (B), and diversity (C) of zooplankton (left column) and macrobenthos (right column) recorded in the Gamtoos Estuary during the November 1996 survey. Measure of abundance is the mean ($n = 3$) number of all individuals recorded per site. Index of biotic richness is the total number of identifiable taxa found per site. Diversity measure is Hill's number $N1$ ($N1 = e^{H'}$, where H' is the Shannon Wiener Index); it roughly gives the number of abundant taxa present.

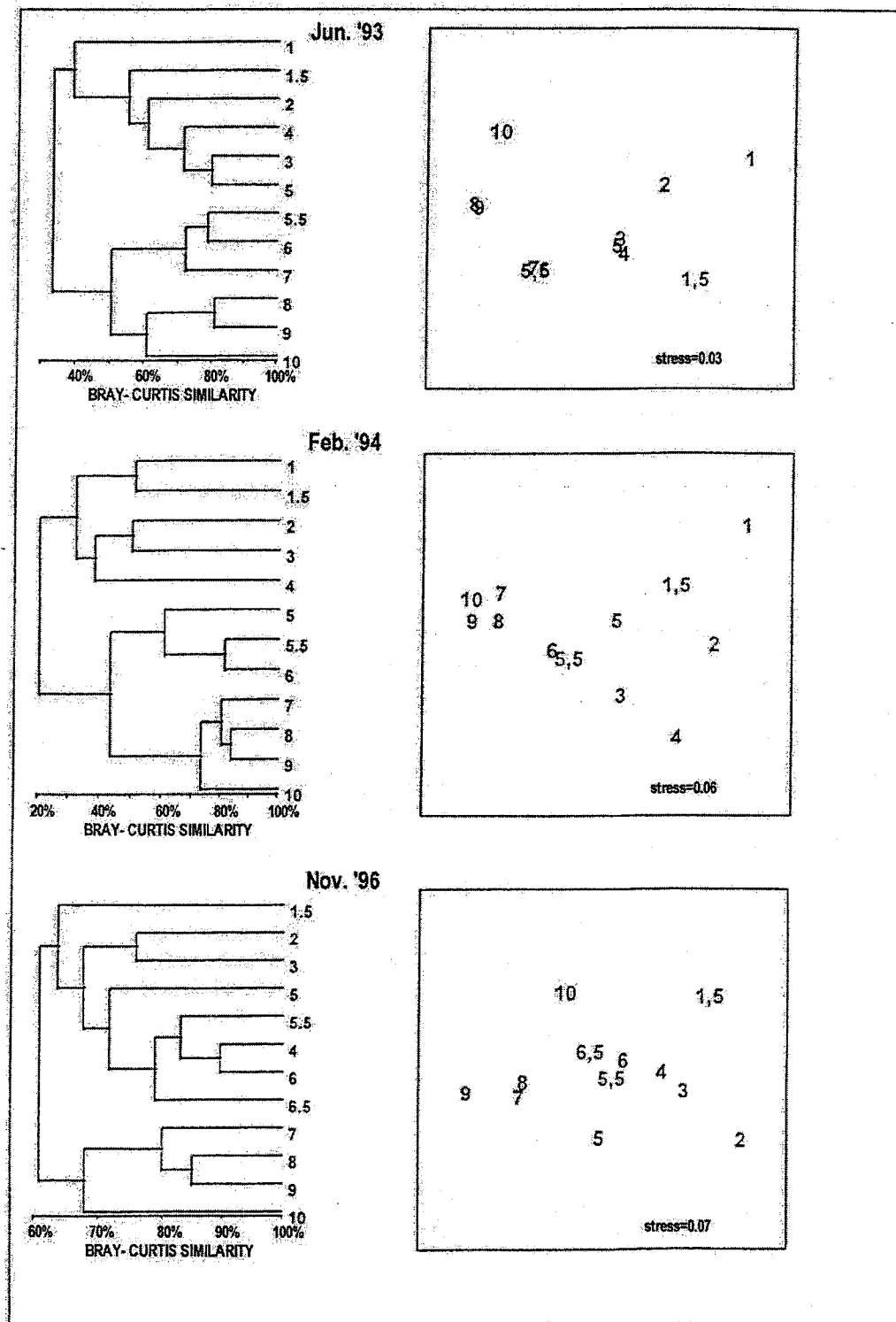


Figure 5.1.3: Grouping of sites similar in macrobenthos community structure by means of group average clustering (left column) and non-metric multidimensional scaling (NMDS - right column) for the three periods in which benthic invertebrate data were recorded. Site 1 is situated closest to the tidal inlet region, while site 10 represents the furthest upstream located habitat .

Table 5.1.1: Breakdown of average dissimilarity (adi) of macrobenthos between the freshwater interface region (sites 7 to 10) and all reaches further downstream into contributions by particular species as well as average abundances of species in each of the two regions.

	Interface region	Seaward reaches	Contribution to average dissimilarity by individual species			
	Mean density (Ind. m ⁻²)	Mean density (Ind. m ⁻²)	mean di	mean/SDdi s(di) ⁻¹	% adi %	cum. %cum adi %
<i>Corophium trianonyx</i>	9,759	696	4.2	1.97	9,2	9.2
<i>Desdemona ornata</i>	9,124	4,928	3.9	1.46	8,6	17.8
<i>Oligochaetes</i>	4,303	597	3.9	1.77	8,5	26.3
<i>Prionospio spp.</i>	2,140	2,413	3.6	1.43	7,9	34.3
<i>Capitella capitata</i>	0	670	3.5	1.53	7,8	42.0
<i>Xenanthura sp.</i>	67	870	3.1	1.51	6,7	48.8
<i>Prionospio sexoculata</i>	0	177	2.9	1.98	6,4	55.2
<i>Iphionoe truncata</i>	186	819	2.6	1.35	5,7	60.9
Chironomids	438	38	2.4	1.76	5,2	66.1
<i>Grandidierella lignorum</i>	3,421	2,570	2.1	1.07	4,7	70.8
<i>Ceratonereis keiskama</i>	987	146	2	1.35	4,5	75.3
<i>Spionid spp.</i>	0	67	1.9	1.51	4,1	79.5
Hirudinea	12	37	1.3	1.5	2,9	82.4
<i>Afrochiltona capensis</i>	12	7	1	1.02	2,2	84.6
<i>Cossura sp.</i>	0	139	0.9	0.51	1,9	86.5
<i>Lumbrinereis sp.</i>	0	19	0.6	0.51	1,3	87.8
<i>Donax serra</i>	0	77	0.5	0.35	1,2	89

Table 5.1.2: Breakdown of average similarity within the freshwater interface region (sites 7 to 10) into contributions by particular species as well as average abundances of species in the region.

	Interface Region	Contribution to average similarity by Individual species			
	Mean density (Ind. m ⁻²)	meanSi	mean/SDSi s(Si) ⁻¹	% aSi %	cum. %cum aSi %
<i>Desdemona ornata</i>	19,124	18.5	8.37	25.2	25.2
<i>Corophium triaenonyx</i>	9,759	12.7	2.48	17.2	42.4
<i>Grandidierella lignorum</i>	3,421	11.2	6.08	15.3	57.7
<i>Oligochaetes</i>	4,303	9.4	4.16	12.7	70.4
<i>Ceratonereis keiskama</i>	987	8.1	6.92	11	81.5
Chironomids	438	7.7	5.59	10.5	92.0
<i>Prionospio spp.</i>	2,140	2.8	0.79	3.8	95.8
Hirudinea	12	1.9	0.9	2.5	98.3
<i>Iphionoe truncata</i>	186	0.6	0.41	0.9	99.2
<i>Afrochiltona capensis</i>	12	0.6	0.41	0.8	100

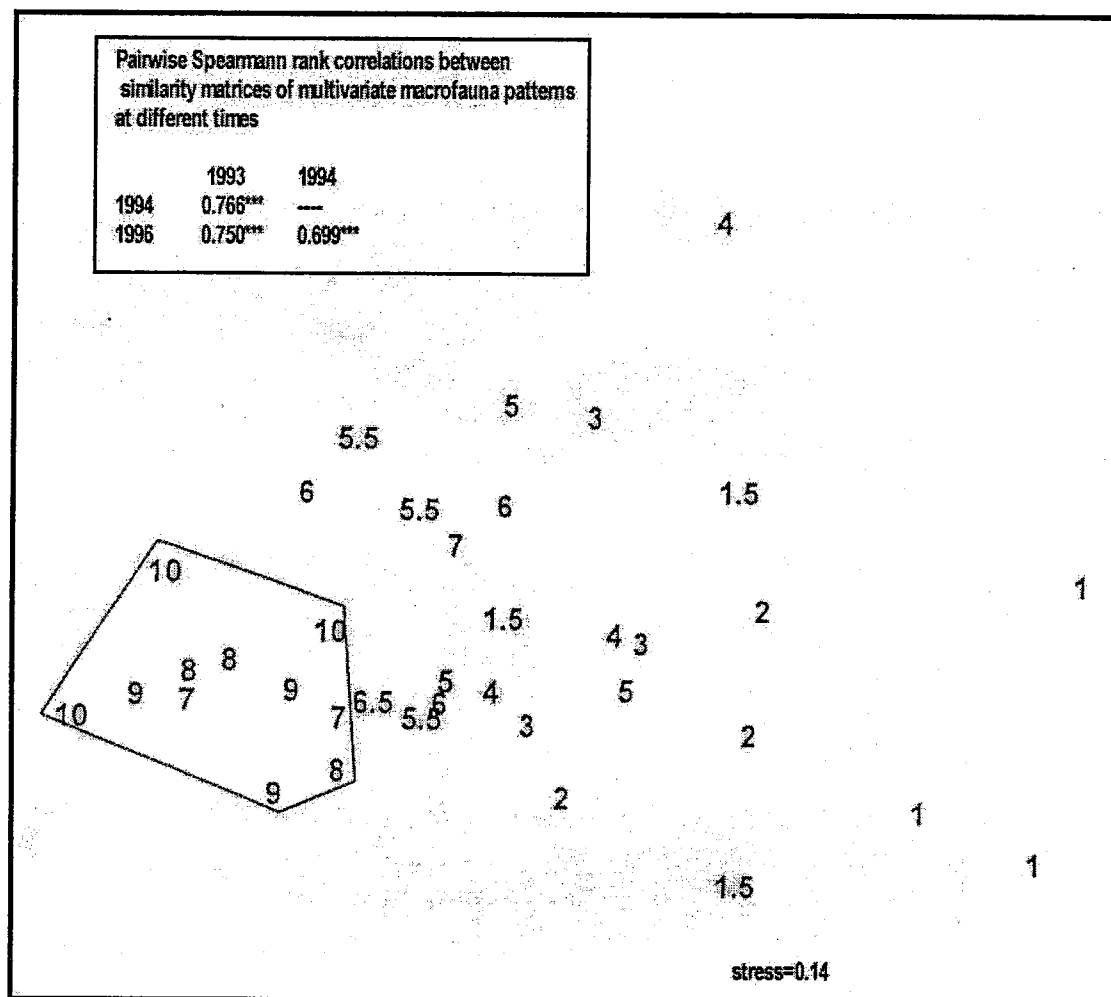


Figure 5.1.4: Ordination of macrobenthos community data recorded over three years in the Gamtoos Estuary at each of 12 sites. Sampling included the entire salinity gradient from predominately marine waters bordering the mouth (site 1) to habitats dominated by riverine freshwater input at the head of the estuary (site 10). Correlation coefficients in the top right insert denote the high degree of agreement in zonation patterns across sampling times. Note that all sites encircled by the shaded area to the left represent distinct communities of the REI region (*cf.* Figure 5.1.3), defined here purely on the grounds of similarity in biotic composition and abundance.

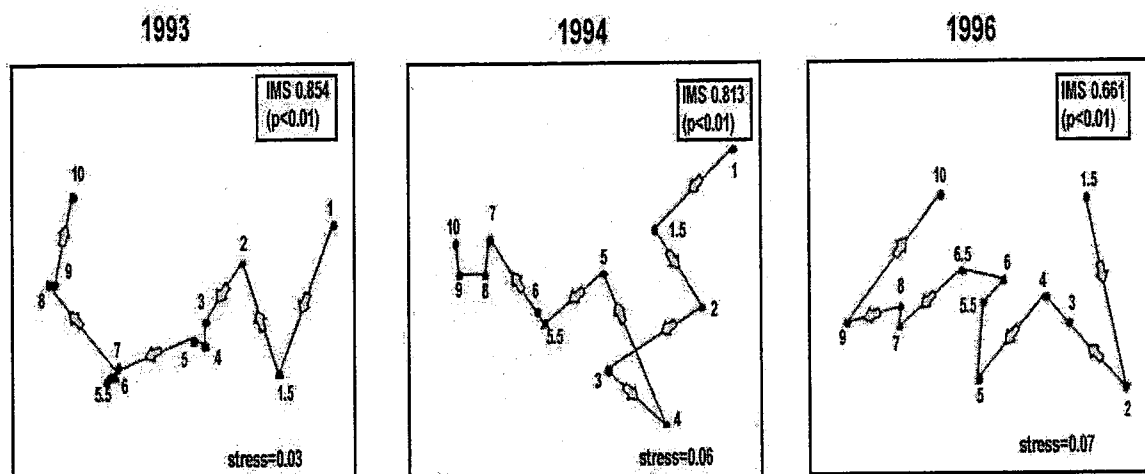


Figure 5.1.5: Interannual comparison of the degree of sequential (linear) changes in macrobenthic community structure (i.e. seriation) from the mouth region to the REI zone. *IMS* stands for Index of Multivariate Zonation and essentially gives an indication of how closely community changes follows a linear sequence (which in our case is physically evident following the estuary from the mouth to the head; Clarke *et al.* 1993). Our specific hypothesis for this analysis was that freshwater inflow represents a disturbance event which could lead to a breakdown in seriation.

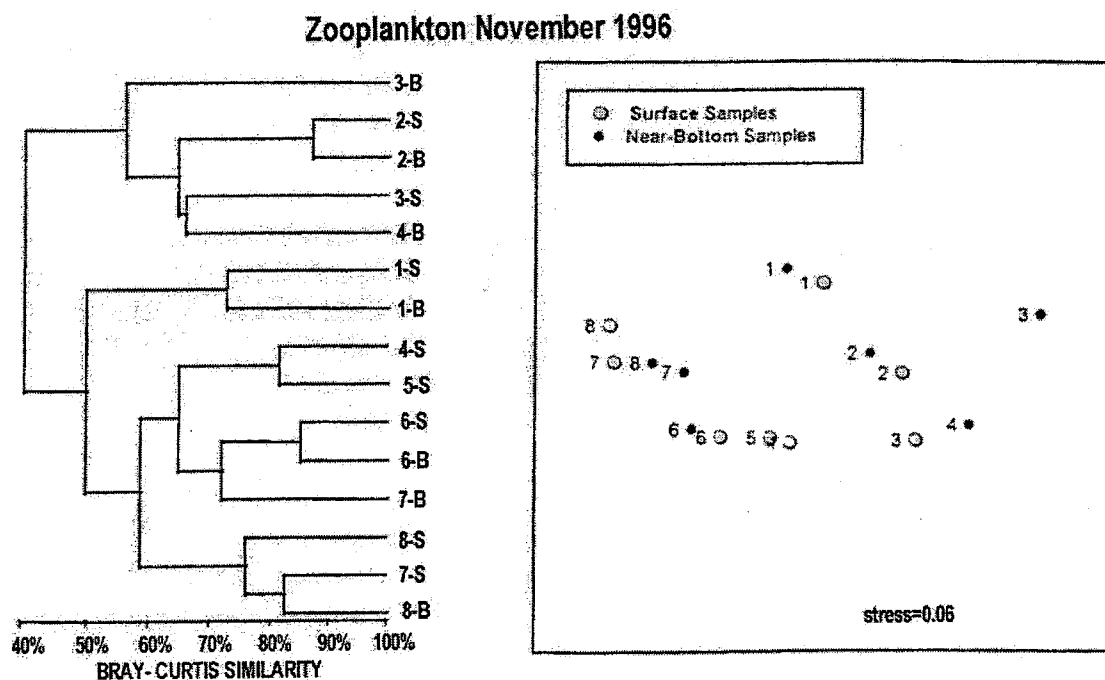


Figure 5.1.6: Classification and ordination of zooplankton distribution patterns in the Gamtoos Estuary during November 1996. Note that patterns closely follow that of the macrobenthos (cf. Figure 5.1.3).

Table 5.1.3: Comparison of macrobenthos density recorded in this study with other South African estuaries. Only data sets which used comparable sieve mesh sizes (< 1.0 mm) for collection were included. Abbreviations for statistics extracted from the literature are: RSM - range of seasonal means; RMS - range of means per site; Max - maximum recorded density for the estuary.

	Ind. m ⁻²	Stat.	Locality	Salinity	Source
Total benthic density	4967 - 5,915	RSM	Bot River	21 - 23	De Decker and Bally 1985
	2020 - 9,200	RMS	Bot River	7 - 12	Koop <i>et al.</i> 1983
	7445 - 11,579	RMS	Lake St Lucia	31 - 45	Blaber <i>et al.</i> 1983
	394 - 57,235	RMS	Gamtoos	0 - 34	This study
	17322 - 89,416	RMS	Berg River		Kalejta and Hockey 1991
	52894 - 91,968	RMS	Gamtoos	0 - 18	Schlacher and Wooldrige 1996c ¹
<i>Grandidierella lutosa</i>	8 - 657	RMS	Berg River		Kalejta and Hockey 1991
<i>Grandidierella lignorum</i>	85 - 1,488	RMS	Lake St Lucia	31 - 45	Blaber <i>et al.</i> 1983
	c. 5,000	Max	Lake Nhlanga	3 - 5	Bolt and Allanson 1975
	173 - 5,169	RSM	Lake Sibayi	0	Bolt 1969
	16 - 19,070	RMS	Gamtoos	0 - 34	This study
	11364 - 31,357	RMS	Gamtoos	0 - 18	Schlacher and Wooldrige 1996c ¹
<i>Grandidierella bonnieroides</i>	c. 100,000	Max	Palmiet	0 - 35	Branch and Day 1984

Table 5.1.3 continued

	Ind. m ⁻²	Stat.	Locality	Salinity	Source
<i>Ceratonereis keiskama</i>	284	Max	Lake Sibayi	0	Bolt 1969
	16 - 3,871	RMS	Gamtoos	0 - 34	This study
	c. 5,000	Max	Palmiet	0 - 35	Branch and Day 1984
	1347 - 5,403	RMS	Berg River		Kalejta and Hockey 1991
	7401 - 14,311	RMS	Gamtoos	0 - 18	Schlacher and Wooldrige 1996c ¹
<i>Desdemona ornata</i>	2 - 9	RMS	Berg River		Kalejta and Hockey 1991
	3 - 45	RMS	Bot River	21 - 23	Decker and Bally 1985
	1894 - 6,910	RMS	Gamtoos	0 - 18	Schlacher and Wooldrige 1996c ¹
	16 - 28,025	RMS	Gamtoos	0 - 34	This study
	c. 50	Max.	Palmiet	0 - 35	Branch and Day 1984
<i>Corophium triaenonyx</i>	45 - 439	RMS	Bot River	21 - 23	Decker and Bally 1985
	0 - 180	RMS	Bot River	7 - 12	Koop et al. 1983
	421 - 1,403	RMS	Gamtoos	0 - 18	Schlacher and Wooldrige 1996c ¹
	93 - 1,680	RSM	Lake Sibayi	0	Bolt 1969
	c. 5,000	Max	Lake Nhlange	3 - 5	Bolt and Allanson 1975
	16 - 14,487	RMS	Gamtoos	0 - 34	This study

¹ Nine localities situated between sites 7 and 10 sampled during December 1991, with salinity values ranging between 0 and 18.0 ‰.

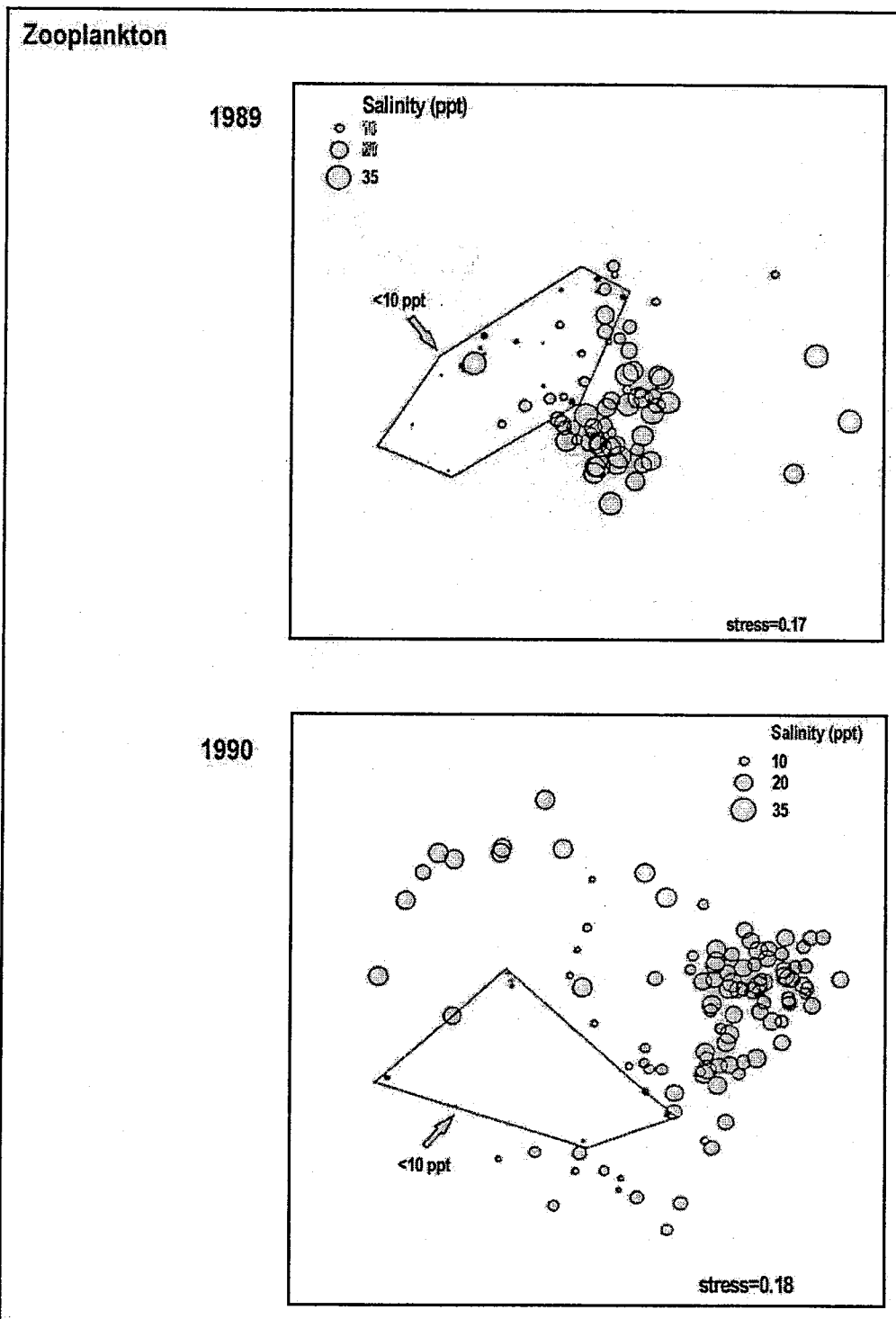


Figure 5.1.7: NMDS plots of zooplankton community data sampled during 1989 and 1990 once every month in surface and near-bottom waters at each of eight axially located sites in the Gamtoos Estuary. Each point in the plots represents an individual site at a particular sampling datum; scaled salinity values are superimposed on site positions.

5.1.5 Discussion

Direct comparisons of macrobenthos density with other systems of the region are at best difficult because sampling protocols vary widely. The most important flaw is the severe bias mesh aperture sizes > 0.25 mm cause for density estimates. We have shown that in the present sampling situation ca. 92.0 % of all individuals are lost from the samples if a 1.0 mm mesh is used for extraction (Schlacher and Wooldridge 1996d). Yet, even mesh sizes of 3.0 mm aperture size have in the past been used to 'quantify' the macrobenthos (e.g. Hanekom *et al.* 1988) and we therefore restrict our comparison to data series obtained with mesh sizes < 1.0 mm. Keeping these limitations in mind, the present analysis shows that the Gamtoos Estuary supports very high abundances of macrobenthos, with most species ranking consistently high in comparison with other South African systems, e.g. the suspension feeding polychaete *Desdemona ornata* reaches densities that are one to two orders of magnitude greater than in other estuaries. Interestingly, this species is most abundant in the REI region and is both a good discriminating and indicator species for this region (Table 2 and 3).

Traditionally, the Venice system (see Glossary) sets the boundary between fresh and saline waters at a salinity of 0.5 ‰. By contrast, a more recent analysis of salinity zones based on the distributional ranges of species in North American estuaries, identified a limit of 4.0 ‰ for the lowest salinity zone (Bulger *et al.* 1993). The initial working definition of this programme set the boundary of the REI region at a vertically averaged salinity of 10.0 ‰. To what extent this value corresponds to the distributional limits of the distinct faunal assemblages identified in the present study is thus a pertinent question. The macrobenthic community of the REI region was always located in salinities between 0.5 and 12.5 ‰; thus the initial working definition of 10.0 ‰ appears to hold reasonably well for the macrobenthos data. It is, however, not unlikely that small-scale sampling within this salinity range (0.5 to 12.5 ‰) could reveal an intermediate salinity value, which defines the seaward limit over which the community of the REI region ranges. Such sampling designs at a smaller spatial scale (ca. 50 to 500 m) are therefore advocated for further research in this direction.

We can also advance a second line of argument for the appropriateness of the upper 10.0 ‰ boundary condition, i.e. the analysis of a medium-term (26 months) data set of zooplankton community structure recorded in 1989 and 1990 across the salinity spectrum of the Gamtoos estuary. For both years analysed, habitats of salinity values < 10.0 ‰ harboured significantly different zooplankton communities compared with more saline waters (Figure 5.1.7; ANOSIM-test: 1989 Global $R = 0.464$ $p < 0.001$; 1990 Global $R = 0.453$ $p = 0.002$). It would thus appear, that the initial working definition of 10.0 ‰ does indeed provide an adequate indication of the upper limit of the REI region for the Gamtoos system.

The data presented above shows that invertebrate communities inhabiting the REI region differ significantly in structure, abundance and diversity from assemblages of more saline habitats, and that this pattern remains stable over time. Thus, strong evidence exists for at least two biota (macrobenthos and zooplankton), that the REI region represents not only a physico-chemical well defined habitat but also a biologically distinct section of the estuary.

5.2 Integrating functional aspects and food-web properties of the REI region

5.2.1 Summary

Two key areas were targeted during this study:

- Application of functional group approaches to the multivariate delineation of benthic invertebrate communities between the REI region and the rest of the estuary.
 - Identification of variability and source information in key-pathways of energy and material flows between primary producers, benthic detritus deposits, and invertebrate consumers.
1. Although biological assemblages within the REI region comprise the same basic set of trophic guilds compared with regions further downstream, the relative contribution of individual functional groups to the community differs between reaches.
 2. Benthic invertebrate assemblages near the head of the estuary are dominated by filter-feeders which are largely replaced by surface-active deposit feeders towards the marine dominated lower estuary. The prevalence of suspension feeders is likely to be a direct response to the high productivity of pelagic microalgae and/or sedimentation of river-borne organic material imported into the ebb and flow region.
 3. Benthic communities of the REI region are not only unique in terms of species composition and abundance but also have a trophic organisation that differs from the more saline estuarine waters. Thus, faunal boundaries along the salinity gradient roughly match steep clines in functional group dominance, giving rise to distinct bands of changing faunal composition along the main axis of the estuary.
 4. Pelagic primary producers in the REI carry carbon signatures that are distinct from that of phytoplankton in polyhaline waters. These differences in stable carbon ratios are mirrored by invertebrate consumers whose tissue carbon composition also differs significantly between the REI region and the rest of the estuary.
 5. High standing crops of fringing macrophytes near the tidal head of estuaries are seen as important sources of detrital material that enters estuarine food webs. In the Gamtoos, the dual (N and C) isotope signature of vascular plants can be clearly traced in particulate sediment deposits of the upper and middle reaches. Thus, fringing macrophytes are identified as a significant source of organic material to the estuarine benthos, with net seaward flow transporting organic material from the REI to estuarine biota.
 6. Intertidal invertebrates are a key element in the flow of energy and material within the estuary. To accurately identify their trophic role, information on spatial variability in consumption processes is a prerequisite. Preliminary findings indicate considerable scope for flexibility in trophic roles across a spatial scale as small as the intertidal zone within a given locality.

5.2.2 Introduction

Estuarine environments display high spatio-temporal variability in biological and ecological parameters, which are in turn externally forced by sharp gradients in physical and chemical properties. Among several gradients that are evident at different scales, the variation in salinity from the marine to the riverine end of an estuary is perhaps the most prominent one. Although this gradient must essentially be regarded as a spatial continuum of changing salinity, estuarine biota appear to integrate clines in physico-chemical parameters over time. This gives rise to distinct bands of changing community structure across the main axis of estuaries. Such segments display above average internal similarity and provide ecologically meaningful zones for the subdivision of estuaries (Bulger *et al.* 1993; Schlacher and Wooldridge 1996a).

The zone where river water meets the tidally imported seawater (REI) is a highly dynamic and important site for chemical and biological reactions (Fauzi and Mantoura 1987). This REI region supports high pelagic primary productivity (Schuchardt and Schirmer 1991), and is a site of enhanced deposition of riverine imported organic material. Enhanced availability of organic material in the REI stimulates bacterial growth and supports high densities of heterotrophs (Montagna and Yoon 1991). From a community perspective, the REI harbours unique benthic communities, which differ significantly in structure from assemblages in more saline waters (Schlacher and Wooldridge 1996a).

Despite a growing recognition of the REI as an essential element of the estuarine ecosystem, its functional role is still poorly understood. In this module we take an essentially two-pronged approach to identify some principal features and properties, which relate to functional processes in and across the REI.

1. The first thrust was to integrate information on feeding modes and trophic roles of benthic invertebrates into patterns of community structure identified for these unique assemblages.
2. Secondly, to determine principal pathways of energy and material flow in the REI and its coupling with other estuarine reaches, stable isotopes were used as chemical tracers of food-web linkages.

5.2.3 Methods

All field work during the 1997 module of this component was undertaken in the Gamtoos Estuary. A detailed description of the estuary is given by Schlacher and Wooldridge (1995) and Schlacher and Wooldridge (1996a). A special emphasis in the spatial scale of investigation was placed on contrasting community structure in the REI region with that of the rest of the estuary. We continue to use a bottom salinity value of 10.0 ‰ as the boundary between the REI and the rest of the estuary. A number of schemes are currently in use to classify segments of estuaries into zones with different regimes of tidal energy and salinity (Bulger *et al.* 1993; McLusky 1993). The terminology of REI used here is consistent with the seminal work of Morris *et al.* (1978; see also Fauzi and Mantoura 1987) and we feel there is little to be gained from coining a new term here. It must, however, be stressed

that an unambiguous definition of both the upper and lower salinity bounds of the REI region need further refinement.

Macrobenthic data analysed in this report stem from three surveys carried out in 1993, 1994 and 1996 (Schlacher and Wooldridge 1996a). A new addition to this report is the incorporation of functional groups (in the sense of trophic guilds) into the multivariate analysis of spatial patterns in community composition. Classification of invertebrates into functional feeding groups is based on Schlacher and Wooldridge (1996b), Fauchald and Jumars (1979) for polychaetes, Branch and Branch (1981) and Branch *et al.* (1994).

To trace material and energy flows in the ecosystem we sampled the dominant classes of primary producers, sediment detrital deposits and invertebrate consumers in both the REI region and the rest of the estuary. Macrophytes were harvested by randomly removing five to ten samples from a stand; plant material was collected from the same sites where detritus and benthic invertebrates were sampled. To ameliorate the effect of intraspecific variation in stable isotope composition, all plant samples in this study were composites of at least five entire individuals, cropped randomly from a specific site. Similarly, invertebrate samples were composites made up of five to several thousand individuals depending on body weight of the species. Water samples for the determination of $\delta^{13}\text{C}$ in dissolved inorganic carbon (DIC) were collected around mid-tide in the centre of the estuarine channel adjacent to sites where benthic organisms were collected. These samples were immediately preserved with saturated HgCl_2 solution to arrive at a final concentration of 1.0 ppt HgCl_2 . One series of mesozooplankton samples was collected at eight sites spanning a salinity spectrum from 2.6 to 31.4 ‰ (see Schlacher and Wooldridge 1995). Benthic invertebrates were collected with a surface dredge in the shallow subtidal or with a yabie-pump in the intertidal. All samples were immediately placed on ice and deep-frozen within six to eight hours of collection.

Biogenic material was thoroughly washed in fresh water to remove any sediment, rinsed in distilled water and decalcified with 5.0 % HCL. After carbonate removal, samples were rinsed twice in deionised water, dried to constant weight and homogenised. The Quaternary Research Dating Unit of the CSIR performed stable isotope measurements. Isotopic compositions of samples are expressed in conventional notation as δ -values, which are deviations from a reference standard:

$$X = [(R_{\text{sample}}/R_{\text{reference}}) - 1] \times 10^3$$
, where X is ^{13}C or ^{15}N , and R is the corresponding ratio $^{13}\text{C}/^{12}\text{C}$ or $^{15}\text{N}/^{14}\text{N}$. Reference material was an internal standard calibrated against the Chicago Pee Dee Belemnite (PDB) for carbon and atmospheric N_2 for nitrogen.

Multivariate analysis of community structure followed the procedure detailed in Clarke (1993). Similarity matrices between sites based on taxonomic structure of assemblages were calculated from $x^{-0.25}$ transformed abundance data. No transformation was applied to relative density data of functional groups within assemblages (i.e. column-standardised data). To test for multivariate differences between the REI and the estuary, a one-way ANOSIM test was performed on similarity matrices from both species and guild data. Differences in trophic guild diversity between the REI and the estuary were tested by an Analysis of Covariance (ANCOVA) specifying species diversity as the covariate. Univariate measures of diversity were Hill's N1 and Hill's N2 (Schlacher and Wooldridge 1996a).

5.2.4 Results

5.2.4.1 Spatial trends in functional group composition

The most striking pattern in the spatial variability of trophic guilds within the invertebrate communities of the estuarine benthos was a sharp increase in filter feeders towards the ebb and flow region of the estuary. This pattern can be expressed in terms of linear distance from the mouth (Figure 5.2.1B) or salinity (Figure 5.2.1D). Concomitant with a dominance of species having filter-feeding mode in the REI region was a sharp decline in surface-active deposit feeders in oligohaline waters (Figure 5.2.1 A and C). These trends are also reflected at a straight linear scale along the main axis of the estuary, showing that benthic communities towards the marine reaches become increasingly dominated by surface-active deposit feeders, which gradually displace the filter-feeding forms that prevail at the head of the estuary (Figure 5.2.1 A and B). Neither burrowing forms of deposit feeders nor predators showed any clear spatial trend in their respective contributions to overall community functional group composition (Figure 5.2.1).

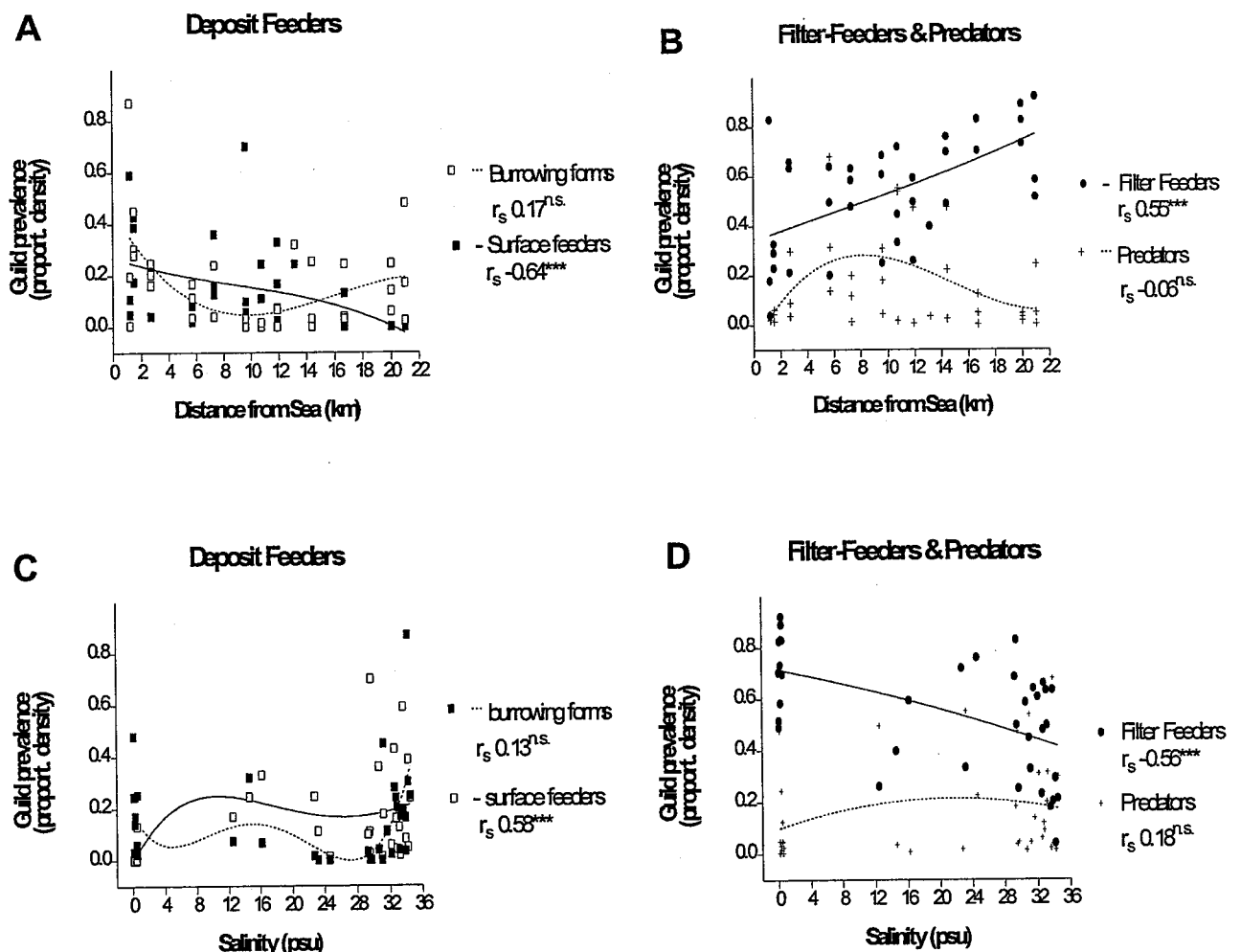


Figure 5.2.1: Axial gradients in trophic guild composition for zoobenthic assemblages along the Gamtoos Estuary (A and B) and relationships between salinity and functional group dominance (C and D) (r_s - Spearman Rank Correlation coefficients).

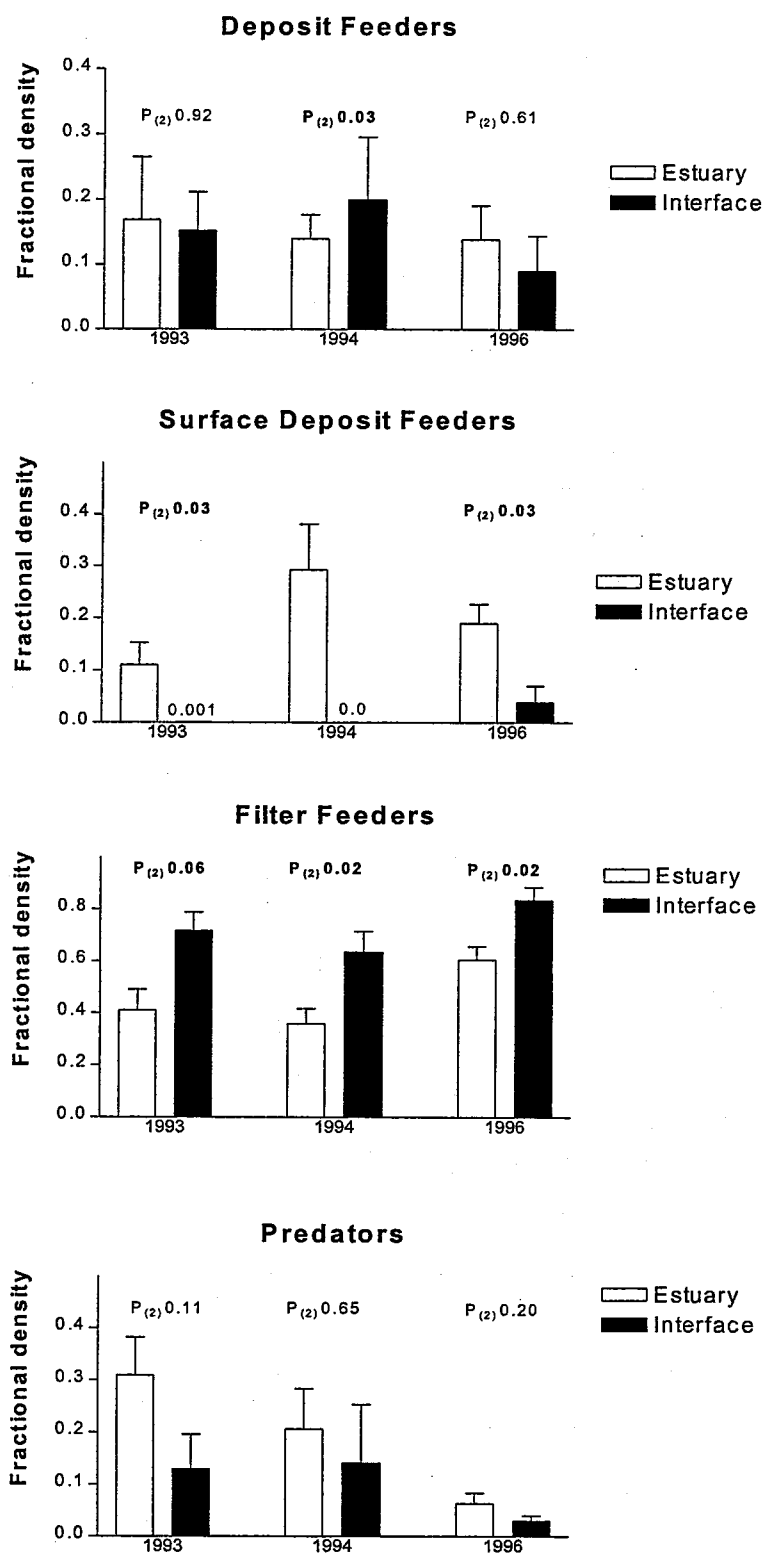


Figure 5.2.2: Contrast in trophic group prevalence between benthic assemblages of the REI (bottom salinities < 10.0 ‰) and communities of all estuarine habitats further seawards. (P values refer to t-tests).

Table 5.2.1: Summary of Analysis of Covariance (ANCOVA) to compare trophic guild diversity of benthic invertebrate assemblages between the REI and the estuary (*cf.* Figure 5.2.2). Data set consists of three series of macrofaunal abundance data covering the full salinity spectrum across the tidal reaches of the Gamtoos Estuary.

		SS	d.f.	F-Ratio	P
Source of variation					
Hill's N1	Covariate: Species Diversity	4.35	11	4.35	< 0.001
	Main effect: Segment (i.e. REI vs. Estuary)	0.91	1	6.09	0.019
	Residual	5.08	34		
Hill's N2	Covariate: Species Diversity	5.62	1	36.89	< 0.001
	Main effect: Segment (i.e. REI vs. Estuary)	0.88	1	5.764	0.022
	Residual	5.18	34		

A direct comparison of the trophic groups associated with the REI with those of the rest of the estuary reveals a clear distinction in the trophic make-up of the benthic invertebrate communities between these two zones. Again, filter-feeders were significantly more abundant in the REI at all times, whereas surface deposit feeders are poorly represented in the assemblages of the ebb and flow region (Figure 5.2.2).

We have previously shown that the macrobenthos in the REI has a lower species diversity compared with regions further seawards (Schlacher and Wooldridge 1996a). This decrease in species diversity is also reflected in a decline in trophic guild diversity towards oligohaline waters (Figure 5.2.3). Irrespective of the measure of diversity employed, benthic communities of the REI are characterised by a less diverse array of functional feeding groups compared with assemblages of polyhaline waters in the estuary (Figure 5.2.3; Table 5.2.1). Low functional group diversity of the REI can be traced to the overriding dominance of filter-feeders coupled with an almost complete absence of surface deposit feeders at the head of the estuary (Figure 5.2.3).

Incorporating functional aspects (e.g. trophic modes, reproductive traits, dispersal traits) of community organisation into a classification of community types found in the tidal reaches of an estuary, expands on the interpretation of traditional approaches, which are usually based purely on taxonomic composition (Schlacher and Wooldridge 1996a). In the context of this programme it becomes pertinent to establish to what degree zonation patterns based on taxonomic community structure match variation in functional group composition across the salinity spectrum (*cf.* Figures 5.2.1 and 5.2.2). Although classification of assemblages based on their trophic make-up largely matches that of taxonomic community structure, there is considerable less consistency in establishing the identity of the REI based on trophic guild composition (Figure 5.2.4). While benthic assemblages

Table 5.2.2: Summary of Analysis of Similarity (ANOSIM) comparing community structure of benthic assemblages between the REI region and the estuary at two scales of resolution for community structure.

	Species (taxonomic composition)		Guilds (functional group composition)	
	Global R	P	Global R	P
1993	0.565	1.80%	-0.054	49.10%
1994	0.559	0.40%	0.250	5.50%
1996	0.599	0.20%	0.184	13.10%

of the REI region show a consistent and significantly different structure compared with all regions further seaward, this distinction is less apparent using compositional prevalence of feeding groups within each estuarine region (Figure 5.2.4, Table 5.2.2).

In addition, there is significantly more year-to-year variability in zonation patterns based on functional group composition, compared with patterns derived from taxonomic structure alone (Figure 5.2.5). Overall, the different nature of the information contained in each approach (taxonomic vs trophic structure) leads to distinct spatial patterns across a time scale of years but is largely complementary at the time scale of a single investigation (Figure 5.2.5).

5.2.4.2 Extending the functional characterisation of the REI

Isotopic signatures of primary producers and invertebrate consumers were used to trace principal routes of material and energy flows in the estuarine ecosystem and showed changing patterns across the salinity gradient of the REI (Figure 5.2.6). Fringing macrophytes in the middle and upper reaches are slightly more depleted in ^{13}C at the head of the estuary (mean $\delta^{13}\text{C} = -26.2 \pm 0.38$) compared with regions further seawards (mean $\delta^{13}\text{C} = -25.2 \pm 0.39$). The overall isotopic signature of the vascular plants does, however, not change dramatically between the REI and the estuary (Figure 5.2.6 A).

Lack of spatial variation in carbon isotope composition is equally evident for benthic detrital deposits within the sediments (Figure 5.2.6 B). By contrast, $\delta^{13}\text{C}$ values of phytoplankton increase steeply with increasing salinity from -34.0 in upper reaches waters to -27.0 in the middle reaches (Figure 5.2.6 C). Phytoplankton cells in the REI have a significantly more depleted carbon isotope composition ($\delta^{13}\text{C} = -32.45 \pm 0.48$) compared with the estuary ($\delta^{13}\text{C} = -30.05 \pm 0.67$; $t = 2.91$, $P_{(2)} = 0.01$). The pattern of carbon isotope variation in pelagic microalgae across the REI is clearly mirrored by benthic consumers (Figure 5.2.6 D), whose $\delta^{13}\text{C}$ at the REI averages -31.40 ± 0.36 compared with a significantly more positive signal of -28.2 ± 0.46 in the estuary ($t = 5.655$, $P_{(2)} < 0.001$).

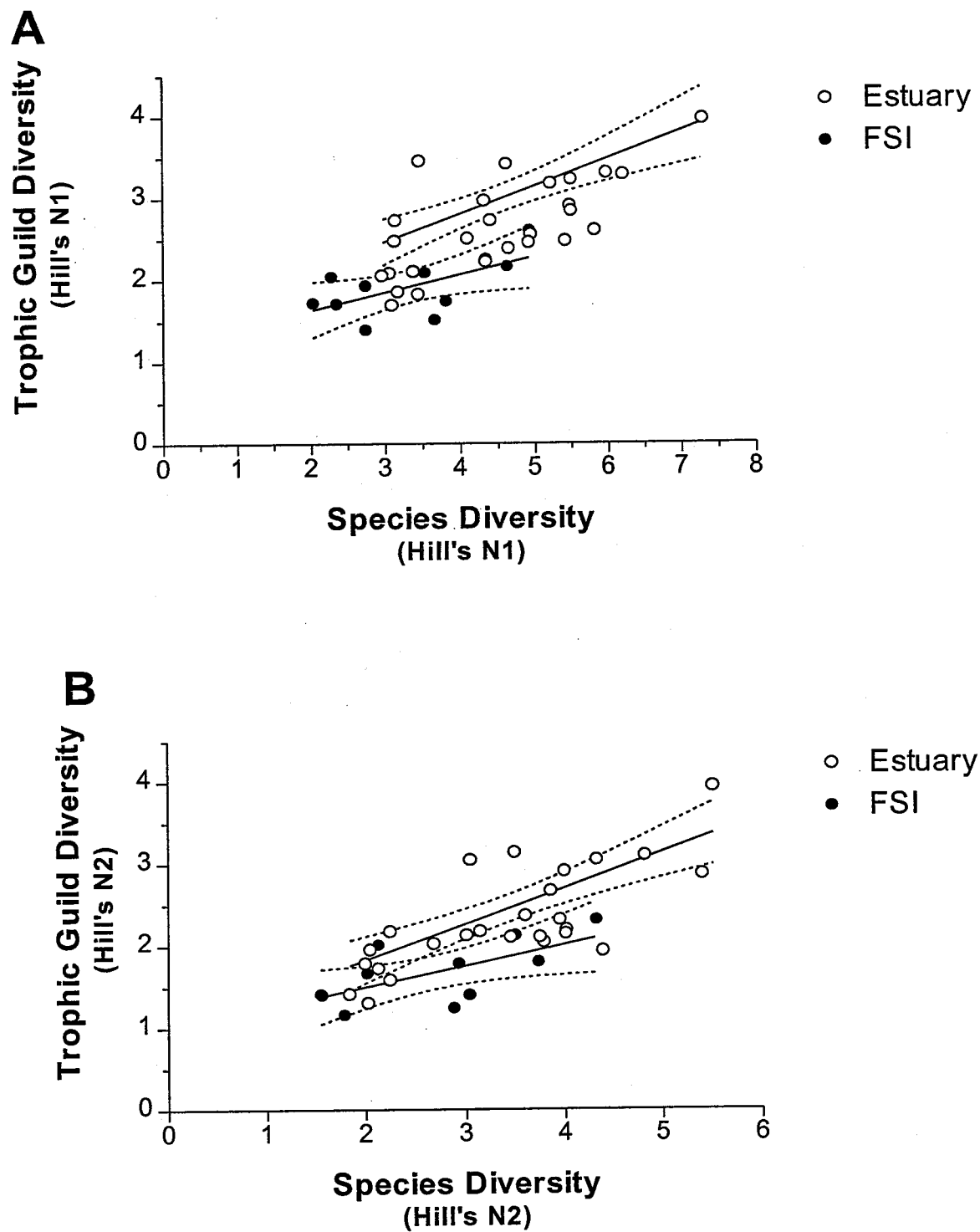


Figure 5.2.3: Relationship between species diversity and functional group diversity (as measured by trophic guilds) in two regions of the Gamtoos Estuary defined by a 10.0 ‰ salinity boundary (*cf.* Figure 5.2.2). Trophic group diversity is significantly lower in the REI region for both univariate measures of diversity (*cf.* Table 5.2.1).

Species

Guilds

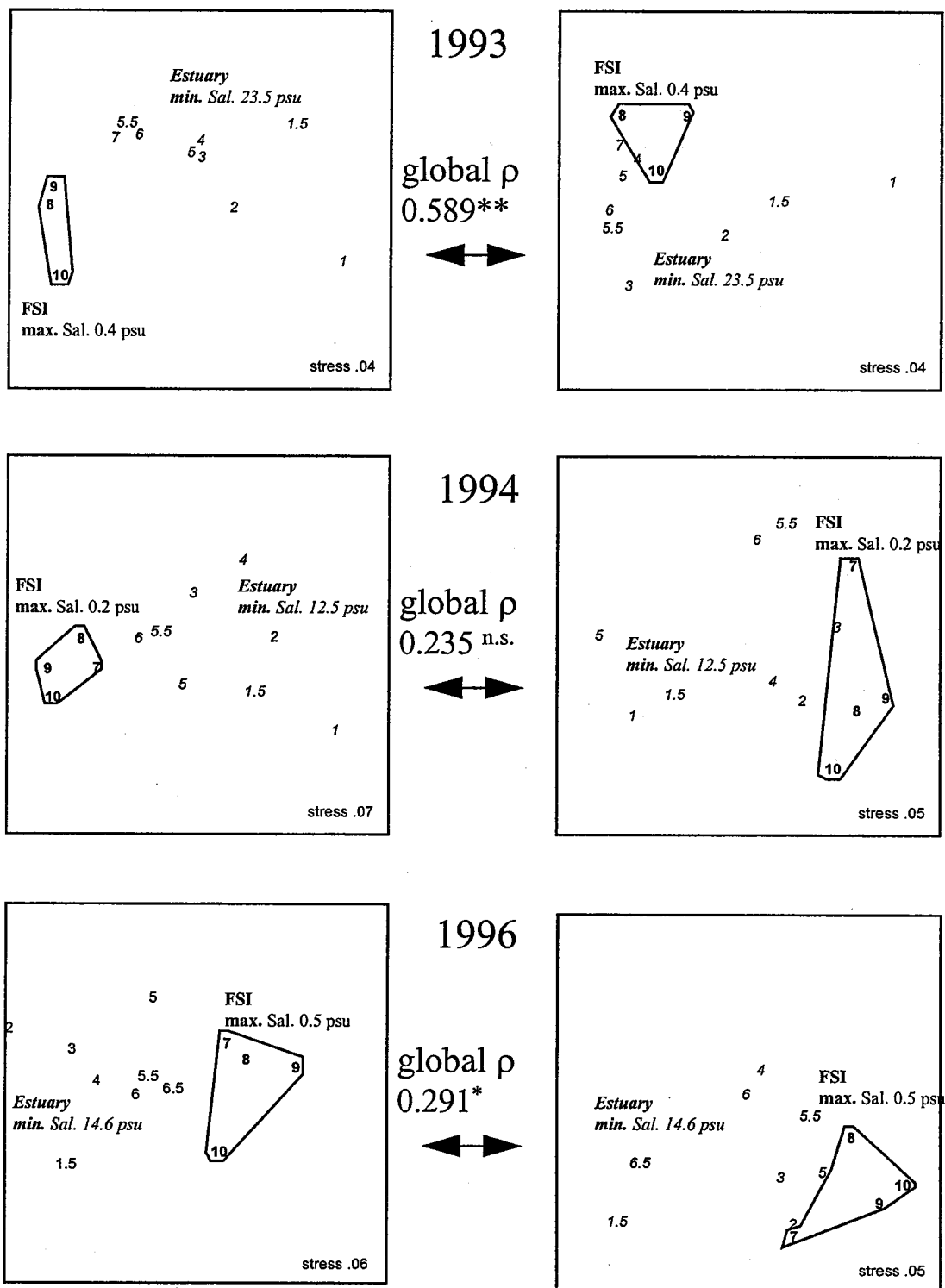


Figure 5.2.4: Ordinations of macrobenthic assemblages sampled at 12 sites in the Gamtoos Estuary. Analysis based on either taxonomic composition or trophic guild composition with p values indicating the degree of concordance between corresponding similarity matrices (Spearman Coefficients).

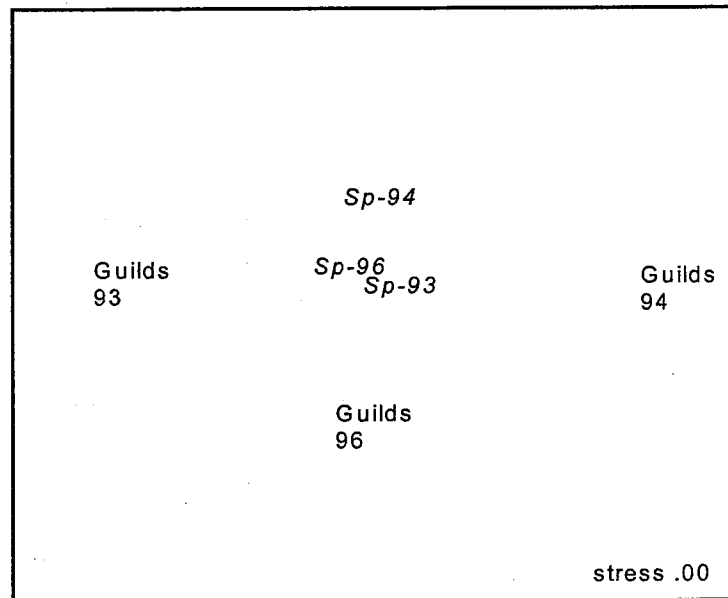


Figure 5.2.5: Second-stage ordination of macrobenthic assemblages to check for similarity of pattern among data structures (i.e. species vs guilds) and sampling times. (Sp - analyses based on species abundance; Guilds refers to similarity calculations done on proportional trophic group representation).

Fringing macrophytes are hypothesised to be a significant source of energy input to the estuary via detritus production - a view consistent with the classical 'detritus paradigm' pertaining to energy flows in estuarine ecosystems (e.g. McLusky 1981). Because standing biomass of macrophytes is highest in the ebb and flow region of the system under study (Adams 1991), detrital inputs are likely to be strongest around the REI zone. A first step in assessing the trophic importance of macrophyte primary production of the low salinity areas is to establish whether fringing vascular plants are indeed the major source of detritus in the estuary. Both the nitrogen and carbon stable isotope signatures from sedimentary detrital deposits can be clearly traced to macrophyte inputs (Figure 5.2.7). There is a slight enrichment in mean $\delta^{13}\text{C}$ between detritus and vascular plants - this could possibly be a reflection of additional inputs of light plant material from adjoining agricultural lands. However, the importance of estuarine macrophytes as the major supplier of particulate detritus to the estuary is unambiguous (Figure 5.2.7).

A considerable proportion of invertebrate production within the estuary is concentrated in the intertidal areas, which harbour dense assemblages of thalassinid prawns (e.g. sand prawn *Callinassa kraussi* and mud prawn *Upogebia africana*). Intertidal invertebrates are therefore a key compartment in understanding energy and material flows within the overall system. Populations inhabiting different tidal levels in the intertidal zone are subjected to distinct physico-chemical regimes (i.e. submergence time, temperature, suspended food concentration). Consequently, a crucial step in establishing pathways of energy and material flow through this compartment is to quantify variation in trophic mode at the scale of tidal elevation. There are strong indications that consumption rates of

different categories of primary production are not uniform across the intertidal gradient (Figure 5.2.8). Populations of thalassinid prawns from the low shore appear to utilise more phytoplankton compared with conspecifics of the high shore in which a greater proportion of energy intake is met by benthic material (Figure 5.2.8). Thus, there appears to be considerable scope for trophic mode flexibility in this group - a critical parameter which calls for closer quantification if energy flows are to be modelled with any reasonable degree of confidence in the future.

5.2.5 Discussion

We have previously shown that the River-Estuary Interface harbours distinct communities and can be clearly defined based on biological criteria (Schlacher and Wooldridge 1996a). Although fluctuations in river run-off are a major external forcing factor to the estuary and result in a continuous shifting of the spatial location of salinity boundaries (Schumann and Pearce 1997), the biological uniqueness of the assemblages in the REI remains essentially intact - a conclusion based on structural features of the communities measured.

An important aspect of this subprogram was to determine the functional characteristics of the REI region and to what extent this area contributes to overall estuarine functioning. In the approach taken here, which focused on integrating process information into more traditional data sets that rely heavily on structural information, an important step has been made. Both the application of functional group concepts and the use of stable isotope techniques extend our knowledge of estuarine processes considerably.

While benthic assemblages of the REI are perhaps slightly less distinct in their functional group composition than in their taxonomic make-up, a pattern of differentiation between polyhaline and oligohaline waters remains a strong one. Most strikingly, the dominance of filter-feeding forms in this region can be interpreted to be a direct response to the high microalgal productivity (e.g. Schuchardt and Schirmer 1991) and to sedimentation of riverborne material when fresh water interacts with sea water. The fate of detrital material derived from the macrophytes in the REI and transported downstream is currently unresolved (see also Schlacher and Wooldridge 1996b). Possible transfer pathways in the food-web are direct mineralisation through the microbial loop and/or consumption by meiofauna, but direct uptake by benthic invertebrates appears to be unimportant (Schlacher and Wooldridge 1996b).

Determining trophic patterns by applying functional classifications to consumers is, however, not always consistent with direct measurements of energy and material flows. In local estuarine settings this can lead to paradigm shifts about the role of some key-stone species (e.g. Schlacher and Wooldridge 1996b). Similarly, our findings on the trophic mode flexibility in intertidal consumers reported here, argue forcefully for a more accurate determination of consumption processes in estuarine ecosystems. This will be a crucial component in establishing energy pathways more reliably and will greatly enhance the resolution of local food-web research. Ultimately, this should prove to be an essential component in advancing our understanding of local estuarine ecosystems from structural properties to functional process dynamics.

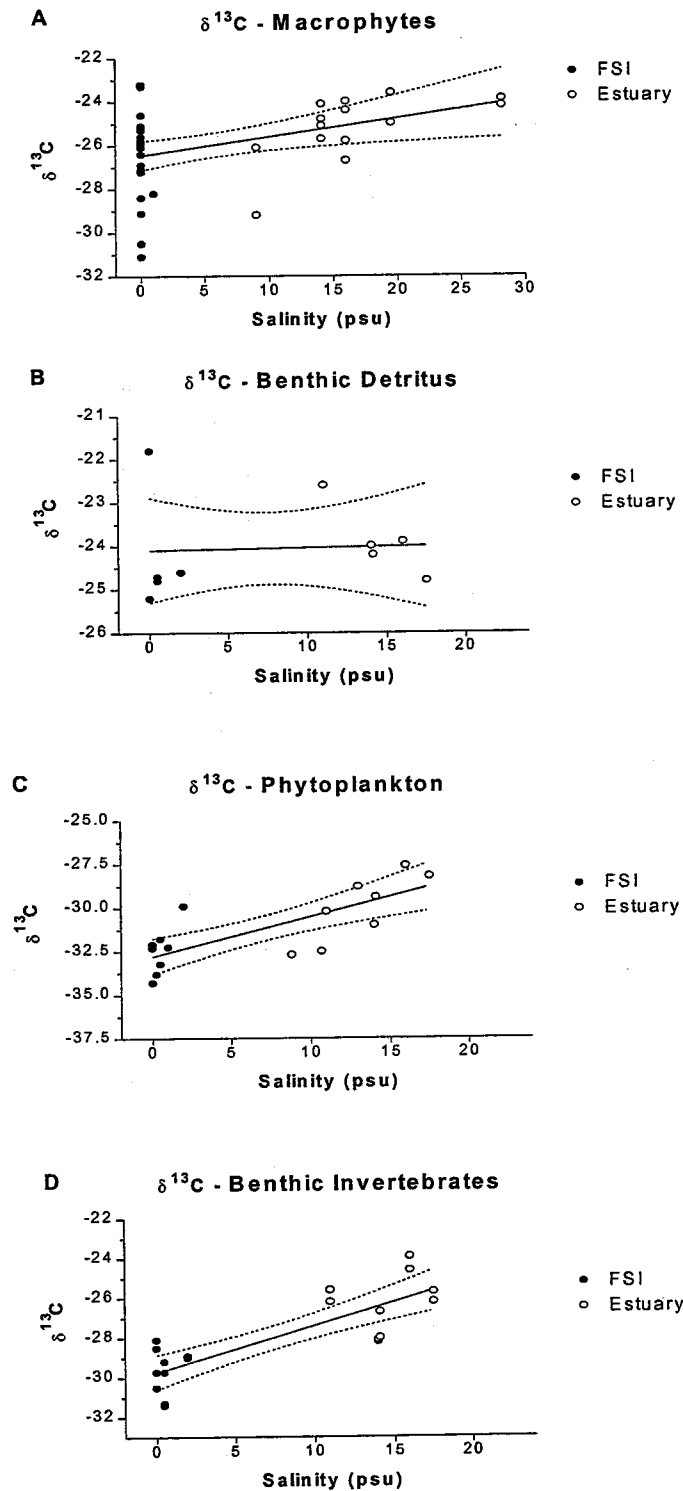


Figure 5.2.6: Relationship between water column salinity and carbon isotope ratios of two classes of primary producers (A-Macrophytes, C-Phytoplankton), particulate detritus in the sediments (B) and benthic invertebrates (D- the amphipod *Grandidierella lignorum*). FSI refers to regions with salinities close to fresh water.

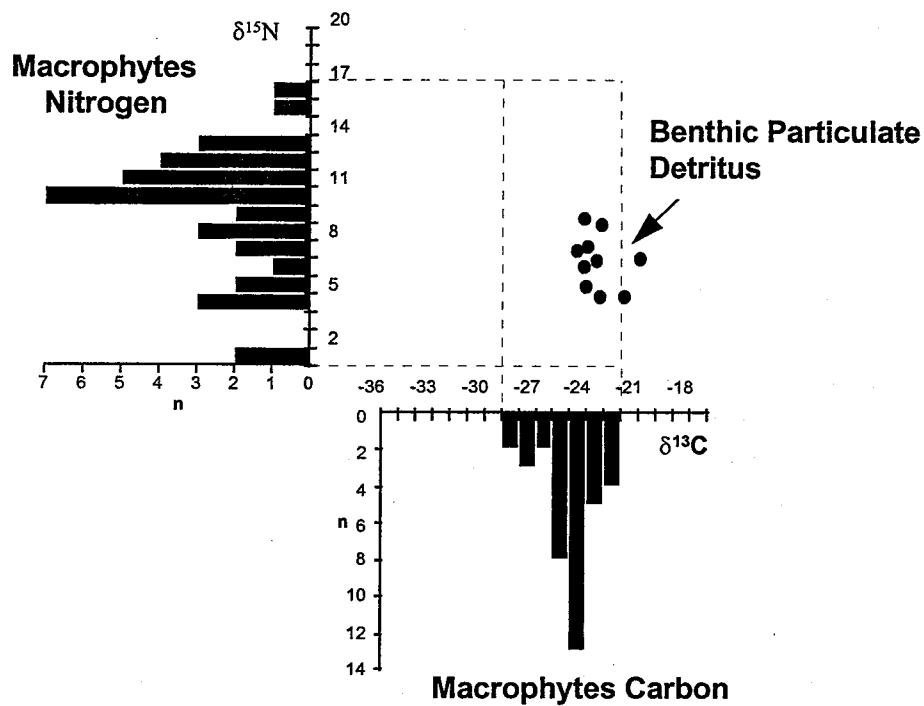


Figure 5.2.7: Stable nitrogen and carbon isotope ratios of macrophytes in relation to isotopic signatures of detrital particles in the estuarine subtidal sediments. Dotted lines encompass range of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ values of the vascular plants.

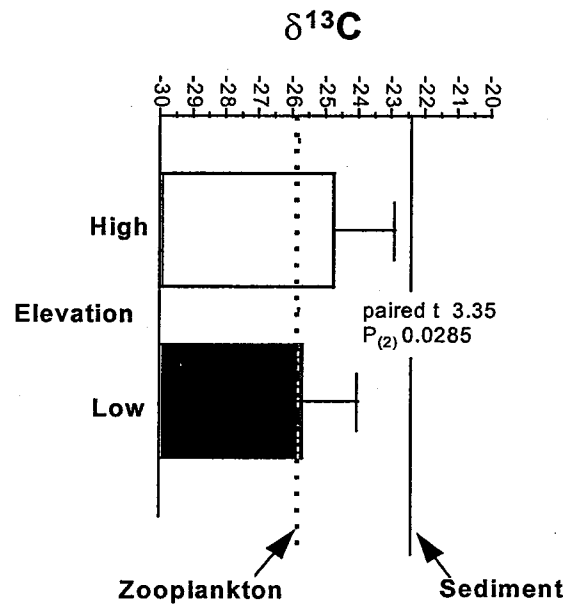


Figure 5.2.8: Comparison of stable isotope ratios of thalassinid prawns at two tidal elevations (HWST and LWST) in the intertidal zone of the Gamtoos Estuary. Dotted lines show mean $\delta^{13}\text{C}$ value of zooplankton sampled adjacent to locations where benthic prawns were collected while solid line is the mean value of burrow sediments.

5.3 Variation of food web processes across estuarine salinity gradients - implications for the ecological characterisation of the REI region

5.3.1 Summary

Because food-web processes in estuaries may be a function of salinity, low salinity reaches such as the REI region may exhibit ecosystem properties that differ substantially from more saline reaches. To test the above hypothesis and advance our understanding of estuarine processes, this study focused on determining gradients in carbon signals and trophic links of the estuarine food web across the salinity spectrum. Significant progress was made in the following areas:

1. Primary producer's (phytoplankton, benthic microalgae and macrophytes) carbon isotope ratios are a function of habitat salinity. Within an estuary, delta ^{13}C values of plants increase significantly and linearly with salinity.
2. Steep gradients in carbon isotope composition of primary producers across the salinity spectrum are mirrored by equivalent relationships between delta ^{13}C and salinity in consumers (prawns, amphiods, crabs and bivalves).
3. Although the basic patterns of energy flow among the principal trophic compartments appear to change little in different salinity regimes and geographic subdivisions of an estuary, carbon isotope signatures of the food web shift systematically from freshwater to marine regions.
4. Not only are carbon isotope ratios a function of salinity in both producer and consumer groups, but this variation across the salinity spectrum is a similar and systematic one in auto- and heterotrophs. This suggests that production and consumption of organic matter is localised and that principal food web processes may be spatially restricted to particular reaches of an estuary. Conversely, horizontal fluxes and mixing of organic matter may be less prevalent.
5. The REI region serves perhaps as a particularly good example where a dualism of unique species assemblages (structure) and spatially restricted food webs (function) may be well developed.
6. Information about the distinct biological make-up of the REI supports the concept of 'localised food webs', thus reinforcing the initial hypothesis of this research programme, that low salinity regions present unique estuarine habitats.

5.3.2 Introduction

Tracking of trophic linkages in food webs of South African estuaries has, over the past decade, seen traditional approaches such as direct feeding observations and stomach content analysis being complemented by stable isotope studies (de Villiers 1990; Jerling and Wooldridge 1995; Schlacher and Wooldridge 1996a; Paterson and Whitfield 1997). Results from this new approach have either supported conventional wisdom, such as the importance of salt marshes and seagrasses as carbon sources in marine-dominated systems (Paterson and Whitfield 1997), queried the apparent energetic importance of macrophytes in food webs of estuaries with considerable freshwater inflow (Jerling and

Wooldridge 1995), or refuted the paradigm that detritus derived from macrophytes is a significant carbon source for benthic invertebrates in estuaries (Schlacher and Wooldridge 1996a).

In the context of this programme, we have already shown that pelagic primary producers of the REI region have carbon signatures that are distinct from that of phytoplankton in more saline waters. While there is some evidence from the first two modules of this programme (see sections 5.1 and 5.2) that food-web structures may vary across the estuarine salinity gradient (including the REI region) there was insufficient sampling coverage at intermediate salinity levels to determine whether key-processes in trophic webs are indeed a function of salinity.

The question of salinity dependence relates directly to ecosystem-wide aspects of the overall programme hypothesis which states that ".....the interface region has strong influences on the physico-chemical as well as biological structure and functioning of the entire estuary". Consequently, to contribute towards testing the overall programme hypothesis outlined above, research undertaken during this study focused on complementing information on estuarine food web functioning with stable isotope data on key components for which trophic linkages under different salinity regimes had not yet been documented. To this end, we determined carbon signals of benthic microalgae and phytoplankton across the estuarine salinity spectrum, and their trophic linkages with pelagic (zooplankton) and benthic (anomuran prawns) invertebrate consumers.

5.3.3 Methods

To trace material and energy flows in the ecosystem we sampled benthic microalgae, sediment-bound organic matter, mesozooplankton (> 0.2 mm), macrobenthos and dissolved inorganic carbon (DIC) in the water column across the estuarine salinity range, incorporating the REI region.

Most invertebrate samples were composites of five (anomuran prawns) to several thousand (copepods) individuals depending on body weight of the species. Individuals were, however, never pooled across sites to preserve habitat-specific information. A few species (e.g. the razor clam, *Solen capensis*, and the crabs, *Hymenosoma orbiculare*, *Partylodiplax edwardsii* and *Thaumastoplax spiralis*) occurred only at low densities, and one to three individuals constituted a sample for stable isotope analysis. Benthic invertebrates were collected either with a surface dredge that sampled the top 2.0 cm in shallow subtidal waters (for amphipods) or with a yabie-pump in the intertidal (for anomuran prawns). Animal samples were kept in cooler boxes until deep-freezing, which was usually completed 6 to 8 hours after field collections.

Mesozooplankton was sampled with WP2-nets at eight sites along the main axis of the estuary, progressing from the tidal freshwater section to predominantly marine waters near the estuarine mouth. The sampling protocol is detailed in Schlacher and Wooldridge (1995). Briefly, hauls were made just below the surface and at approximately 20.0 cm above the bottom, excess water removed from the catch by gently shaking it in a sieve on board, transferred to storage bottles and stored on ice overnight. The following day, zooplankton samples were briefly frozen, re-thawed and sorted under a stereo microscope into monospecific samples for stable isotope analysis.

Extraction of benthic microalgae essentially followed the procedure of Riera *et al.* (1996). The top 2.0 to 3.0 mm of intertidal sediment was gently scraped off with a spatula; about 400 to 500 ml of surface layer were collected from each site. Surface scrapings were generally made from areas where dense microalgal growth was visible on the sediment surface in the form of greenish to brownish patches. Three to five hours after collection, these sediment samples were spread about 1.0 cm thick on trays (20 x 30 cm) and held under fluorescent light overnight. Next morning, nylon screens (0.09 mm mesh size) were placed on top of the sediment and covered with a 1.0 to 5.0 mm thick layer of quartzite sand. The sand used was previously sieved into a 0.25 to 0.5 mm size fraction, washed twice with tap water, decalcified in 10 % HCL and combusted in a muffle furnace to remove organics. Trays were kept moist by a slight spraying of GF/C-filtered estuarine water. After a further five hours of illumination, benthic microalgae had migrated into the quartzite layer as evidenced by green-yellow growth patches on the sand surface. The sand layer was transferred twice to Erlenmeyer flasks and vigorously shaken in pre-filtered estuarine water to dislodge diatoms from sand particles. The supernatant was sieved through a 0.055 mm screen to remove meiofauna, and the microalgae finally collected on Whatman GF/C filters. Filters with microalgae were acidified with 5.0 % HCL to remove carbonates, rinsed with distilled water, and dried at 60°C for 12 hours.

Water samples for DIC $\delta^{13}\text{C}$ determinations were collected in the centre of the estuarine channel. Exactly 500 ml of water was filtered through Whatmann GF/C filters and preserved with saturated HgCl_2 solution to arrive at a final concentration of 1.0 ‰ HgCl_2 . Suspended particulate organic matter (SPOM) samples were collected several metres offshore from intertidal populations of anomuran prawns at a depth of 0.3m. Water samples for SPOM were filtered immediately onto GF/C filters; the volume of water filtered depended on the amount of suspended material and ranged between 300 and 700 ml. Filters were kept in cooler boxes overnight, acidified with 5.0 % HCL, and dried at 60 to 70°C.

Coarse benthic detritus particles were collected from surface dredgings on a 1.0 mm mesh, and living macroscopic components manually removed under a stereo microscope. If the spatial scale and integration of sediment collection does not match distributional patterns of benthic organisms, so-called 'bulk' sediment samples may not accurately describe substrate features at biologically relevant scales. To ensure that sediment readings reflected, or at least approximated the organic matter sources actually available to consumers *in situ*, sediment samples in this project were collected from animal burrows. We sampled the tunnel linings and a 0.5 to 1.0 cm layer of substrate surrounding each tunnel from burrows of anomuran prawns. About 5 to 10 burrows per site were sampled in this manner for sediment, and pooled for d^{13}C analysis of organic matter. Sediment samples were acidified with 10.0 % HCL until all CO_2 production stopped and dried at 80°C. To avoid loss of dissolved organic matter, sediment samples were not rinsed after acidification.

All plant and animal samples were thoroughly washed in fresh water to remove any sediment, rinsed in distilled water and decalcified in 5.0 % HCL. After carbonate removal, samples were rinsed twice with deionised water, dried to constant weight and homogenised.

Stable isotope ratios ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$) of organic materials were determined on a VG SIRA 24 mass spectrometer. Analytical precision was typically 0.2 ‰ over five replicate analyses. Isotopic compositions of samples are expressed in conventional notation as δ -values, which are deviations from a reference standard: $X = [(R_{\text{sample}}/R_{\text{reference}}) - 1] \times 10^3$, where X is ^{13}C or ^{15}N , and R is the corresponding ratio $^{13}\text{C}/^{12}\text{C}$ or $^{15}\text{N}/^{14}\text{N}$. Reference material was an internal standard calibrated against the Chicago Pee Dee Belemnite (PDB) for carbon and atmospheric N_2 for nitrogen.

5.3.4 Results

5.3.4.1 Primary food sources - isotopic composition

Stable isotope ratios ($\delta^{13}\text{C}$) of carbon sources that are potentially available to higher trophic levels in the estuary ranged widely from -14.7 ‰ for benthic microalgae to a minimum of -31.1 ‰ for macrophytes (Figure 5.3.1, Table 5.3.1). Despite the broad range in $\delta^{13}\text{C}$ values, the major classes of primary producers could be clearly separated. Benthic microalgae were most enriched in ^{13}C (mean $\delta^{13}\text{C}$: -17.8 ‰ $\pm$ 2.2), and were always more positive compared with their pelagic counterparts (phytoplankton mean $\delta^{13}\text{C}$: -29.7 ‰ $\pm$ 3.6). Benthic microalgae were distinctly more positive than macrophytes (mean macrophyte $\delta^{13}\text{C}$: -26.0 ‰ $\pm$ 1.9), and carbon isotope ratios of these two groups of primary producers did not overlap (Figure 5.3.1).

Diatoms were the numerically dominant group of the benthic microalgal assemblages, comprising up to 98.0 % of cell numbers, followed by blue-green bacteria and euglenophytes (Table 5.3.2). Other taxa such as greens and flagellates were not abundant, and although there was some contamination of samples with meiofauna, this was not significant. Some evidence for size fractionation in the taxonomic make-up of benthic algal communities was evident: size fractions > 55.0 μm were almost exclusively composed of blue-greens, whereas diatoms again dominated the < 55.0 μm fraction at the same site (Table 5.3.2).

Although some individual samples of macrophytes had $\delta^{13}\text{C}$ values similar to phytoplankton values, macrophytes were generally enriched in ^{13}C compared with phytoplankton by 3.0 to 6.0 ‰.

Benthic detritus invariably consists of a mix of different source materials. Carbon isotope ratios of coarse (> 1.0 mm) organic particles that could be visually separated from the sediment broadly mirrored those of vascular plants (mean detritus $\delta^{13}\text{C}$: -24.1 ‰ $\pm$ 1.1). By comparison, organic matter in sediment samples that was not separated into size fractions was markedly more enriched in ^{13}C (mean sediment $\delta^{13}\text{C}$: -21.3 ‰ $\pm$ 1.8). This shift possibly reflects the contribution of 'heavy' carbon by benthic microalgae.

5.3.4.2 Invertebrate consumers - isotopic composition

Although pooled frequency distributions of $\delta^{13}\text{C}$ values of benthic invertebrates and zooplankton broadly encompass the whole range of $\delta^{13}\text{C}$ values found in primary producers - from enriched benthic microalgae to depleted phytoplankton values - the carbon signal from benthic microalgae could only be traced in a small fraction of consumer samples (Figures 5.3.2 and 5.3.3). Macrophyte and phytoplankton isotope values on the other hand were closer to the range found in consumers.

Table 5.3.1: Summary of carbon isotope composition ($\delta^{13}\text{C}$ ‰) of major groupings at lower trophic levels in the Gamtoos Estuary.

		Salinity Range			
		0-5 ‰	5-18 ‰	18-35 ‰	all reaches pooled
Benthic Invertebrates	N	13	21	21	55
	min. (max.)	-34.1 (-27.1)	-32.3 (-20.6)	-26.8 (-12.2)	-34.1 (-12.2)
	Mean (s)	-30.1 (-2)	-26.9 (-3.2)	-22.1 (-3.1)	-25.8 (-4.3)
Zooplankton	N	3	6	11	20
	min. (max.)	-31.3 (-25.9)	-29.6 (-23.5)	-25.4 (-21.5)	-31.3 (-21.5)
	Mean (s)	-29.1 (-2.9)	-25.9 (-2.5)	-22.9 (-1.1)	-24.7 (-2.9)
Phytoplankton	N	8	10	3	21
	min. (max.)	-34.3 (-29.9)	-32.7 (-26.6)	-23.7 (-22)	-34.3 (-22)
	mean (s)	-32.5 (-1.4)	-29.5 (-2.1)	-22.8 (-0.8)	-29.7 (-3.6)
Benthic Microalgae	n		3	4	7
	min. (max.)		-21.2 (-17.7)	-19.1 (-14.7)	-21.2 (-14.7)
	mean (s)		-19.4 (-1.7)	-16.7 (-1.8)	-17.8 (-2.2)
Sediment organic matter	n		3	10	13
	min. (max.)		-23.4 (-22.3)	-22.6 (-16.6)	-23.4 (16.6)
	mean (s)		-22.8 (-0.6)	-20.8 (-1.9)	-21.3 (-1.8)
Detritus benthic	n	5	5		10
	min. (max.)	-25.2 (-21.8)	-24.8 (-22.6)		-25.2 (-21.8)
	mean (s)	-24.2 (-1.4)	-23.9 (-0.8)		-24.1 (-1.1)
Macrophytes	n	27	10	4	41
	min. (max.)	-31.1 (-23.2)	-29.2 (-24)	-25 (-23.6)	-31.1 (-23.2)
	mean (s)	-26.4 (-2)	-25.6 (-1.6)	-24.2 (-0.6)	-26 (-1.9)

Table 5.3.2: Taxonomic composition of benthic microalgal communities, extraxted from surface sediment samples. Percentage values refer to proportional abundance of a class at a given site with dominant taxa in bold face. Isotopic composition of benthic microalgae is compared with bulk sediment taken from macrobenthos burrows at the same site.

Phylum / (Division) Class	Genus	Site 1 (cells/ml)	Site 2 <55 µm (cells/ml)	Site 2 >55 µm (cells/ml)	Site 3 (cells/ml)	Site 5 (cells/ml)	Site 6 (cells/ml)	Site 7 (cells/ml)
Heterokontophyta / Bacillariophyceae ('Diatoms')	<i>Achnanthes</i>	0	0	0	1377	0	0	2041
	<i>Amphora</i>	459	0	459	0	14692	0	1020
	<i>Cyclotella</i>	0	0	0	459	0	0	1020
	<i>Cylindrotheca</i>	0	35354	0	0	0	0	0
	<i>Diploneis</i>	0	918	0	0	459	3673	0
	<i>Entomoneis</i>	459	0	0	3214	1377	306	340
	<i>Navicula >20µm</i>	5969	49587	13315	37649	12856	34282	12924
	<i>Navicula <20µm</i>	0	63361	918	3214	0	11325	0
	<i>Melosira</i>	0	1837	0	0	0	0	0
	<i>Nitzschia</i>	0	1837	0	459	1377	612	680
	<i>Pleurosigma</i>	0	0	0	0	0	0	680
		6887	152893	14692	46373	30762	50199	18706
		88.20%	73.00%	4.90%	89.40%	29.30%	98.20%	42.00%

Table 5.3.2 continued

Phylum / (Division) Class	Genus	Site 1 (cells/ml)	Site 2 <55 µm (cells/ml)	Site 2 >55 µm (cells/ml)	Site 3 (cells/ml)	Site 5 (cells/ml)	Site 6 (cells/ml)	Site 7 (cells/ml)
Euglenophyta / Euglenophyceae	<i>Euglena</i>	0	0	0	0	73462	306	0
		0.00%	0.00%	0.00%	0.00%	69.90%	0.60%	0.00%
Cyanophyta / Cyanophyceae (Blue-greens')	<i>Oscillatoria large</i>	0	19284	9183	2296	459	0	0
	<i>Oscillatoria small</i>	0	18365	275482	0	0	0	0
	<i>Chroococcus</i>	0	0	0	0	0	0	21086
		0	37649	284665	2296	459	0	21086
Chlorophyta / Chlorophyceae (Greens')		0.00%	18.00%	95.10%	4.40%	0.40%	0.00%	47.30%
		0	0	0	918	459	306	680
		0.00%	0.00%	0.00%	1.80%	0.40%	0.60%	1.50%
		0	0	0	0	0	0	680
Cryptophyta / Cryptophyceae (Flagellates')		0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	1.50%
		0	14233	0	2296	0	0	3401
Nematodes		0.00%	6.80%	0.00%	4.40%	0.00%	0.00%	7.60%
		918	4591	0	0	0	306	0
Ciliates		11.80%	2.20%	0.00%	0.00%	0.00%	0.60%	0.00%
		-19.1	-14.7	-17	-15.9	-17.7	-19.2	-21.2
Delta 13C - Benthic Microalgae		-18.1	-20.7	-20.7	-22	-22.4		
Delta 13C - bulk sediment		30.6	27.3	27.3	23.4	16	13.7	8.6
Salinity (‰)								

Carbon isotope ratios of the sandprawn *C. kraussii*, varied from -34.1 to -18.9 ‰ (mean $\delta^{13}\text{C}$: -27.4 ± 5.0 ‰; Figure 5.3.2). By comparison, mudprawn (*U. africana*) isotope ratios were shifted towards more ^{13}C enriched values with a mean $\delta^{13}\text{C}$ of -22.9 ± 2.2 ‰. Corophioid amphipods (*Grandidierella lignorum*), had isotopic tissue compositions ($\delta^{13}\text{C}$ range: -34.9 to -23.9 ‰) that overlapped both the phytoplankton and macrophyte producer group (Figure 5.3.3). Pooled $\delta^{13}\text{C}$ values for zooplankton ($\delta^{13}\text{C}$ range: -31.3 to -21.5 ‰) were broadly similar to the benthic crustaceans, and appear to indicate both macrophytes and phytoplankton as carbon sources (Figure 5.3.3).

5.3.4.3 Carbon isotope variation across the estuarine salinity gradient

Carbon isotope ratios of most food web compartments increased with salinity (Figures 5.3.4 and 5.3.5). Among organic matter groups that are potential carbon sources for higher trophic levels, marked shifts towards ^{13}C enriched tissue carbon in more saline habitats are particularly pronounced in phytoplankton (Figure 5.3.4 A), sediment organic matter and benthic microalgae (Figure 5.3.4 B). Carbon isotope ratios of coarse benthic detritus particles did not vary with salinity, although $\delta^{13}\text{C}$ values of the source material (macrophytes) of these deposits increased slightly from oligohaline to euhaline waters (Figure 5.3.4 D).

That carbon isotope composition is a function of salinity was also evident for invertebrate consumers (Figure 5.3.5). All species that occurred over a sufficiently wide salinity range to allow for the modelling of $\delta^{13}\text{C}$ values in relation to salinity, showed a strongly positive relationship. Within the zooplankton, carbon isotope ratios of the calanoid copepod, *Pseudodiaptomus hessei*, rose more steeply with salinity when compared to the mysid, *Rhopalophthalmus terranatalis* (Figure 5.3.5C).

5.3.4.4 Salinity dependence of carbon isotope ratios and trophic pathways

Because carbon isotope ratios of both producers and consumers are a function of salinity (Figures 5.3.4 and 5.3.5), comparisons of $\delta^{13}\text{C}$ values between data sets pooled over different salinity regimes have the potential to seriously confound conclusions about trophic links. Information on systematic variation in carbon isotope ratios across the estuarine salinity spectrum is thus seen as a critical step in tracing material and energy flows.

Carbon signatures of the thalassinid prawns, *C. kraussii* and *U. africana*, were closest to that of phytoplankton over much of the salinity range sampled, with some highly ^{13}C depleted values for *C. kraussii* (Figure 5.3.6). In euhaline (> 30.0 ‰) habitats adjacent to the mouth, isotopic composition of thalassinid prawns resembled that of both sediment organic matter and phytoplankton. There was little evidence to differentiate between sandprawns and mudprawns based on their carbon isotope composition. Indeed, in all habitats where the two species are sympatric (mainly the lower reaches), $\delta^{13}\text{C}$ values of prawn tissues did not differ markedly (Figure 5.3.6).

Isotopic values of benthic amphipods also resembled that of phytoplankton most closely (Figure 5.3.6). There appeared, however, to be additional contributions from more ^{13}C enriched carbon sources, possibly macrophyte detritus. By contrast, $\delta^{13}\text{C}$ values of crown crabs, *Hymenosoma orbiculare*, were markedly shifted towards sediment organic matter and benthic microalgae. Primary carbon sources for the razor clam, *Solen capensis*, appear to be a combination of phytoplankton-derived material and sediment organic matter (Figure 5.3.6).

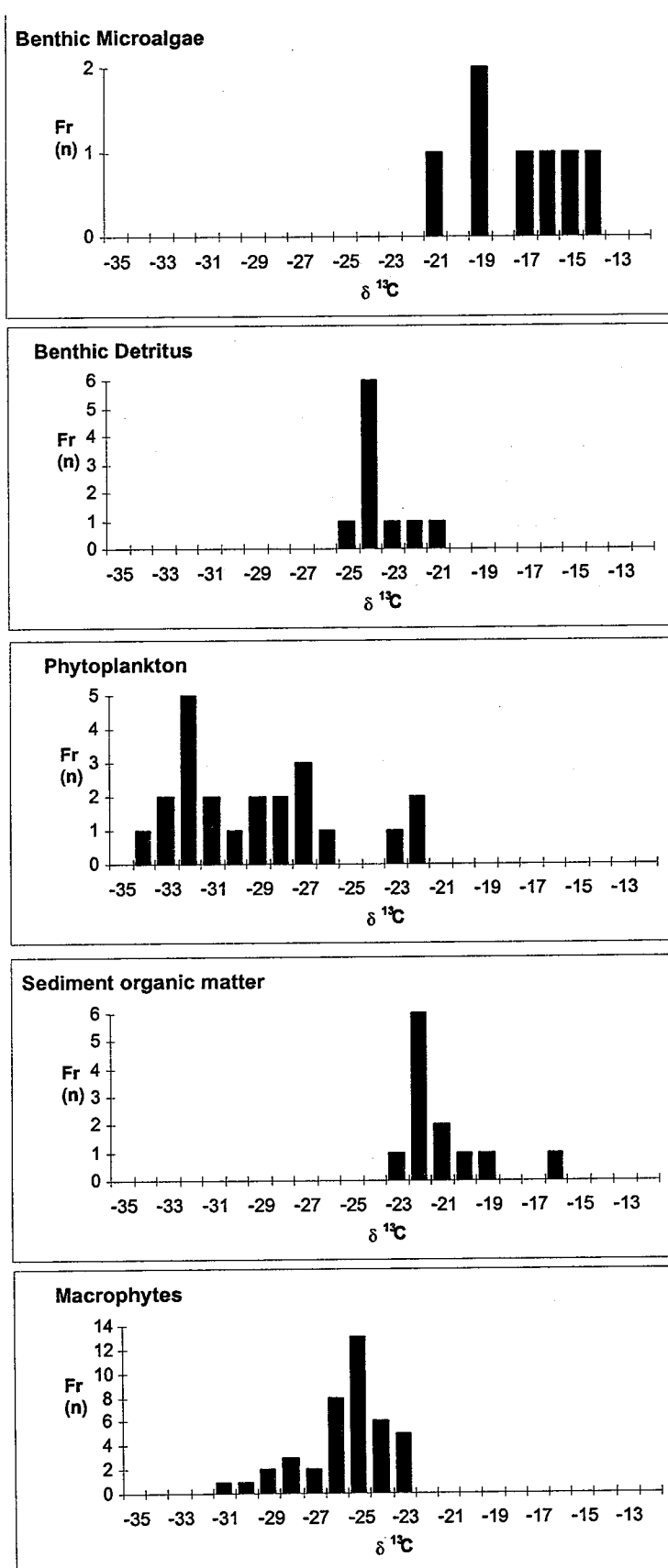


Figure 5.3.1: Frequency (Fr) distribution of carbon isotope ratios ($\delta^{13}\text{C}$ ‰) of five classes of primary food sources and autotrophs in the Gamtoos Estuary. Major classes of primary producers include the small algal component of both surface sediments (benthic microalgae) and the water column (phytoplankton), and vascular plants and macroalgae (macrophytes). Benthic detritus refers to coarse (>1.0 mm) particulate organic material in the bottom sediments, while sediment organic matter refers to the organic fraction of bulk bottom sediment samples.

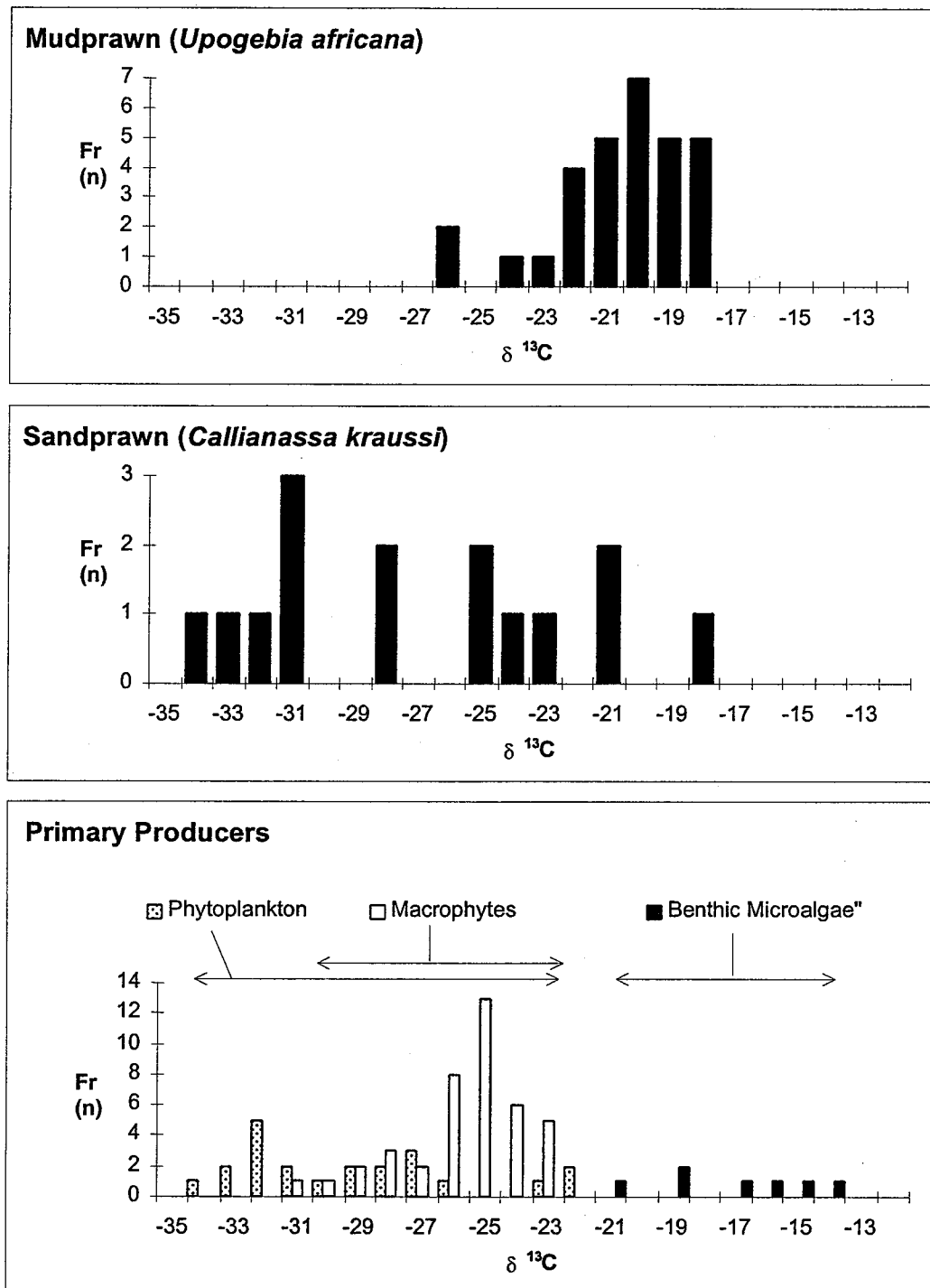


Figure 5.3.2: Frequency (Fr) distribution of carbon isotope ratios ($\delta^{13}\text{C}$ -values) in two species of burrowing anomuran prawns, *U. africana* (top panel) and *C. kraussii* (middle panel). Tissue values in the two species of prawns are compared with potential food sources (bottom panel), to determine trophic modes of these consumers. This approach is based on the empirically established observation that carbon isotope ratios in consumers closely match carbon isotope ratios of their principal diet.

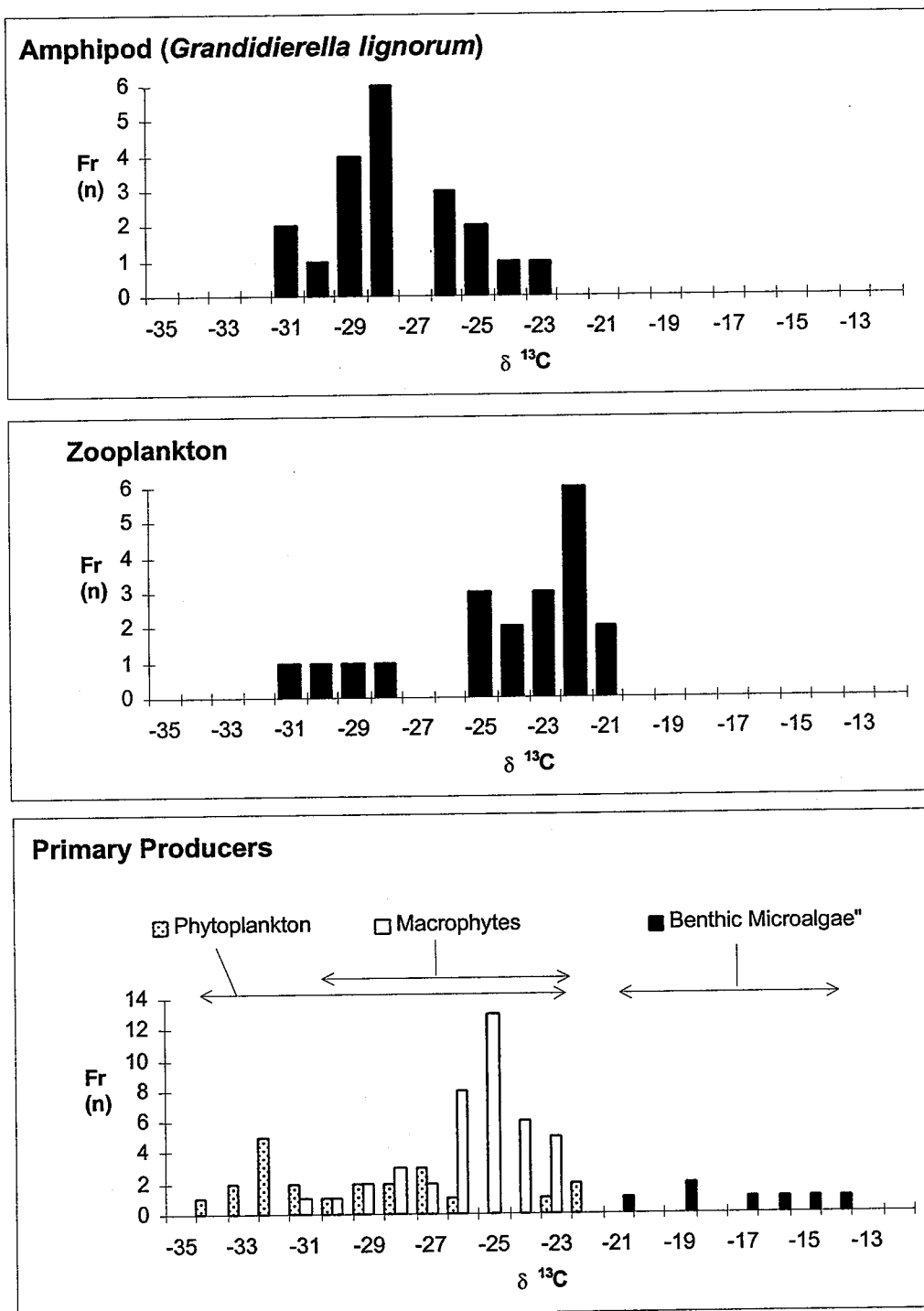


Figure 5.3.3: Frequency (Fr) distribution of carbon isotope ratios ($\delta^{13}\text{C}$ -values) of the benthic consumer, *Grandidierella lignorum* (top panel), and of zooplankton (various species, middle panel, cf. Figure 5.3.5C). To determine principal energy sources of these consumers, carbon isotope ratios of the three main classes of primary producers are plotted for comparison (bottom panel).

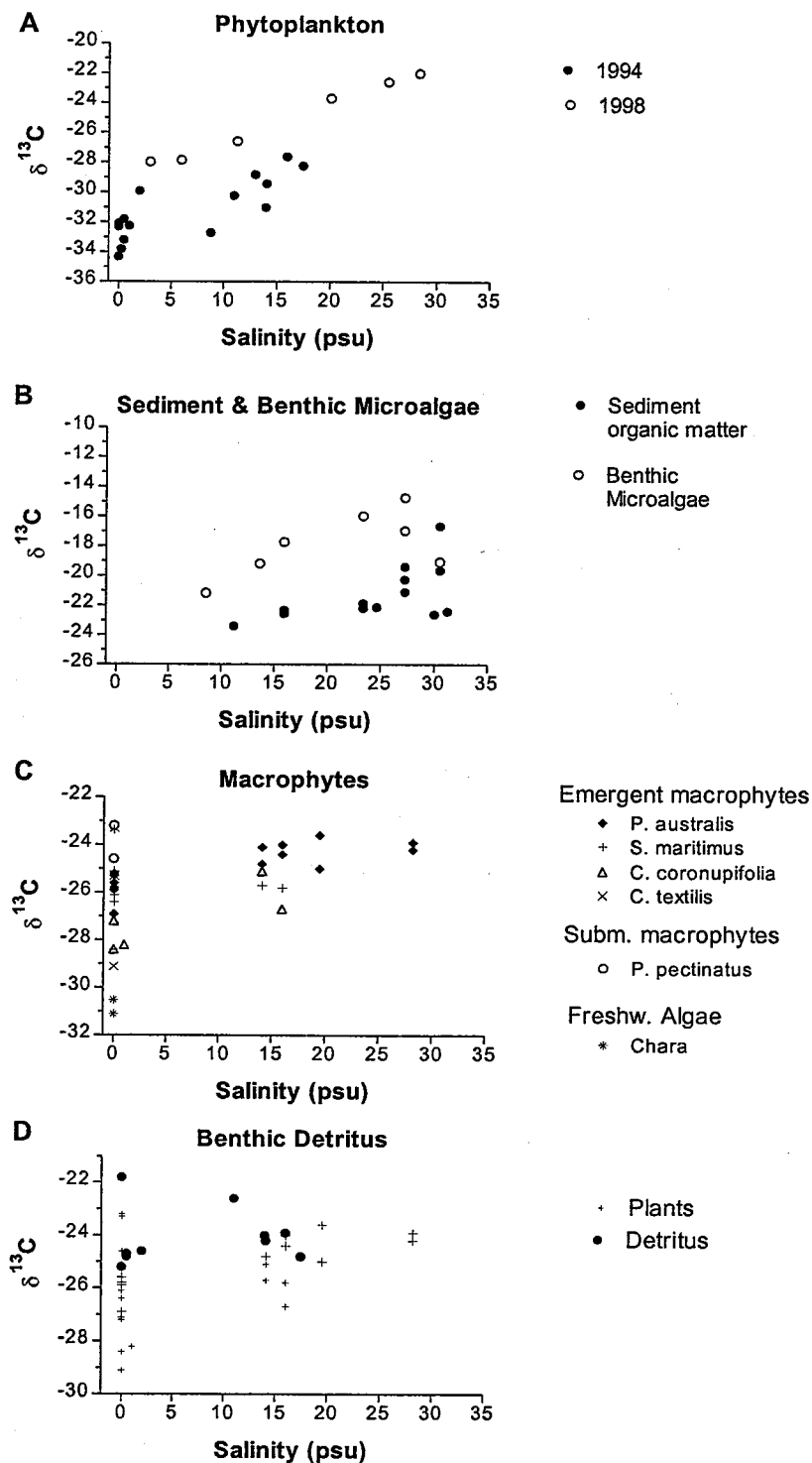


Figure 5.3.4: Isotopic composition ($\delta^{13}\text{C}$ ‰) of primary food sources as a function of salinity (psu = ‰). The main classes of carbon sources for estuarine consumers encompass benthic and pelagic microalgae (A and B), vascular plants and macroalgae (C), unidentified organic material in bulk sediment samples from surface layers and prawn burrows (B), and coarse detrital particles recovered from benthic substrates (D).

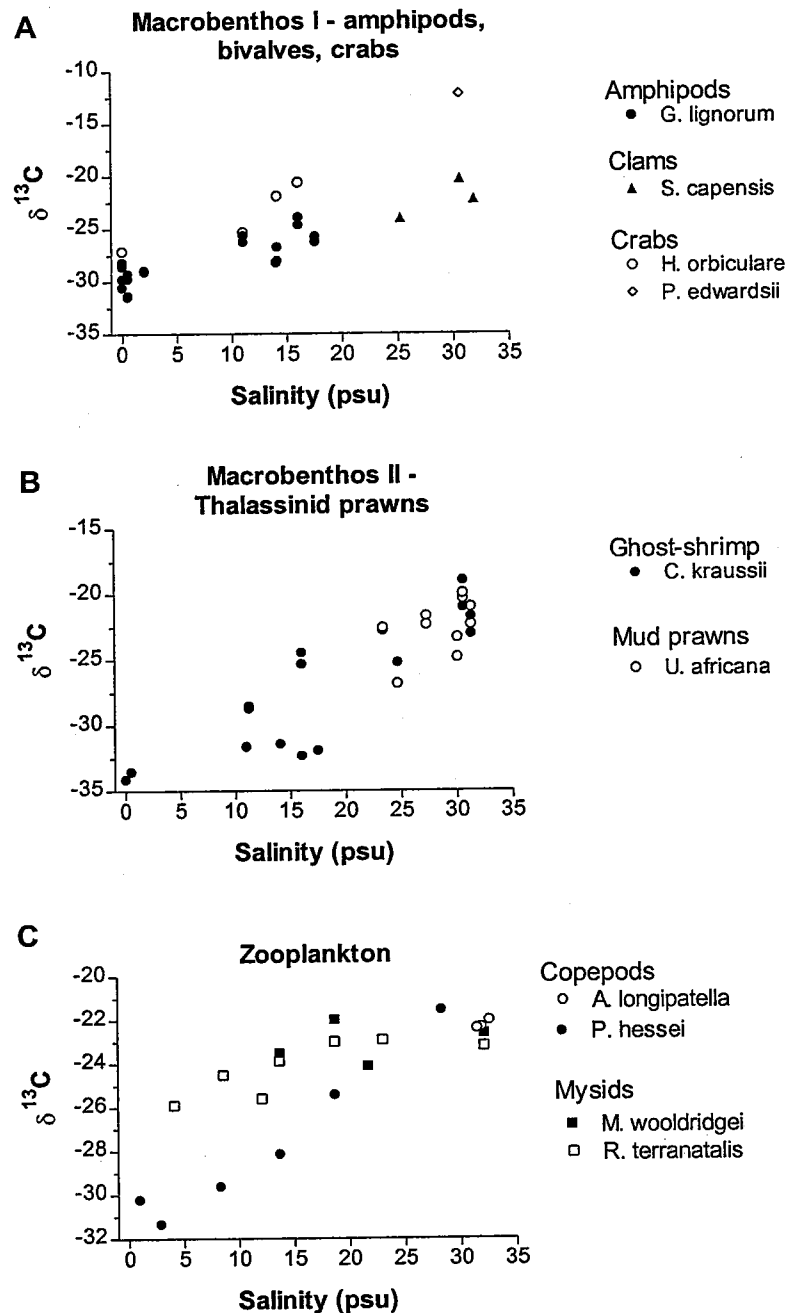


Figure 5.3.5: Isotopic composition ($\delta^{13}\text{C}$ ‰) of estuarine invertebrate consumers as a function of salinity. Consumer species are grouped into benthic (A and B), and pelagic species (C), with two species of anomuran prawns plotted separately from smaller bottom-living species (B). The zooplankton comprises two species of calanoid copepods, which are phytoplankton grazers, and two species of mysids, which are either omnivorous (*M. wooldridgei*) or predatory (*R. terranatalis*). For benthic components, salinity refers to the tidally averaged salt concentration (psu = ‰) of the overlying water.

Carbon isotope values of the calanoid copepod *P. hessei*, mirrored those of phytoplankton, clearly identifying pelagic microalgae as the principal food source of this consumer (Figure 5.3.6). Similarly, in euhaline waters where the copepod *Acartia longipatella* was sympatric with *P. hessei* and two mysid species, $\delta^{13}\text{C}$ -values of *A. longipatella* were similar to those of the phytoplankton-grazing *P. hessei* (Figure 5.3.6). Isotopic tissue composition of mysids appeared to indicate a more varied diet than that of copepods. However, allowing for a 0.5 ‰ trophic fractionation of ^{13}C in estuarine food webs (France and Peters 1997), the primary carbon sources of the predatory mysid species *R. terranatalis* and *M. wooldridgei* can be traced to the smaller zooplankton and phytoplankton components (Figure 5.3.6).

5.3.4.5 Estuarine food web structure - the importance of salinity regimes

Just as the tracking of principal food sources for individual consumers must account for isotopic variability related to salinity changes (Figure 5.3.6), analysis of the estuarine food-web using carbon isotopes should incorporate information on salinity regimes. Consequently, we grouped samples into three salinity ranges based on the classification of the Venice System:

1. Limnetic (< 0.5 ‰) and oligohaline (0.5 to 5.0 ‰) waters. These salinity zones correspond to the geographic subdivisions of 'tidal freshwater section' and 'upper estuarine reaches' respectively.
2. Mesohaline waters (5.0 to 18.0 ‰) which correspond to 'inner estuary', and
3. Polyhaline (18.0 to 30.0 ‰) and euhaline (> 30.0 ‰) waters which correspond to 'middle and lower estuarine reaches' and the 'mouth region' of estuaries (McLusky 1993).

Four distinct features stand out from the above approach of analysing the structure of lower trophic levels in different salinity zones (Figure 5.3.7):

1. Benthic and pelagic invertebrates have similar carbon isotope ratios in each salinity zone, thus indicating that both groups use similar primary carbon sources.
2. There appears to be little trophic transfer of carbon fixed by benthic microalgae to pelagic and benthic crustacean consumers.
3. Phytoplankton appears to be a major source of carbon for both benthic and pelagic consumers.
4. Most major classes of organic matter sources show a marked shift towards more positive $\delta^{13}\text{C}$ values from low salinity reaches towards high salinity waters. This may suggest a relatively low degree of mixing along the main axis the estuary and localised consumption of primary production.

5.3.5 Discussion

5.3.5.1 Salinity as a co-variate in trophic analysis of estuarine systems

The application of pooled frequency data to compare carbon isotope signatures between potential carbon sources and consumers tends to imply several trophic links that run counter to conventional wisdom derived from natural history and known feeding relationships (Figures 5.3.2 and 5.3.3). These include the following:

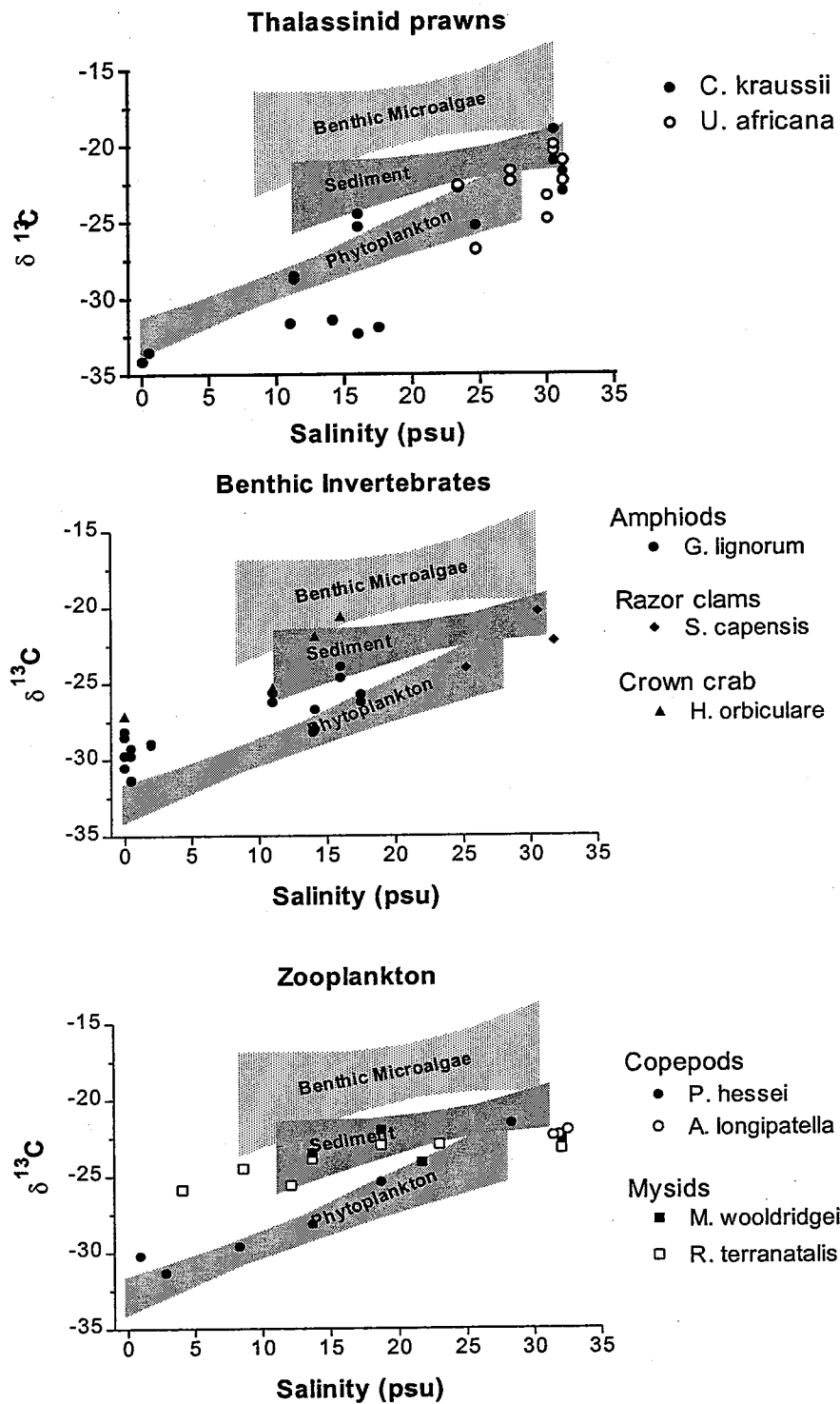


Figure 5.3.6: Comparison of the isotopic composition of the main groups of invertebrate consumers with that of their potential food sources. Shaded areas for carbon sources encompass the 95% confidence limit of linear regressions used to model carbon isotope ratios as a function of salinity (cf. Figures 5.3.4 and 5.3.5).

1. Filter-feeding mudprawns (*U. africana*) appear to be markedly more depleted in ^{13}C compared with their assumed food source phytoplankton.
2. Although some deposit-feeding sandprawns (*C. kraussii*) apparently have carbon ratios that resemble sediment-bound organic material, an appreciable shift towards phytoplankton values is evident.
3. At least two species of the zooplankton, namely *P. hessei* and *A. longipatella*, are phytoplankton grazers, but their carbon isotope ratios seem to be more closely matched by vascular plants (Figure 5.3.3).
4. Benthic tube-dwelling amphipods are ostensibly more depleted in ^{13}C than pelagic crustaceans and their isotopic tissue composition bears little resemblance to the bulk of organic material present in the sediment.

Such interpretations are, however, clearly an artefact of unbalanced sampling coverage across the estuarine salinity spectrum. Because carbon isotope ratios are a function of salinity (Figures 5.3.4 and 5.3.5), any inferences about estuarine trophic pathways are severely confounded by ignoring the salinity dependence of $\delta^{13}\text{C}$ (Table 5.3.3). Misinterpretation of trophic links can, however, be reduced by explicitly incorporating salinity-isotope models into food-web analysis (Figure 5.3.6, Table 5.3.3).

Most of the apparent idiosyncrasies in feeding relationships outlined above fall away when salinity is included as a co-variate in carbon isotope comparisons (Figure 5.3.6, Table 5.3.3). *Upogebia africana* do indeed closely match phytoplankton and not macrophytes, the benthic amphipods, *G. lignorum*, are enriched in ^{13}C , and not depleted, compared with phytoplankton, and there is a remarkably close match between phytoplankton and copepods.

5.3.5.2 Carbon isotope gradients - evidence for localised food webs

The close coupling between the systematic variation of carbon isotope ratios with salinity in primary producers and consumers presents compelling evidence that production and consumption processes are spatially restricted. Consumers tend to assimilate carbon in roughly the same estuarine reaches in which it is fixed by the autotrophs. This local consumption pattern can be clearly traced in different salinity regimes of the estuary (Figure 5.3.7). Although the basic trophic pathway patterns show relatively little spatial heterogeneity between estuarine reaches, the isotopic signal of the primary food sources and consumers shifts systematically from the freshwater to the marine-influenced habitats. Thus, local carbon sources enter higher trophic levels close to the site of production, a finding which concurs with spatial heterogeneity of food webs in estuaries which display higher habitat diversity than our study system (Deegan and Garritt 1997).

Opposing our conclusion about the importance of local production to estuarine energetics, are conceptual models that stress the importance of horizontal advection and tidal mixing of organic matter along the main axis of the estuary. Such models, which focus on significant fluxes of organic matter between different estuarine reaches, make two simple predictions about carbon isotope ratios as a function of salinity:

Table 5.3.3: Confounding of trophic link conclusions arising from a failure to incorporate the salinity dependence of carbon isotope ratios into the identification of primary carbon sources.

Frequency distributions from $\delta^{13}\text{C}$ data pooled across estuarine salinity gradient (Figures 5.3.2 and 5.3.3)	Salinity dependence of carbon isotope ratios integrated into trophic analysis (Figure 5.3.6)	Confounding effect of pooling $\delta^{13}\text{C}$ across salinity
Filter-feeding mudprawns (<i>Upogebia africana</i>) more depleted in ^{13}C than assumed food phytoplankton.	Mudprawn isotopic composition parallels phytoplankton values; some benthic inputs likely.	Severe
Delta ^{13}C of most deposit-feeding sandprawn samples (<i>Callinassa kraussi</i>) match phytoplankton and not sediment-bound organic material.	Sandprawn carbon isotope ratios do indeed correspond to that of phytoplankton; benthic carbon sources negligible.	Severe
Benthic tube-dwelling amphipods (<i>G. lignorum</i>) more depleted in ^{13}C than zooplankton.	Benthic amphipods enriched in ^{13}C compared with copepods but similar to mysids.	Severe and species-specific
Phytoplankton-grazing copepods (<i>P. hessei</i> and <i>A. longipatella</i>) resemble vascular plants and not phytoplankton in isotopic composition.	Close match between isotopic composition of copepods and phytoplankton	Severe

1. Differences in organic carbon signals fixed by aquatic plants from different $\delta^{13}\text{C}$ DIC pools (which is in turn a function of habitat salinity, Fry *et al.* 1992), are predicted to be smoothed by hydrodynamic transport and mixing between estuarine reaches.

2. Consumers feeding over a wide range of salinities are predicted to have isotopic compositions that would reflect average estuarine conditions rather than steep gradients.

Both predictions do not hold true for the present data set, which confirms systematic increases of $\delta^{13}\text{C}$ values with salinity in both producer and consumer groups. The only exception to the local consumption model is macrophyte detritus, which does not vary in isotopic composition with salinity (Figure 5.3.4 A). Macrophyte detritus is, however, energetically not important in this particular food web (Schlacher and Wooldridge 1996a).

The prevalence of localised food webs across the estuarine salinity spectrum suggests that functional processes may spatially be tightly coupled to different community types (see Schlacher and Wooldridge 1996b for zonation of estuarine benthic assemblages across salinity gradients). Thus, different estuarine reaches may not only harbour distinct biological assemblages, but also process material and energy at localised spatial scales. The distinct biological make-up of the REI, together with the concept of 'localised food webs' presented in this section, clearly reinforces the initial hypothesis of this research project that low salinity regions present unique estuarine habitats.

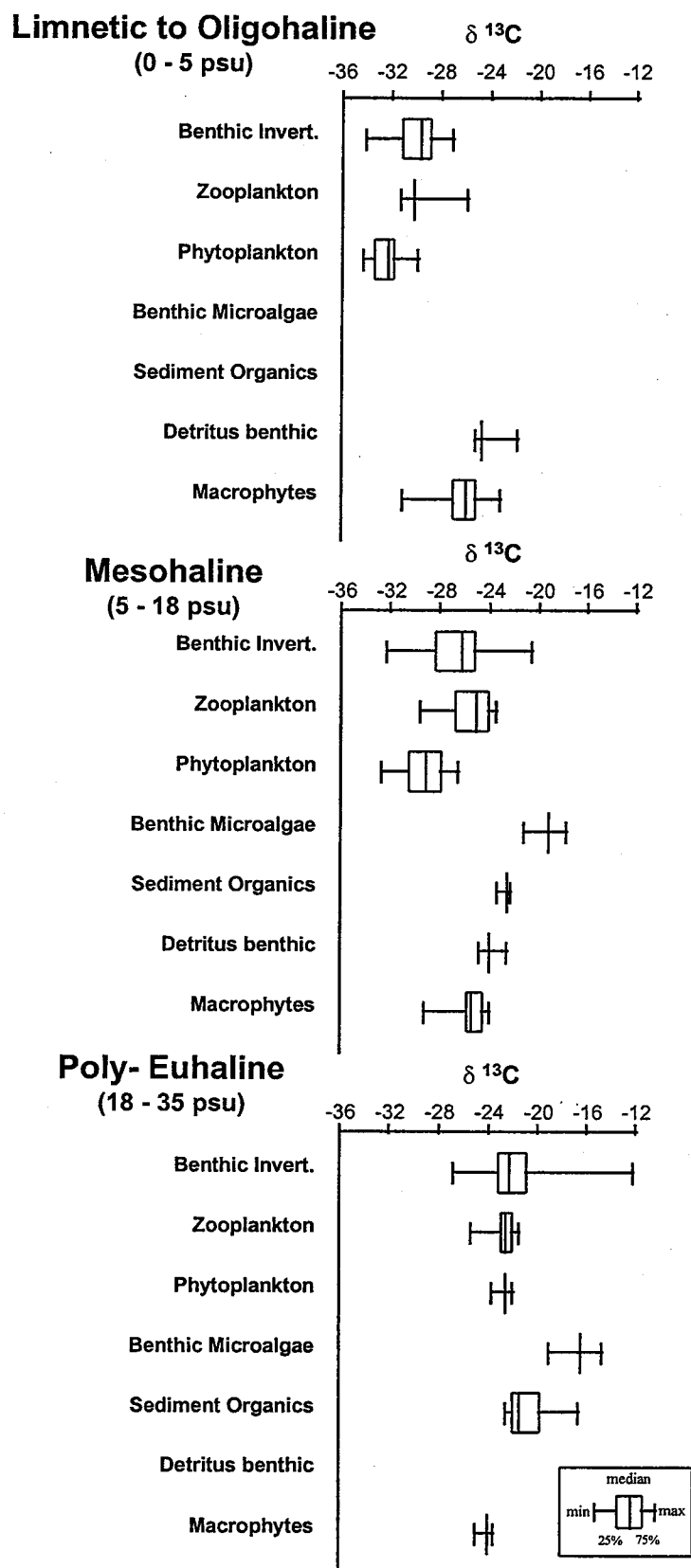


Figure 5.3.7: Isotopic composition ($\delta^{13}\text{C}$ ‰) of the main groups of the lower trophic levels in different salinity regimes (psu = ‰). Limnetic to oligohaline waters correspond roughly to the tidal fresh water and the upper estuarine reaches. Similarly, the mesohaline waters can be equated with the middle reaches, and polyhaline to euhaline salinity conditions with lower reaches and mouth regions of the estuary. Display form is a box & whiskers plot with boxes extending from the 25th to the 75th percentile with a line at the median (50th percentile). Whiskers extend from the lowest to the highest value.

5.4 Boundary conditions and carbon flows in low-salinity estuarine regions

5.4.1 Summary

The upper reaches of estuaries are important regions for bio-geochemical transformations, support high levels of primary and secondary productivity, have enhanced deposition of river borne organic material and harbour ecological communities of distinct structural and functional organisation. Yet, these estuarine areas appear to lack a clear definition of their salinity boundaries. Consequently, this section's first thrust was to characterise salinity zones of upper estuarine regions based on salinity distributions of the major faunal components. Low salinity regions are also important foraging/nursery areas for estuarine fish, but the trophic structure that supports fish production in these areas is not clearly defined. Thus, the second thrust of this module focused on tracing energetic pathways leading to higher trophic levels.

Key research findings in this section included:

- I. While the operational working definition of the boundary that delineates the low-salinity region of estuaries was set at 10.0 ‰ for the purpose of this programme, multivariate analysis of species' distributions across the salinity gradient revealed more diverse patterns of biologically-based salinity zones within this region:
 - A. The underlying structure of salinity distributions of the three major faunal groups (fish, zooplankton and macrobenthos) was heterogeneous at salinities < 10.0 ‰.
 - B. Estuarine regions of the Gamtoos where salinities were < 10.0 ‰, were not characterised by a homogenous set of biological structures and thus do not represent a single zone within the estuary.
 - C. Three biologically-defined subdivisions existed within the low-salinity region which broadly span the freshwater to lower mesohaline salinity classes.
 - D. Boundaries delineating these subdivisions, which constitute the low salinity region, occurred at salinities of 0.5 to 1.0, 2.0 to 4.0 and 7.0 to 9.0 ‰, resulting in three salinity zones for the upper section of the estuary:
 - Zone 1: 0 to 0.5 (1) ‰: tidal freshwater
 - Zone 2: 1.0 to 2.0 (4) ‰: lower oligohaline
 - Zone 3: 3.0 to 4.0 ‰ to 7.0 to 9.0 ‰: lower mesohaline
- II. Carbon in pelagic fish predators was sourced from benthic microalgal production. The transfer of microalgal carbon to higher trophic levels was channelled via two steps, namely grazing of benthic microalgae by mugilids, and consumption of these species by piscivores. Conversely, both benthivorous and zooplankton-feeding fish comprise only a minor contribution to the top predators' energy intake. These results stress the energetic importance of benthic microalgae in supporting piscivorous fish production within the Gamtoos estuarine environment.

- III. In addition to benthic microalgae, which formed a clear energy transfer route to fish, a second energetic pathway that originates from phytoplankton was also important for higher trophic levels. Benthic invertebrates that assimilate mostly phytoplankton made a substantial contribution to the diet of benthic-feeding fish that were abundant in the upper section of the estuary. Similarly, zooplankton grazing on phytoplankton represented the intermediate trophic step that links pelagic primary production with pelagic zooplankton-feeding fish.
- IV. The structure of this food web argues forcefully for the pivotal role of microalgae in the energetics of this estuarine compartment. Since freshwater inflow promotes productivity of both benthic and pelagic microalgae in estuaries, supply rates of freshwater to estuaries are clearly a dominant structuring force for estuarine communities at several trophic levels.

5.4.2 Introduction

5.4.2.1 Objective 1: Boundary definitions of low-salinity estuarine regions

The region where the upper section of an estuary meets the lower riverine reaches appears clearly identifiable by rapid changes in hydrochemistry, particularly the ratios of the major ions and concentrations of the total suspended and dissolved solids (Morris *et al.* 1978; Fauzi and Mantoura 1987). Steep gradients in the physico-chemical properties of this habitat are matched by marked changes in the make-up of the biological communities inhabiting these transition zones.

In the context of this programme, we have already shown that assemblages of the REI region have a distinct biological structure (species composition, diversity and trophic group prevalence) compared with estuarine regions, and that ecosystem processing of energy may be largely confined to specific sections of an estuary. While the spatial extent of salinity zones varies as a result of variation in freshwater inflow (Schumann and Pearce 1997), the biological uniqueness of the assemblages in the low-salinity regions remains essentially intact.

Sharp longitudinal haloclines delineated the freshwater and mesohaline reaches during previous surveys in the Gamtoos Estuary. Minimum salinities in areas that held typical estuarine communities were 12.5 ‰, but 2.5 km further upstream fell sharply to 0.2 ‰ in regions harbouring assemblages that characterise the REI. A salinity value of 12.5 ‰ is indeed comfortably close to the working definition of 10.0 ‰ that was used in this programme as the boundary between the REI region and the estuary. It is, however, not unreasonable to assume that patterns and subdivisions do exist at finer scales, and that the true limit of the interface zone may lie within the above salinity range (0.2 to 12.5 ‰). Thus, although the biological uniqueness of the low-salinity zone could be unambiguously established, its exact spatial extent and its salinity boundary conditions required a closer definition. Consequently, a primary thrust of this invertebrate survey was to improve the resolution of the salinity boundary values that delineate the interface region.

5.4.2.2 Objective 2: Carbon pathways to estuarine fish

The structure of carbon pathways in South African estuaries appears to be closely linked to hydrodynamic regime and geomorphology, which in turn determine the dominant classes of primary producers. Estuaries with well developed inter- and supra-tidal areas and small volumes of freshwater inflow support extensive stands of macrophytes (seagrasses and saltmarshes) which form the primary carbon sources for consumers at several trophic levels (Paterson and Whitfield 1997). By contrast, in channel-like estuaries with poorly developed intertidal areas and which receive substantial amounts of freshwater inputs, the bulk of primary production is centred on microalgae (Adams and Bate 1999). Although stands of fringing macrophytes may be a prominent feature in the upper, low-salinity sections of these channel-like estuaries, carbon fixed by these macrophytes appears to be energetically unimportant for higher trophic levels (Jerling and Wooldridge 1995). Food webs in estuaries characterised by substantial freshwater input may instead revolve around microalgae (phytoplankton and microphytobenthos) and not higher plants (Schlacher and Wooldridge 1996c).

While the principal carbon pathways could be clearly identified for plants and invertebrates, carbon sources for consumers at higher trophic levels have not been adequately identified to date. An understanding of food web structures is, however, seen as an important management tool, as estuaries are crucial nursery and foraging areas for fish (Whitfield 1994). This may be especially pertinent for low-salinity regions of estuaries, as these areas provide crucial habitats for the juveniles of important marine linefish species such as dusky kob (*Argyrosomus japonicus*) and spotted grunter (*Pomadasys commersonnii*) (Ter Morshuizen *et al.* 1997).

Given the importance of low-salinity estuarine regions as foraging/nursery areas for fish, the second thrust of this section of work was to determine the broad outline of food web structures at higher trophic levels. This second task was a joint programme between the Department of Zoology, University of Port Elizabeth and the J.L.B. Smith Institute of Ichthyology.

5.4.3 Methods

5.4.3.1 Characterisation of boundaries of low-salinity regions

Analytical framework

In this section, identification of estuarine salinity zones and their boundaries is derived from biological information on the biota's field distributions. The key criterion for the identification of salinity zones was similarity in species composition amongst waters of different salinities. Thus, the prime focus was on extracting the underlying biological structure in species' distribution patterns across the estuarine salinity spectrum. The principles of this analytical approach not only implicitly underpin classical schemes that subdivide estuaries into salinity zones based on the distribution of species along the salinity gradient, but have recently been applied in a more rigorous and objective way to derive 'biologically-based' estuarine salinity zones (Bulger *et al.* 1993).

Data Sources

We used four sets of data that comprise multiple taxa (zooplankton, macrobenthos and fish), multiple spatial scales (from estuarine reaches to regions), and multiple temporal scales (from tidal cycles to evolutionary) to extract common patterns of distribution across the salinity range (Table 5.4.1). In the context of this analysis, spatial scale is defined as the geographic area over which data on species distributions in estuaries were collected. Temporal scale is defined as the time-frame over which the processes that produce the observed patterns operate. Because a single process does not govern distribution of species across the salinity gradient, the magnitude of temporal scale encompasses a range of values that depend on the combination of at least three factors:

1. Variability of the physical forces (tides, freshwater inflow, etc.) that produce salinity conditions to which species are exposed.
2. The rate at which species respond to changes in salinity, and their mobility.
3. The degree to which species have evolved physiological adaptations to varying salinities.

For example, the greater mobility of zooplankton and fish compared with benthic invertebrates would tend to shorten temporal scales of a given data set as species can adjust their exposure to changed salinity fairly readily. By contrast, infaunal forms are more likely to integrate salinity conditions of their habitat over longer time periods. However, all species' salinity ranges do have at least an evolutionary component in their time scales, as physiological adaptations are pivotal for the colonisation of estuarine habitats. Three of the four data sets (2x macrobenthos and 1x zooplankton) originate from our own field work in the Gamtoos Estuary, while the salinity ranges of fish recorded from southern African estuaries are taken from Whitfield (1998) (Table 5.4.1). Details of macrobenthic sampling protocols are given in Schlacher and Wooldridge (1996a). In brief, three surveys (1993, 1994 and 1996) were conducted in which the species composition and abundance of subtidal macrofauna was quantified at each of 12 sites from three replicated van Veen grabs. Sampling sites were situated along the main axis of the estuary and spanned the full estuarine salinity gradient from near-marine conditions (34.0 ‰) near the mouth to the tidal freshwater section. Because the primary focus of this study was the low salinity regions of estuaries, an additional set of longitudinal macrobenthos data was sampled in the upper section of the Gamtoos Estuary (subsequently referred to as 'low-salinity macrobenthos'). Spatial coverage was the upper 6.0 km of the main estuarine channel with nine sites spaced 300 to 800 m apart, encompassing a salinity range from 1.0 to 21.0 ‰ (Table 5.4.1). Zooplankton field collections followed the protocol detailed in Schlacher and Wooldridge (1995). As was the case for macrobenthos, sampling sites were spaced along the main axis of the estuary to cover the salinity gradient. Spatial coverage in the upper sections of the estuary was more restricted than for macrobenthos due to non-navigable waters during the nocturnal plankton collections, resulting in eight sites for the estuary. Mesozooplankton (> 0.2 mm) was collected monthly at each site for 26 months from February 1989 to March 1991, yielding a total of 208 (8 sites x 26 months) density estimates for zooplankton. During all field surveys, vertical salinity profiles were measured at each site immediately preceding the faunal sampling.

Data Analysis

All analyses used to extract commonality in species distributions across the salinity spectrum are based on identification of similarity patterns in community structure amongst samples. The multivariate statistical approach (PRIMER) of Clarke (1993) was employed. Species distributions across the salinity gradient were analysed in two complementary ways:

1. The two data sets on macrobenthos were analysed from a conventional species x sites matrix, where sites represent sampling localities for which salinities were measured; matrix cells contained abundances of each species per site and sampling time. The output displays the relationship between sites based on their biological structure. This analysis always retains site identification (geographic locality), and any inferences about salinity zones are made by linking salinity information with site groupings in a second step.
2. For zooplankton and fish records, the computational steps are identical but start from a different data matrix: 'sites' are replaced by salinity increments, giving rise to a matrix of species x salinity increments. Salinity increments are integer steps of one from 0.0 to 35.0 ‰, plus an additional step for hypersaline conditions (> 35.0 ‰). For the zooplankton data, the first salinity increment was divided into two ranges of 0.0 to 0.5 ‰ and 0.5 to 1.0 ‰ to improve resolution at very low salinities and allow for comparison with the traditional Venice system; these increments could not be used for the fish data as the resolution of the field records was not adequate below values of 1.0 ‰. Each fish species scored a one if recorded in a salinity increment and a zero in salinity increments outside their reported range, resulting in a presence/absence matrix. Abundance of each of the dominant zooplankton species in each salinity increment was averaged from the 26 months data set, yielding a matrix of mean density (individuals m⁻²) per species in each of 36 salinity increments (0.0 to 0.5, 0.5 to 1.0 ‰ and integer values of step size one from 1.0 to 35.0 ‰). Conceptually, this statistical analysis yields the same results as for a conventional species x site matrix, but identifies salinity zones over which the species composition shows a higher degree of concordance compared with other zones. Thus, site labels in the conventional ordination plots are replaced by salinity values of the increments used to construct the matrix (see also Bulger *et al.* 1993).

5.4.3.2 Food web analysis

The principal energy pathways in the estuarine ecosystem were traced with stable carbon isotopes. The method rests on the principle that carbon isotope ratios ($\delta^{13}\text{C}$) of food sources are closely (apart from a small shift of 0.5 ‰ per trophic transfer) mirrored by consumers utilising a particular carbon source. If the food sources that are potentially available to consumers differ in their $\delta^{13}\text{C}$ signals, then inferences can be made about the type of carbon assimilated by a consumer. Phytoplankton, benthic microalgae and macrophytes have clearly distinct carbon signatures, and thus their relative importance as carbon sources to higher trophic levels can be estimated. For applications of this method in South African estuaries see Jerling and Wooldridge (1995), Schlacher and Wooldridge (1996c) and Paterson and Whitfield (1997). Field collection and sample preparation protocols are detailed in sections 5.2 and 5.3 of this report.

Table 5.4.1: Summary information on data used in the definition of estuarine salinity zones in the present study. *'Spatial scale' is here defined as the geographic range over which data were collected. 'Temporal scale' is the time-frame in which the processes that produce the observed patterns operate. In the case of species' distribution in different salinities, temporal scale can be any combination of: (1) the variability of the physical processes (e.g. tides, freshwater inflow) that produce salinity conditions in which biota live, (2) the response times and mobility of biota to changes in physical conditions, and (3) the evolution of physiological adaptations in species to varying salinity values.

Data set	Spatial scale (*)	Temporal scale (*)	Salinity range	Data type	Spatio-temporal replication	Source
1 - Low-salinity macrobenthos	Landscape element - low-salinity section of individual estuary (Upper Gamtoos Estuary, 6 km)	Days to weeks (variable, depending on freshwater inflow regime)	1 to 21 ‰	Zonation of community structure of subtidal macrofauna along the upper, low-salinity section of an estuary.	One macrobenthos survey in meso- and oligohaline waters; 9 sites.	This study
2 - Macrobenthos zonation	Landscape - individual estuarine system (Gamtoos Estuary, 21 km)	Days to months (variable, depending on freshwater inflow regime)	0 to 34 ‰	Zonation of community structure (species abundances) of subtidal macrofauna along the full estuarine salinity gradient.	Three sets (1993, 1994, 1996) of macrobenthos surveys along the length of the estuary at each of 12 sites.	This study and Schlacher & Wooldridge (1996a)
3 - Zooplankton salinity distribution	Landscape - individual estuarine system (Gamtoos Estuary, 21 km)	Tidal cycle to years	0 to 35 ‰	Mean abundance of dominant mesozooplankton species in salinity ranges of 0 to 0.5, 0.5 to 1, and 34 subsequent salinity classes of interval size 1 (1 to 35).	26 sets of abundance data (monthly sampling from February 1989 to March 1991) at each of 8 sites located along the length of the estuary.	This study
4 - Salinity ranges of estuarine fish	Region - (1000s of km)	Decades to evolutionary	0 to > 35 ‰	Salinity ranges of estuarine-associated fish species in southern Africa.	Long-term records (decades) from field surveys.	Whitfield (1998)

5.4.4 Results

5.4.4.1 Faunal salinity ranges

Macrobenthos

Benthic macrofaunal assemblages of the low-salinity regions are structurally distinct from assemblages in more saline waters (Figure 5.4.1). In the three longitudinal surveys of macrobenthos undertaken, the most landward located three to four sites were always situated in waters with bottom salinities of 0.2 to 0.5 ‰, while salinities at the next downstream site increased abruptly to between 12.5 and 23.5 ‰. Thus, a sharp longitudinal halocline separated the tidal freshwater reaches from mesohaline to polyhaline waters over a short distance of 2.5 km, and coincided with a marked faunal discontinuity. While it may be convenient to fix the salinity boundary between the tidal freshwater reaches and the estuarine reaches at a salinity of 12.5 ‰ (which represents the minimum salinity recorded in areas with a distinct estuarine community) it is equally likely that the true boundaries between upper sections of the estuary may be located within the above salinity range (0.5 to 12.5 ‰) and that additional biological subdivision may exist over this range.

Data on trends in community properties and structure across the length of the upper estuary do indeed provide some support for the above hypothesis that benthic communities within the low salinity regions are not homogenous (Figures 5.4.2 and 5.4.3). Benthic invertebrate densities declined steadily and significantly from polyhaline to oligohaline waters (Figure 5.4.2A), while no clear longitudinal trend was evident for species richness, diversity or evenness over the range of salinities sampled (Figures 5.4.2B to D). However, communities situated in waters of very low salinity are more diverse even though the total number of species dropped (Figure 5.4.2B and C). This rise in diversity can be attributed to the lower numerical abundance of estuarine corophioid amphipods and sabellid polychaetes that dominate the communities further downstream. Conversely, freshwater taxa such as chironomid larvae and oligochaetes are only found in substantial numbers in the lowest salinity areas. The penetration of freshwater taxa into the upper section of the estuary gives rise to a marked change in overall community structure between habitats of 1.0 and 4.0 ‰ (Figure 5.4.3). It thus appears that the upper salinity boundary of assemblages in the very low-salinity regions may lie close to 1.0 ‰ (the upper boundary of this zone could not be clearly defined as the salinity increment to the next site was 3.0 ‰). A second, albeit less prominent, subdivision in the fauna appears evident when progressing from oligohaline waters to the meso-/polyhaline estuarine reaches (Figures 5.4.3B and C). Communities in habitats with salinities between 4.0 and 9.0 ‰ form a distinct cluster compared with the remainder of the upper estuarine section (Figure 5.4.3). However, this biological grouping cannot be clearly matched with a continuous salinity range, as communities of 7.0 ‰ are more similar to those in the more saline reaches. Despite this slight inconsistency, indications are that benthic assemblages of the oligohaline reaches may differ from the estuarine fauna further seawards - the boundary between these two zones being a band of salinities between 7.0 and 9.0 ‰. Thus, the biological structure of macrobenthos is not uniform at < 12.5 ‰, but comprises at least two salinity zones with approximate ranges from 0.0 to 1.0 ‰ and 4.0 to 9.0 ‰.

Zooplankton

The underlying structure of zooplankton distribution patterns across the estuarine salinity gradient comprises three distinct salinity zones, namely waters of < 1.0 ‰, oligohaline waters of 1.0 to 4.0 ‰, and the meso/polyhaline waters of 5.0 to 35.0 ‰ (Figure 5.4.4). The marked salinity boundaries of 1.0 and 4.0 ‰ correspond largely with the distributional limits of individual species. The copepod *Pseudodiaptomus hessei* was the only species penetrating into fresh water (Figure 5.4.5A), while the two mysid species (*M. wooldridgei* and *R. terranatalis*) which make up the bulk of the larger zooplankton component have a lower salinity limit of 1.2 ‰ (Figure 5D and E). The salinity boundary of the second zone at 4.0 ‰ appears to be largely set by the lower salinity range limit of the most abundant copepod species, *Acartia natalensis* (Figure 5.4.5C). By contrast, the congeneric *A. longipatella* reaches peak densities in near-marine waters in the vicinity of the estuarine mouth, with abundances dropping sharply towards the upper sections to a lower range limit of 8.0 ‰ (Figure 5.4.5B).

Estuarine Fish

Salinity ranges of the estuarine-associated fish fauna show a primary distributional structure of five salinity zones (Figure 5.4.6). Zone 1 identifies the freshwater sections of estuaries, and has the lowest fish diversity of all estuarine regions, only 23.0 % of marine teleosts having been recorded in salinities of < 1.0 ‰, and 61.0 % of estuarine teleosts (Table 5.4.2, Figure 5.4.7). The division of fish into marine and estuarine groups rests on the primary spawning locality (marine vs estuarine; Whitfield 1998), but does not necessarily reflect the extent to which fish species utilise estuaries.

Euryhaline marine teleosts which make extensive use of southern African estuaries as foraging and nursery areas comprise the majority of species found in the region (61 species or 39.0 % of total ichthyofaunal diversity). Zone 2 comprises the 1.0 to 2.0 ‰ salinity range (Figure 5.4.6), and is characterised by a marked increase in the number of marine (35 to 37 species) and estuarine (21 to 23 species) teleosts when compared with the freshwater section (Table 5.4.2, Figure 5.4.7). However, total species richness of zone 2 is still substantially less than zone 3 which comprises the next salinity range of 3.0 to 7.0 ‰ (Figures 5.4.6 and 5.4.7, Table 5.4.2). Apart from a single marine species, *Pomadasys olivaceum*, all marine and estuarine teleosts are capable of occupying salinities of 8.0 ‰ (Table 5.4.2, Figure 5.4.7), with the range of 8.0 to 35.0 ‰ constituting salinity zone 4 (Figure 5.4.6). About half of the estuarine fish species have been recorded from hypersaline waters (zone 5), with only one freshwater species penetrating waters with salinities > 35.0 ‰ (Figure 5.4.6).

The pattern of five distinct salinity zones for the fish fauna appears largely driven by marine teleosts (Figures 5.4.8A, 5.4.10). As stated earlier, the marine component makes up the bulk of estuarine fish diversity in the region. Salinity ranges of estuarine teleosts do, however, follow a similar underlying pattern to marine teleosts, but have a compressed range of salinity zones (Figure 5.4.8B). While the freshwater and 1.0 to 2.0 ‰ salinity range form distinct zones, no salinity boundaries are evident between 3.0 ‰ and marine conditions (Figure 5.4.8B). This can be traced to the majority (26 species, 93.0 %) of estuarine teleosts being capable of penetrating up-estuary to a lower salinity limit of 3.0 ‰ (Table 5.4.2, Figure 5.4.10).

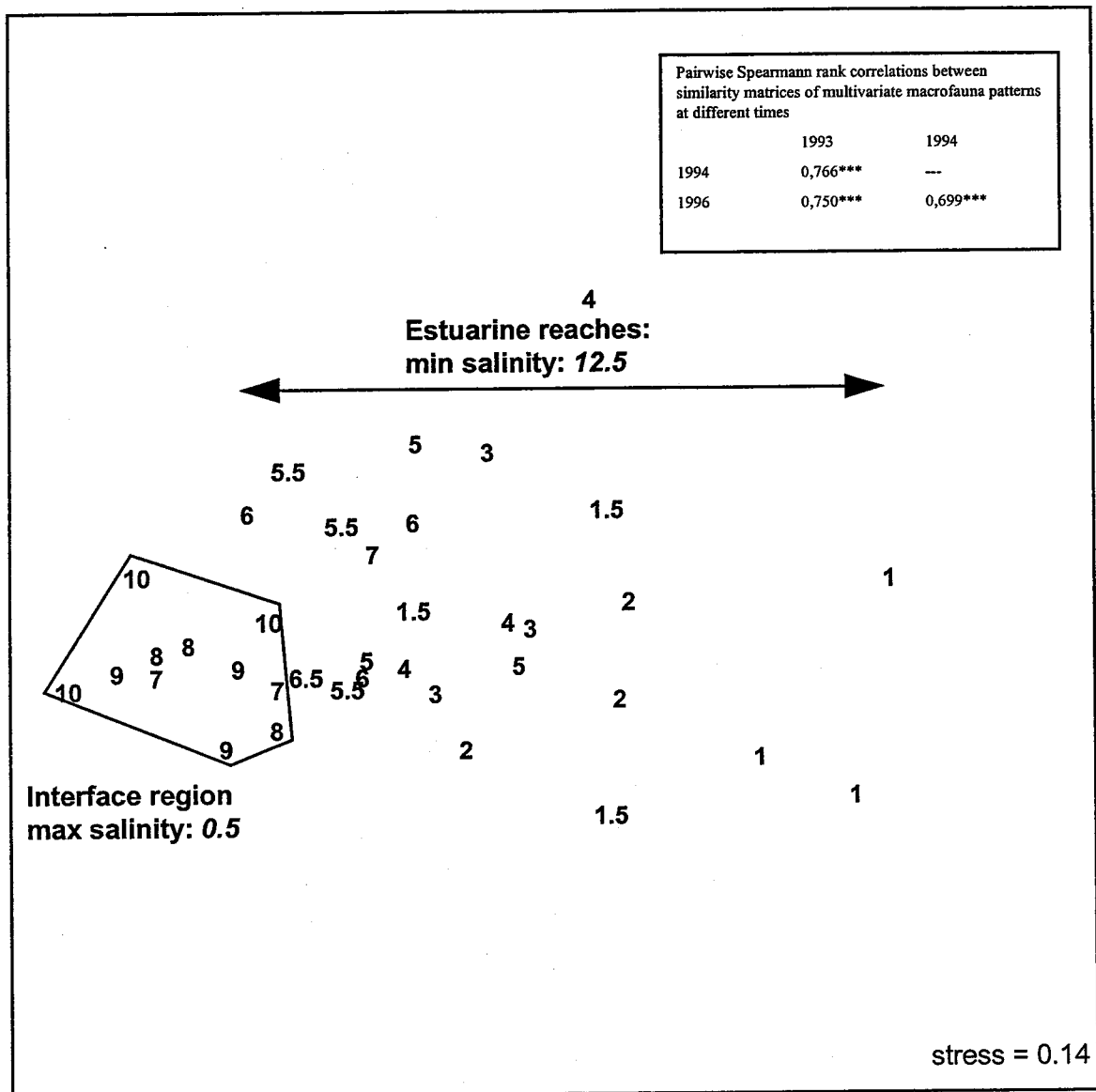


Figure 5.4.1: Ordination of benthic macrofaunal communities recorded during three (1993, 1994 and 1996) axial surveys in the Gamtoos Estuary. During each survey, 12 sites were sampled which spanned the salinity gradient from near-marine waters bordering the tidal inlet at salinities of 35.0 ‰ (site 1) to habitats dominated by riverine freshwater input at the tidal head of the estuary at salinities of < 1.0 ‰ (site 10). Correlation coefficients in the top right insert denote the very high degree of agreement in zonation patterns amongst the surveys. Sites encircled by the polygon to the left represent assemblages of the REI region where the maximum salinity value recorded was 0.5 ‰ (modified from Figure 5.1.4 of this report)

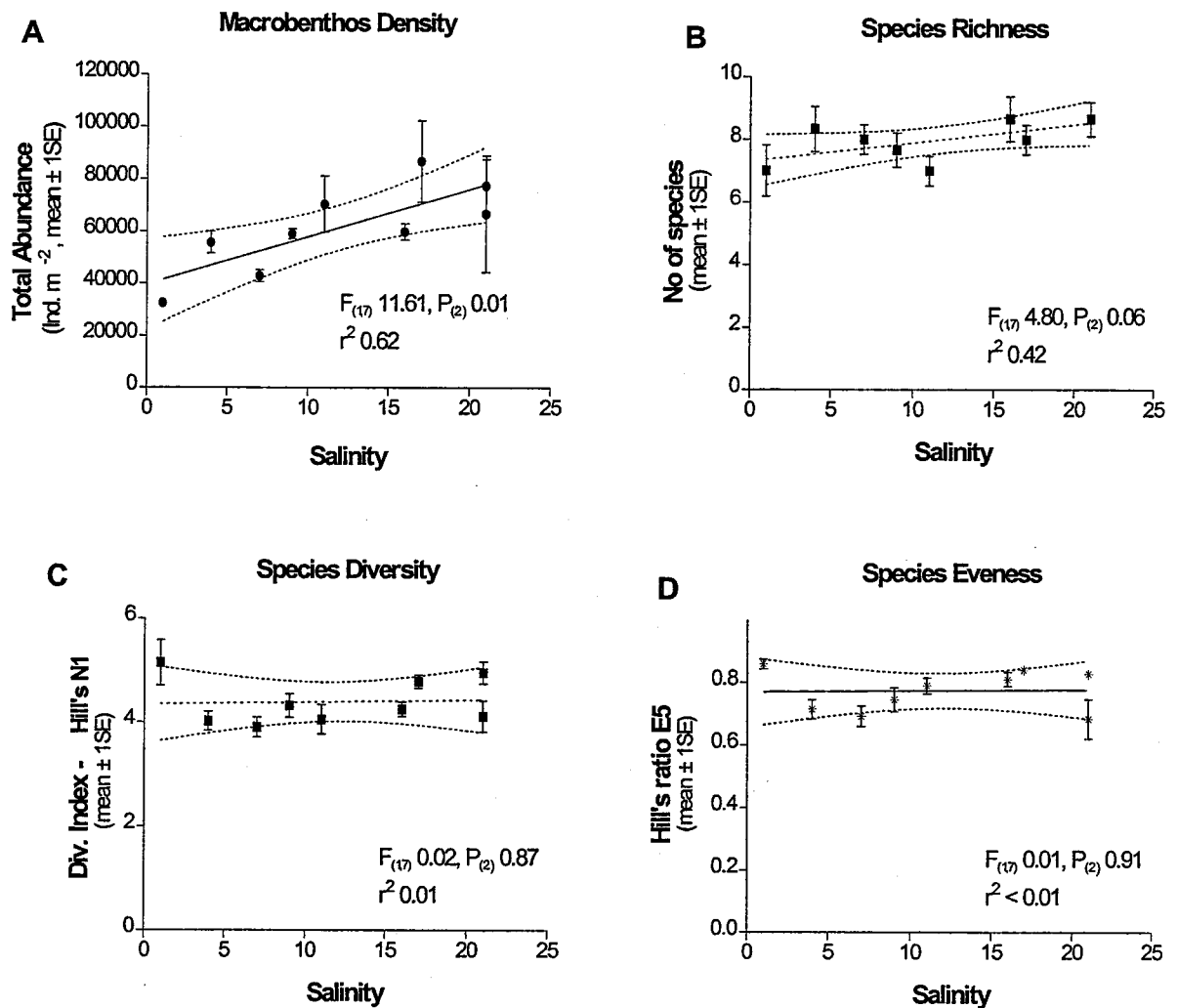


Figure 5.4.2: Patterns of macrobenthos community measures across the salinity gradient in the upper section of the Gamtoos Estuary. Benthic macrofauna was sampled at 9 sites that were positioned across the REI, encompassing a salinity gradient between 21.0 ‰ at site i1 and 1.0 ‰ at site i9 (cf. Figure 5.4.3). Univariate and composite measures of community structure include: density of all fauna > 1.0 mm body size (A), the total number of species per site (B), species diversity as measured by Hill's N1 index (C), and evenness as the modified Hill's ratio (D). Reported values are means of three 110.0 mm $\pm$ cores per site with one standard error. F- and P-values refer to ANOVA testing of the slope of fitted linear regressions, with dotted lines denoting 95 % confidence bands.

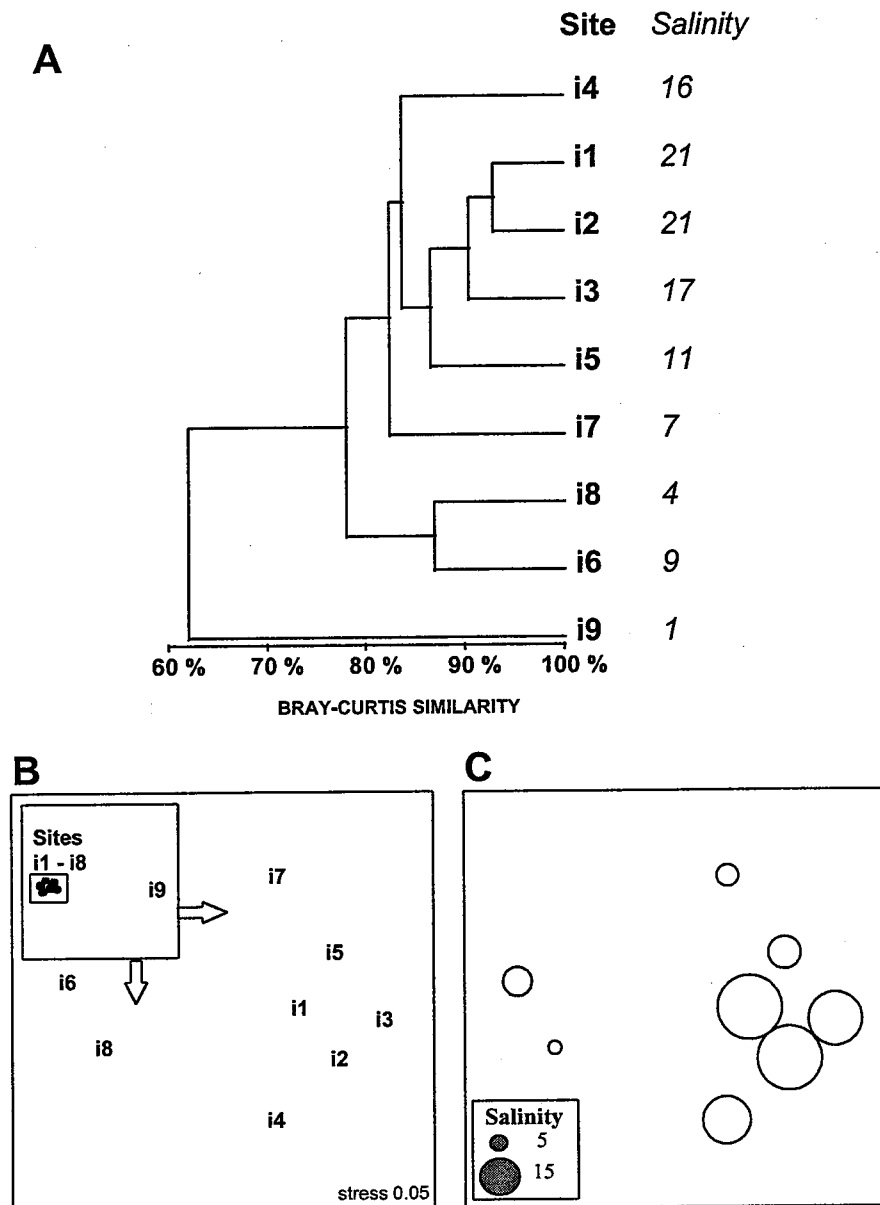


Figure 5.4.3: Classification (A) and ordination (B) of macrobenthic assemblages in the upper sections of the Gamtoos Estuary. The prefix "i" ("interface") for site labels indicates the position of sampling sites near the interface of the freshwater and estuarine section, with the most seaward site i1 located in salinities of 21.0 ‰, while the furthest upstream site i9 was situated in waters of 1.0 ‰. The geographical position of site i1 corresponds to site 7 of the main survey (*cf.* Figure 5.4.1) and site i9 corresponds to site 10. Spatial sampling coverage in this module was a 6 km long section of the main estuarine channel with neighbouring sites spaced *ca.* 300 to 800 m apart. Figure C is the site ordination shown in B, with scaled salinity values superimposed (analysis based on $x_i = x_i^{0.25}$ transformed abundance values of the full macrobenthic species set and Bray Curtis similarity measure).

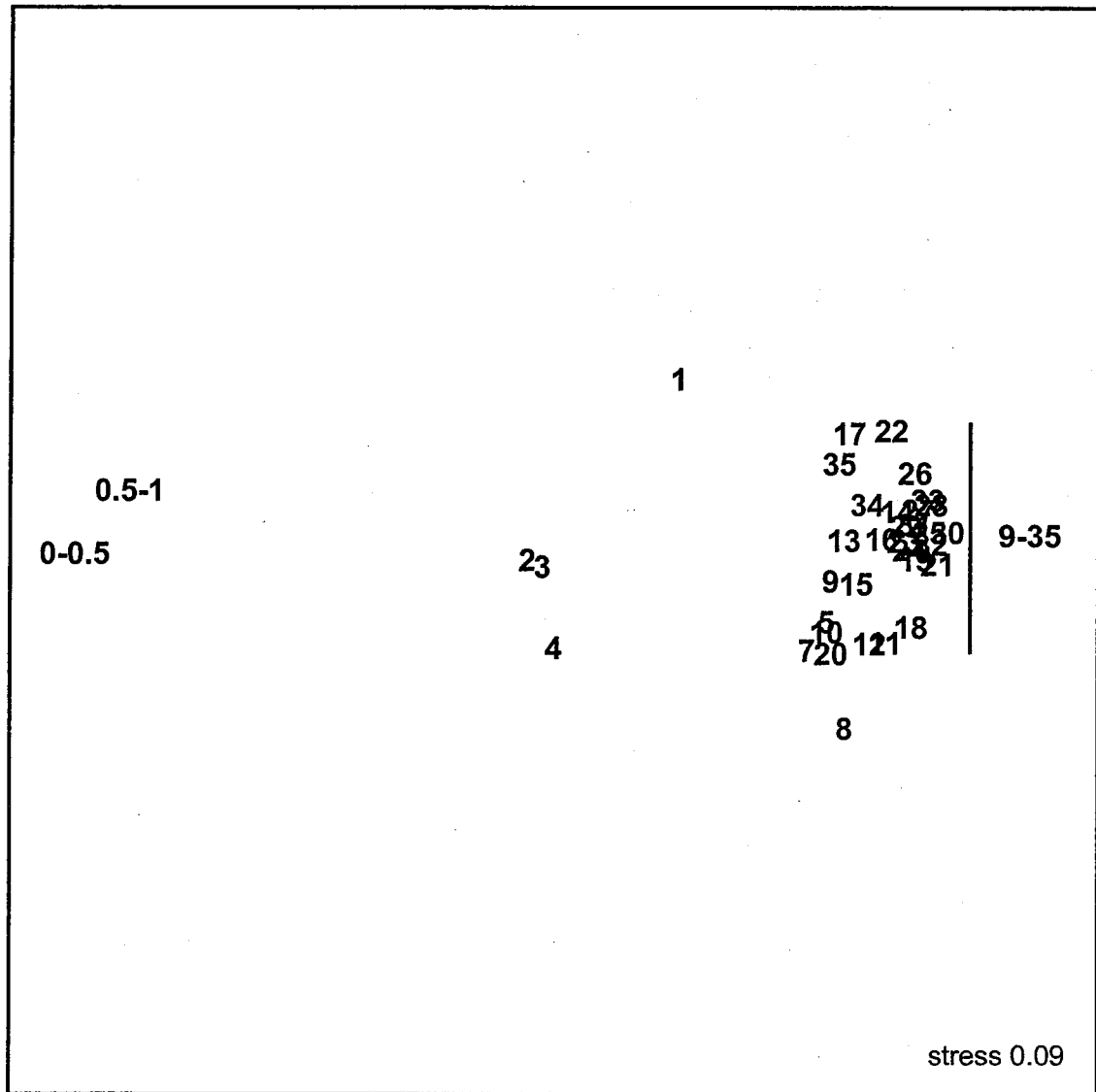


Figure 5.4.4: Ordination of estuarine zooplankton community composition across the salinity spectrum. Analysis based on the occurrence and mean density of the dominant mesozooplankton species in each of 36 salinity ranges (0 to 0.5, 0.5 to 1 ‰, and then increments of one from 1 to 35 ‰). The ordination technique is conceptually similar to a normal species x sites analysis, but here sites are replaced by salinity classes; thus, numbers in the ordination plot refer to salinity ranges and not site labels. Zooplankton was sampled monthly for 26 months at each of eight sites along the main axis of the Gamtoos Estuary, spanning the estuarine salinity gradient from near-marine conditions at the tidal inlet to the low-salinity regions near the REI.

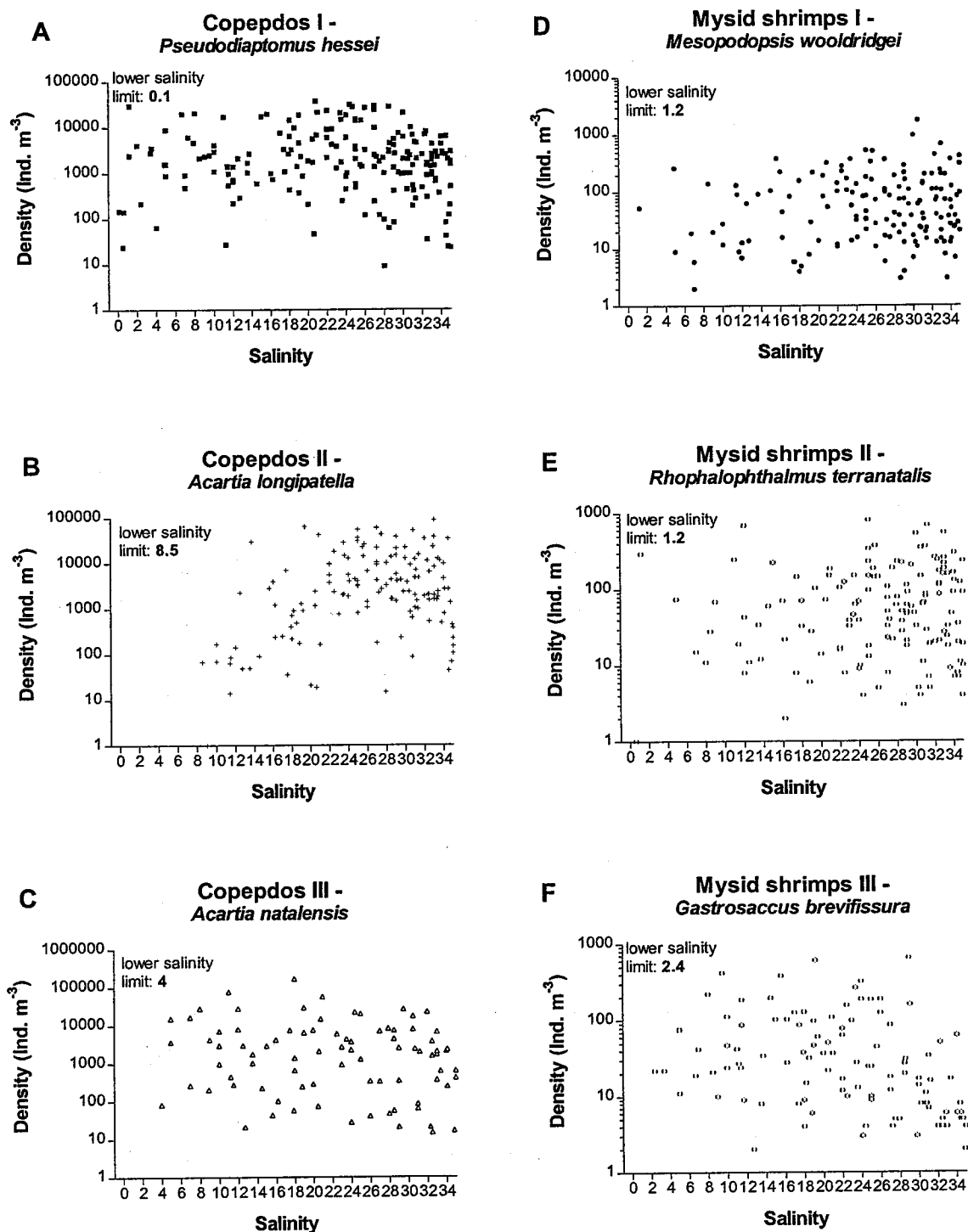


Figure 5.4.5: Distribution patterns of the dominant estuarine mesozooplankton species across the salinity spectrum. For each species, density and salinity data originate from 26 plankton surveys conducted monthly between February 1989 and March 1991 in the Gamtoos Estuary. During each survey, eight sites that were situated from near-marine to low-salinity waters along the main axis of the estuarine channel were sampled.

Patterns for marine elasmobranchs are less clear, but the small number of species that do enter estuaries, and for which salinity ranges are available ($n = 6$), precludes the extraction of general salinity zones (Figure 5.4.9A). Two species of marine elasmobranchs do penetrate into freshwater and the lower salinity range limits of three species cluster in the range of 17.0 to 20.0 ‰ (Figure 5.4.10C). All freshwater teleost species for which salinity ranges have been published are capable of occupying salinities of 10.0 ‰, and three out of eight species are recorded from seawater (Figure 5.4.10D). As is the case for elasmobranchs, the small number of freshwater teleosts occurring in estuaries makes the demarcation of salinity zones difficult. Also, the absence of most freshwater teleosts from even the upper reaches of estuarine systems (Whitfield 1998) provides little ecological relevance in trying to establish estuarine zones for this group.

5.4.4.2 Carbon pathways to estuarine fish

Carbon isotope values ($\delta^{13}\text{C}$ ratios) of fish from the Gamtoos separated into two distinct groups, namely species with ^{13}C depleted values close to -24.0 ‰, and species with markedly more ^{13}C enriched ratios, ranging from -18.4 to -21.9 ‰ (Figure 5.4.11). The first group comprised both benthic-feeding species such as gobiids, Cape stumpnose (*Rhabdosargus holubi*), spotted grunter (*Pomadasys commersonnii*) and white steenbras (*Lithognathus lithognathus*), as well as the zooplankton predator *Gilchristella aestuaria*. Carbon ratios in both these consumer groups overlapped broadly with their reported macrobenthic food sources (mean $\delta^{13}\text{C} = -26.9$ ‰) and zooplankton (mean $\delta^{13}\text{C} = -25.3$ ‰). However, in the case of the benthivores there are indications of a broader feeding spectrum that possibly also includes benthic detritus, macrophyte and sedimentary carbon sources (Figure 5.4.11).

The second group includes fish species that have traditionally been grouped into two strikingly different feeding categories, namely pelagic predatory species such as *Argyrosomus japonicus*, *Elops machnata* and *Lichia amia*, and benthic substratum-feeding mugilids. Both groups have broadly similar carbon isotope ratios (range of $\delta^{13}\text{C}$ values for predatory species was -21.9 to -19.3 ‰ compared with a mean $\delta^{13}\text{C}$ of -18.4 ‰ in mullets; Figure 5.4.11). This apparent dichotomy between vastly different feeding strategies and matching carbon isotope ratios of the consumer's tissues can be resolved by tracing the ultimate carbon source of each species group. The highly enriched $\delta^{13}\text{C}$ tissue values of mugilids indicates that benthic microalgae are the dominant food source. Similarly, tissue $\delta^{13}\text{C}$ values of the top predators in the system are closer to those of benthic microalgae than any other primary producer (Figure 5.4.11). This suggests that the ultimate carbon source of predatory fish is also benthic microalgae. Since none of these species feeds directly on the bottom sediments, but all are pelagic predators with a diet dominated by small fish, the transfer of carbon fixed by benthic microalgae to the top predators appears to be channelled via the mugilids.

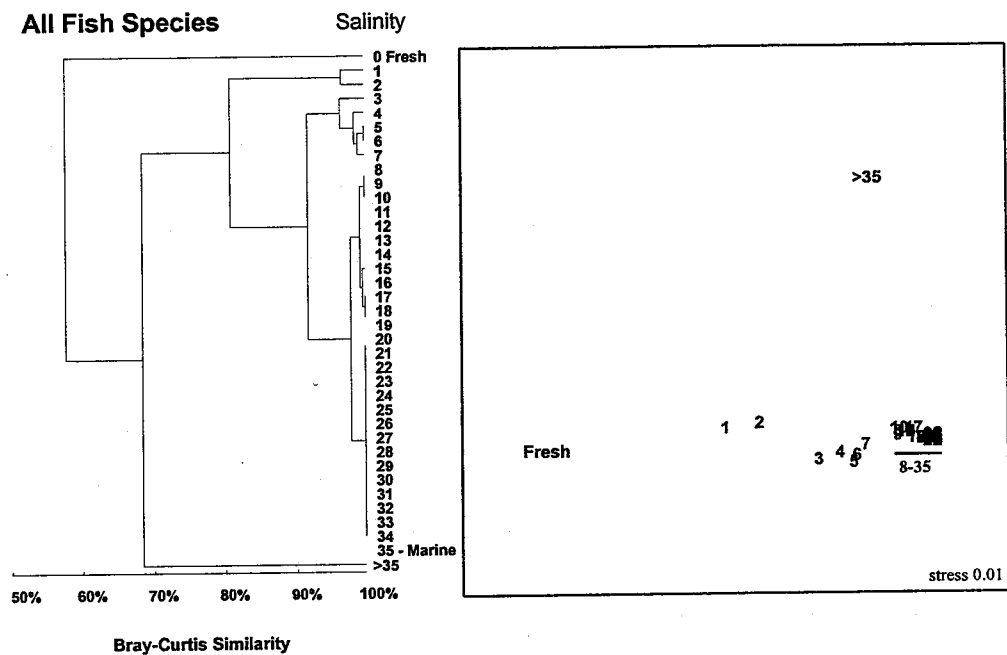


Figure 5.4.6: Classification and ordination of salinity ranges over which fish occur in southern African estuaries. Both analyses are complementary and indicate groups of salinity values for which species' records are similar. Composite analysis including records of salinity ranges for marine elasmobranchs, and freshwater, marine and estuarine teleosts (*cf.* Figures 5.4.8 and 5.4.9).

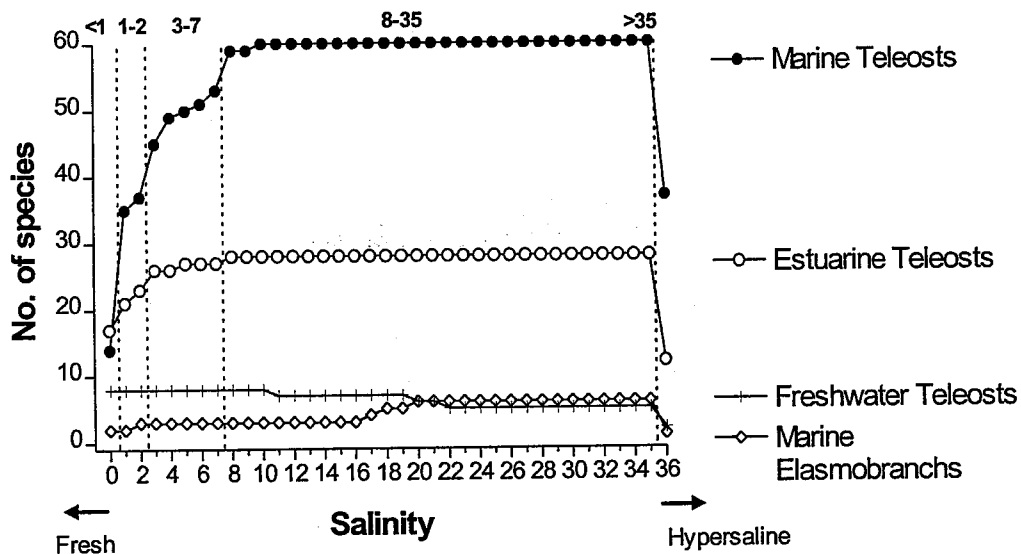


Figure 5.4.7: Distribution of fish species richness along the estuarine salinity gradient, based on published salinity ranges for Southern Africa (*cf.* Figure 7, Whitfield 1998). Dotted lines indicate the boundaries of salinity zones that were identified from concordance of species' ranges.

Table 5.4.2: Species distribution of estuarine fish in different salinity regimes of Southern African estuaries. Definition of salinity zones derived from multi-species concordance of salinity ranges (cf. Figure 5.4.6) of all fish species for which distributional records are available in Whitfield (1998). Grouping of estuarine-associated species according to life history styles *sensu* Whitfield (1994), with marine elasmobranchs grouped separately due to their different physiological mode of osmoregulation compared with teleosts.

	Salinity zone (‰)				
	< 1, fresh	1 - 2	3 - 7	8 - 35	> 35, hypersaline
Marine teleosts (n=60)					
No. of species	14	35 - 37	45 - 53	59 - 60	37
% of species	23	58 - 62	75 - 88	98 - 100	62
Estuarine teleosts (n=28)					
No. of species	17	21 - 23	26 - 27	28	12
% of species	61	75 - 82	93 - 96	100	43
Freshwater teleosts (n=8)					
No. of species	8	8	8	5 - 8	2
% of species	100	100	100	63 - 100	25
Marine elasmobranchs (n=6)					
No. of species	2	2 - 3	3	3 - 6	1
% of species	33	33 - 50	50	50 - 100	17
All species (n=102)					
No. of species	41	66 - 71	82 - 91	95 - 102	52
% of species	40	65 - 70	80 - 89	93 - 100	51

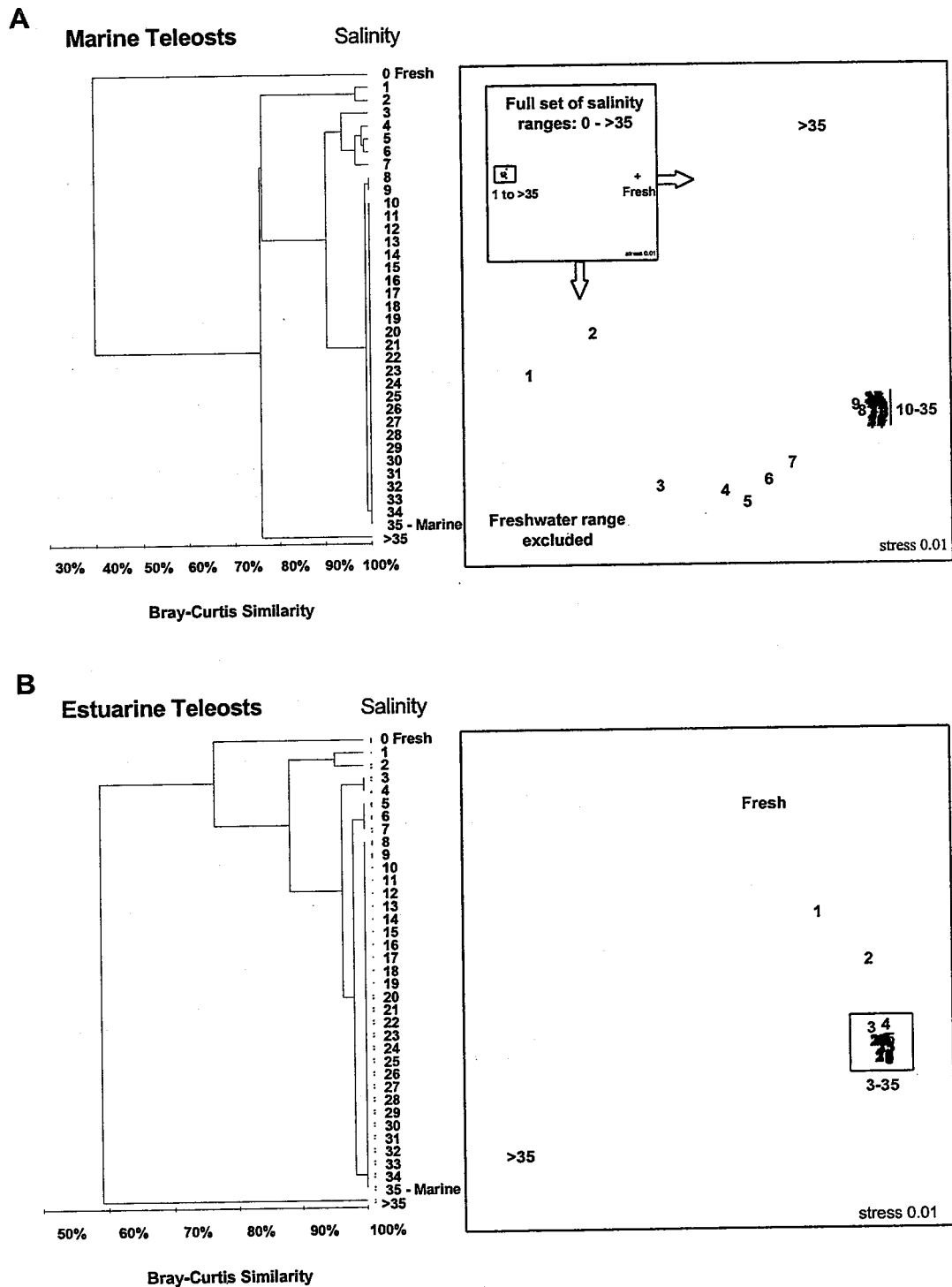


Figure 5.4.8: Classification and ordination of salinity ranges over which marine (A) and estuarine (B) teleosts occur in southern African estuaries. Both analyses are complementary and indicate classes of salinity regimes which have similar species records. The grouping of estuarine-associated fish species into marine and estuarine categories is based on life-history styles (Whitfield 1994) and salinity ranges extracted from Whitfield (1998).

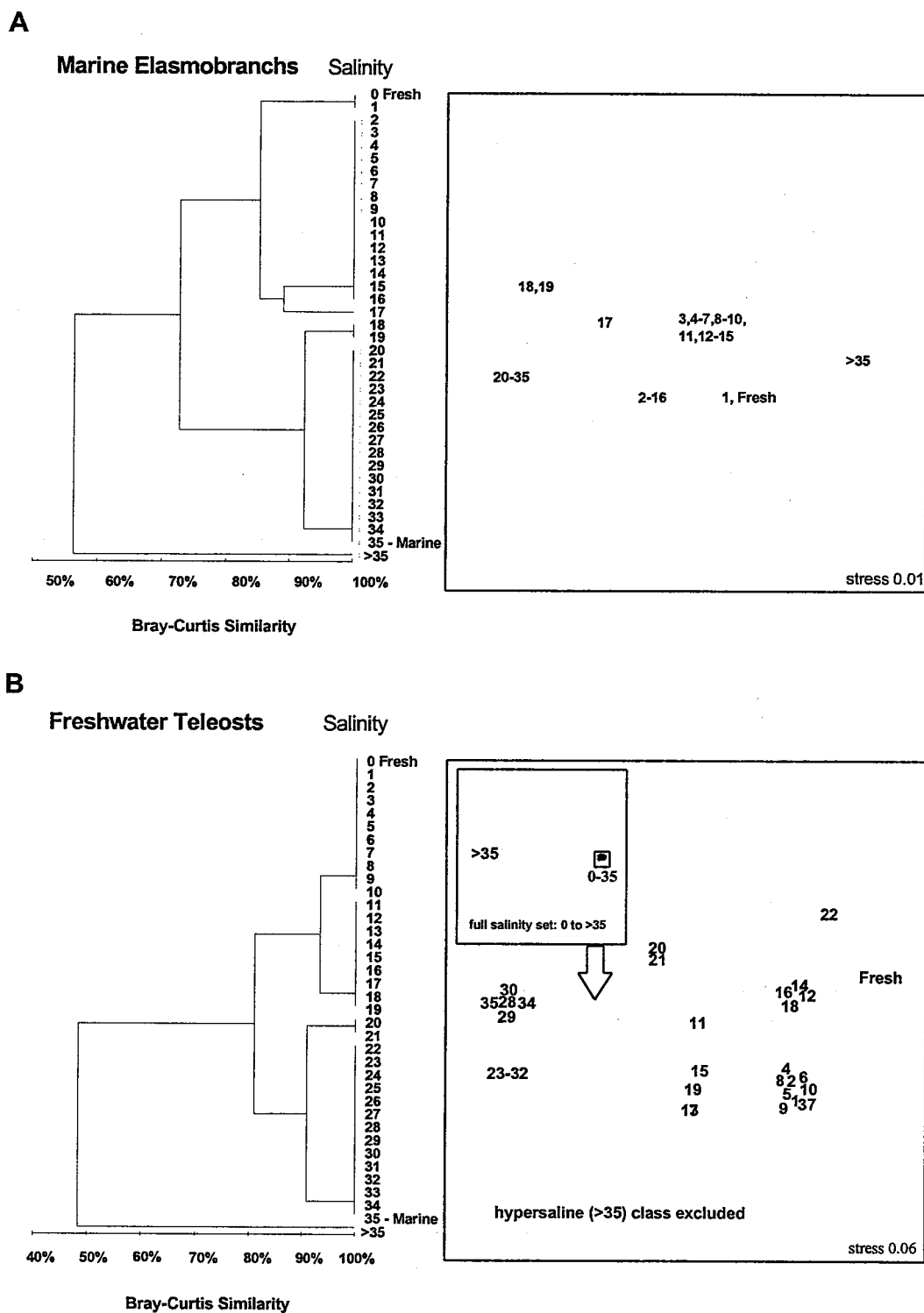


Figure 5.4.9: Classification and ordination of salinity ranges over which marine elasmobranchs (A) and freshwater teleosts (B) occur in southern African estuaries. Both analyses are complementary and indicate classes of salinity regimes which have similar species records. Grouping of fish is based on Whitfield (1994) and salinity ranges are extracted from Whitfield (1998).

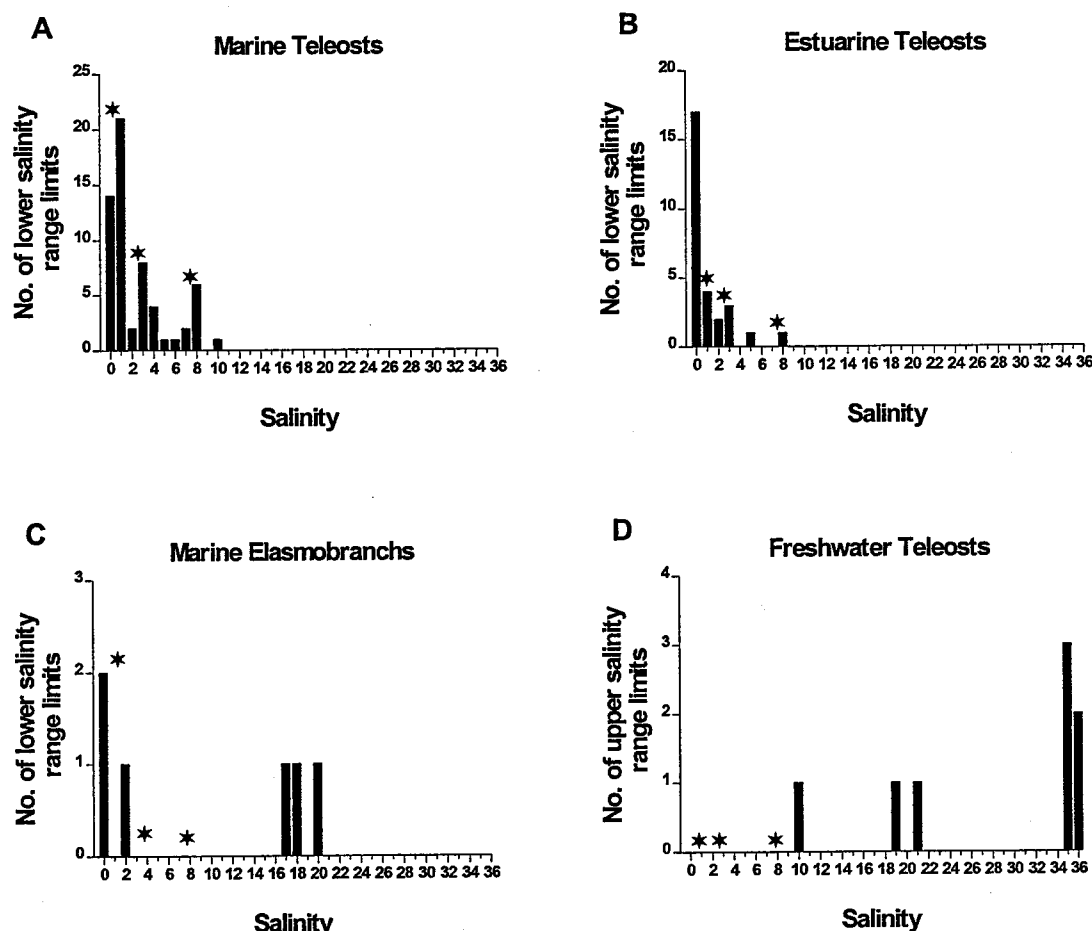


Figure 5.4.10: Distribution of lower limits of recorded salinity ranges for (a) marine teleosts, (b) estuarine teleosts, and (c) marine elasmobranchs which occur in southern African estuaries, as well as upper limits of salinity ranges for freshwater teleosts penetrating into estuaries (d). Symbols indicate position of zonal boundaries identified from similarity in distributional records for the estuarine fish fauna (cf. Figure 5.4.6).

5.4.5 Discussion

5.4.5.1 Biologically defined salinity zones

Salinity zones derived from the fish salinity ranges represent largely the lower boundaries of marine and estuarine teleost fish penetration of estuaries (Figure 5.4.10). Since few freshwater fish species extend into the upper sections of southern African estuaries (Whitfield 1994), the upper salinity limits of freshwater taxa contribute little to the overall fish structure of salinity zones. Similarly, freshwater plankton forms occur only sporadically in upper estuarine regions and, as was the case for fish, salinity zones extracted from zooplankton distributions represent largely the lower salinity limits of

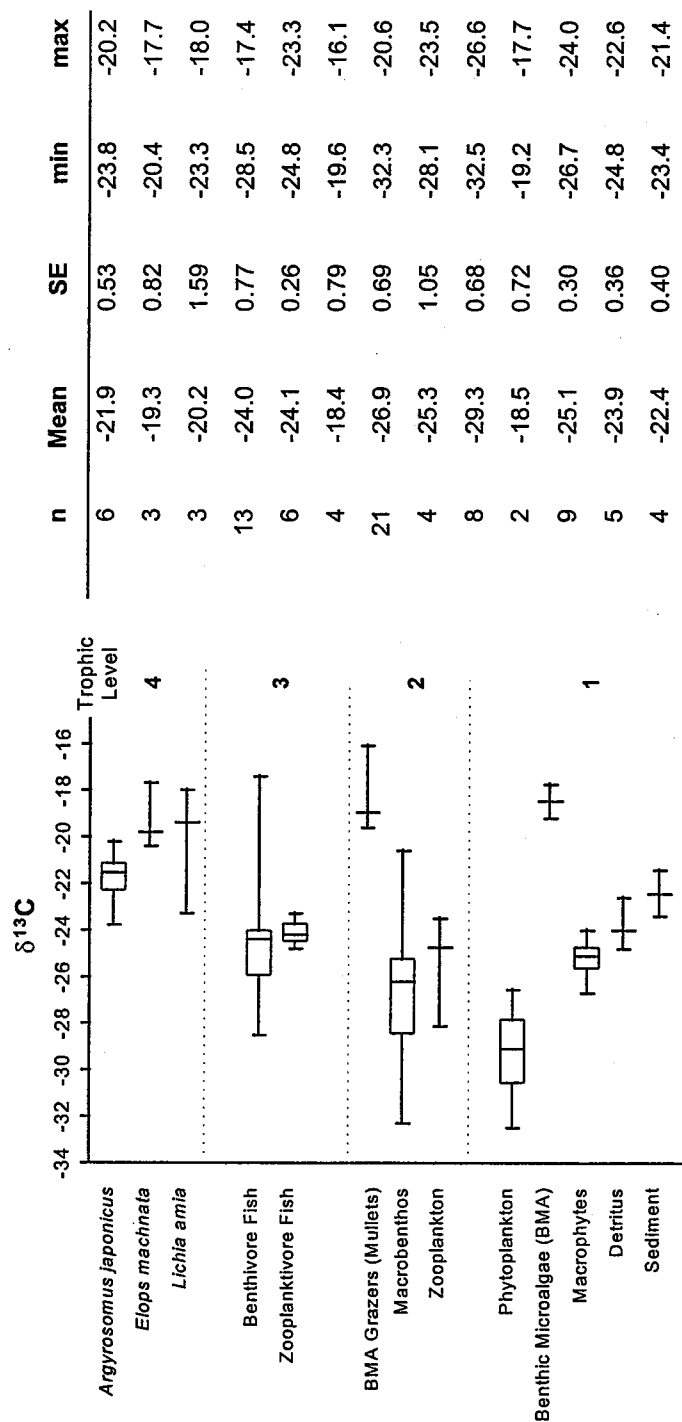


Figure 5.4.11: Comparison of carbon signals in primary producers, detrital deposits, and consumers to trace carbon flows in higher trophic levels. Plotted data encompass the upper estuarine regions of the Gamtoos estuary. Because carbon isotope ratios in lower trophic levels are dependent on habitat salinity, only carbon data from salinities which matched the range of salinities in which the top predator species were caught (10.0 to 17.0 ‰) are included in the comparison.

the component species (Figures 5.4.4 and 5.4.5). By contrast, macrobenthos in the tidal freshwater sections includes taxa of freshwater affinities such as oligochaetes and chironomid larvae which are excluded from the species spectrum in meso- to polyhaline waters, and extremely euryhaline estuarine species such as corophioid amphipods (Schlacher and Wooldridge 1996a).

The sharpest and most consistent salinity boundary derived from biota distribution data is the delineation of the tidal freshwater and oligohaline waters at approximately 1.0 ‰ (Figure 5.4.12). While the lower limit of saline waters is traditionally set at a salinity of 0.5 ‰ by the Venice system, a more recent analysis of salinity zones based on the distributional ranges of species in North American estuaries, identified an upper limit of 4.0 ‰ for the lowest salinity zone (Bulger *et al.* 1993). An upper bound of 4.0 ‰ for the most landward portion of the estuary does not apply to our data sets. By contrast, the boundary between tidal freshwater and oligohaline communities identified in all data sets is close to that defined by the Venice system. Because records of salinity ranges of estuarine fish analysed in this paper do not include decimal steps in the range 0.0 to 1.0 ‰, the precise endpoint of salinity tolerances around 0.5 ‰ cannot be established. Similarly, salinity values during field surveys of macrobenthic assemblages had no incremental steps from 0.0 to 1.0 ‰ between adjoining sites. Some evidence is available from zooplankton distributions, that changes in the species composition occurs in waters where the salinity is > 1.0 ‰ (Figure 5.4.4).

A physiological barrier separating brackish water and freshwater species occurs in a number of estuarine species in the salinity region 0.5 to 1.0 ‰ (Deaton 1981; Deaton and Greenberg 1986). Salinities between 0.0 and 1.0 ‰ are also the range of the most rapid changes in hydrochemistry across the salinity gradient, with most of the major biogeochemical transformations occurring in the initial mixing below 1.0 ‰ (Fauzi and Mantoura 1987). While a salinity limit of 1.0 ‰ appears to best reconcile our data with physiological and hydrochemical processes, the conventional delineation between fresh and saline water (0.5 ‰) appears a satisfactory boundary value to separate the tidal freshwater from the oligohaline estuarine section (McLusky 1993).

While there is considerable commonality in patterns of salinity distribution at the REI among all taxa and scales, assemblages of oligohaline (0.5 to 5.0 ‰) and polyhaline (5.0 to 18.0 ‰) waters show considerably more variable salinity distribution patterns (Figure 5.4.12). A value of 2.0 ‰ appears to be a significant biological/physiological barrier for a substantial number of estuarine and marine teleost fish, resulting in a marked drop in the number of species that have been recorded in the 1.0 to 2.0 ‰ range (Figure 5.4.7). This salinity boundary is clearly defined and consistent for southern African estuarine fish (Figure 5.4.6), but has no direct equivalent in either the Venice system or the classification scheme of Bulger *et al.* (1993) (Figure 5.4.12). This zone is also broadly mirrored by zooplankton (Figures 5.4.4 and 5.4.11), having the upper boundary extended to 4.0 ‰. Macrobenthic field data are ambiguous about the existence of a zone that occupies the lower end of the oligohaline range (Figures 5.4.3 and 5.4.11). There is no apparent physiological reason for the existence of a boundary at 2.0 ‰, as most marine teleost fishes are isosmotic with saline water of 11.0 to 14.0 ‰, and freshwater fishes are hyperosmotic up to 15.0 ‰ (Whitfield and Blaber 1976; Schmidt-Nielsen 1983).

Conventionally, mesohaline waters encompass the wide range of 5.0 to 18.0 ‰ (Venice System), a class also broadly supported by the 'inner' estuarine zone of 2.0 to 14.0 ‰ identified by Bulger *et al.* (1993). Salinity distributions of two phylogenetic groups examined here (fish and macrobenthos) do, however, not concur with this range. Fish salinity distributions show a distinct salinity zone extending from 3.0 to 7.0 ‰, and macrobenthos assemblages in the range 4.0 to 9.0 ‰ are structurally distinct from more saline waters (Figure 5.4.12). These findings support a further boundary that subdivides the mesohaline range between 7.0 and 9.0 ‰ (Figure 5.4.12).

5.4.5.2 Evaluation of boundaries of low-salinity regions

In this section, the upper boundary of the low-salinity regions of estuaries was set at a vertically averaged salinity of 10.0 ‰. This value constitutes primarily an operational working definition based on the personal experience of participating team members. No objective criteria are, however, available to judge the exact derivation of this boundary value.

By contrast, the present analysis clearly demonstrates that:

1. The underlying structure of salinity distributions of the three major faunal components (fish, zooplankton and macrobenthos) is heterogeneous at salinities < 10.0 ‰.
2. Estuarine regions of salinities < 10.0 ‰ are not characterised by a homogenous set of biological structures and thus do not represent a single zone of the estuary.
3. Three biologically-defined subdivisions exist in the low-salinity regions which broadly span the freshwater to lower mesohaline salinity classes.
4. Boundaries delineating the subdivisions that constitute the low salinity regions occur at salinities of 0.5 to 1.0, 2.0 to 4.0 and 7.0 to 9.0 ‰, resulting in three salinity zones for the upper sections of the estuary:

Zone 1: 0.0 to 0.5 ‰ (1): tidal freshwater

Zone 2: 1.0 to 2.0 ‰ (4): lower oligohaline

Zone 3: 3.0 to 4.0 ‰ to 7.0 to 9.0 ‰: lower mesohaline

5.4.5.3 Food web structures at higher trophic levels

Although benthic detritus has frequently been described as an important component in the diet of mugilids in southern African estuaries (Whitfield 1998), our data do not support any significant assimilation of benthic detritus by either of the mullet species analysed. In the upper sections of the Gamtoos Estuary, mugilids may consume benthic detritus but these species are assimilating mainly the benthic microalgal component of their diet (Figure 5.4.11).

Carbon isotope ratios ($\delta^{13}\text{C}$) of all potential food sources (except benthic microalgae and mugilids) are too depleted to make a significant contribution to the carbon assimilated by the top predators in the system. Carbon in piscivorous, pelagic predators is thus ultimately derived from benthic microalgal production. The transfer of microalgal carbon to higher trophic levels is channelled via two steps, namely grazing of benthic microalgae by mugilids, and predation on these species by piscivorous fish. Conversely, both zoobenthivorous and zooplanktivorous fish appear to make a minor

contribution to the top predators' energy intake. These results stress the energetic importance of benthic microalgae in supporting top fish predators in the Gamtoos estuarine environment.

In addition to the clear carbon pathways leading from microphytobenthos to fish identified above, a second pathway that originates from phytoplankton is important for higher trophic levels. Several species of benthic feeding fish are abundant in the upper, low-salinity sections of the estuary. All members of this group appear to derive a substantial part of their assimilated carbon from benthic invertebrates. Benthic invertebrates in turn assimilate mostly phytoplankton and not macrophyte carbon (Schlacher and Wooldridge 1996c). Thus, phytoplankton primary production ultimately fuels populations of benthic fish in the system. Similarly, zooplankton grazing on phytoplankton is the intermediate trophic step that links pelagic primary production with the abundant pelagic zooplanktivore, *Gilchristella aestuaria* (Figure 5.4.11). As was the case for benthic feeding fish, phytoplankton is the ultimate carbon source for this species.

The structure of the food web in the upper Gamtoos Estuary argues forcefully for the pivotal role of microalgae in the energetics of this estuarine compartment. Since freshwater inflow promotes productivity of both benthic and pelagic microalgae in estuaries (Schlacher and Wooldridge 1996d; Adams and Bate 1999), supply rates of freshwater to estuaries are clearly a dominant structuring force for estuarine communities at several trophic levels.

5.5 Acknowledgements

The University of the Sunshine Coast (Australia) is thanked for granting sabbatical leave to Dr T.A. Schlacher for this project. Monika Schlacher-Hoenlinger helped with field sampling and laboratory processing of invertebrate samples. Dr Angus Paterson's contributions and advice with the 'fishy' aspects of the work was much appreciated.

Section 5.1: The Austrian Ministry of Science and the University of Vienna provided additional funding for this component of the programme. Field assistance was provided by Monika Schlacher-Hoenlinger, Brent Newman and Ursula Scharler.

Section 5.3: We thank Pierre Richard (Ifremer, France) for advice in extracting the benthic microalgae from sediments. Assistance in the field and laboratory processing of invertebrate samples was provided by Monika Schlacher-Hoenlinger. Johan van der Molen identified the benthic algae.

5.6 References

- Adams, J.B. 1991. The distribution of estuarine macrophytes in relation to freshwater in a number of Eastern Cape estuaries. M.Sc. Thesis, University of Port Elizabeth, Port Elizabeth.
- Adams, J.B. and Bate, G. 1999. Primary producers - estuarine microalgae. In: Estuaries of South Africa (ed. B.R. Allanson and D. Baird). Cambridge University Press, Cambridge: pp. 91-100.
- Blaber, S.J.M., Kure, N.F., Jackson, S. and Cyrus, D.P. 1983. The benthos of South lake, St Lucia following a period of stable salinities. *South African Journal of Zoology* 18: 311-319.

- Bolt, R.E. 1969. The benthos of some southern African lakes. Part II: the epifauna and infauna of the benthos of the benthos of Lake Sibaya. *Transactions of the Royal Society of South Africa* 38: 249-269.
- Bolt, R.E. and Allanson, B.R. 1975. The benthos of some southern African lakes. Part 3: The benthic fauna of Lake Nhlange, Kwazulu, South Africa. *Transactions of the Royal Society of South Africa* 41: 241-261.
- Branch, G. and Branch, M. 1981. The living shores of southern Africa. Struik Publishers, Cape Town: 272 pp.
- Branch, G.M., Day, J.A. 1984. Ecology of southern African estuaries: Part XIII. The Palmiet river estuary in the south-western Cape. *South African Journal of Zoology* 19: 63-77.
- Branch, G.M., Griffiths, C.L., Branch, M.L. and Beckley, L.E. 1994. Two Oceans: a guide to the marine life of southern Africa. David Philip Publishers, Johannesburg: 360 pp.
- Bulger, A.J., Hayden, B.P., Monaco, M.E., Nelson, D.M. and McCormick-Ray, M.G. 1993. Biologically-based estuarine salinity zones derived from multivariate analysis. *Estuaries* 16: 311-322.
- Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* 18: 117-143.
- Clarke, K.R. and Green, R.H. 1988. Statistical design and analysis for a 'biological effects' study. *Marine Ecology Progress Series* 46: 213-226.
- Clarke, K.R., Warwick, R.M. and Brown, B.E. 1993. An index showing breakdown of seriation related to disturbance in a coral-reef assemblage. *Marine Ecology Progress Series* 102: 153-160.
- Deaton, L.E. 1981. Ion regulation in freshwater and brackish water bivalve mollusks. *Physiology Zoology* 54: 109-121.
- Deaton, L.E. and Greenberg, M.J. 1986. There is no Horohalinicum. *Estuaries* 9: 20-30.
- De Decker, H.P., Bally, R. 1985. The benthic macrofauna of the Bot River estuary, South Africa, with a note on its meiofauna. *Transactions of the Royal Society of South Africa* 45: 379-396.
- Deegan, L.A. and Garritt, R.H. 1997. Evidence for spatial variability in estuarine food webs. *Marine Ecology Progress Series* 147: 31-47.
- de Villiers, C.J. 1990. Aspects of the biology of the infaunal bivalve mollusc *Solen cylindraceus* in the Kariega estuary. Ph.D. Thesis, Rhodes University, Grahamstown: 194 pp.
- Field, J.G., Clarke, K.R. and Warwick, R.M. 1982. A practical strategy for analysing multispecies distribution patterns. *Marine Ecology Progress Series* 8: 37-52.
- France, R.L. and Peters, R.H. 1997. Ecosystem differences in the trophic enrichment of ^{13}C in aquatic food webs. *Canadian Journal of Fisheries and Aquatic Science* 54: 1255-1258.
- Fauchald, K. and Jumars, P.A. 1979. The diet of worms: a study of polychaete feeding guilds. *Oceanography and Marine Biology Annual Review* 17: 193-284.
- Fauzi, R. and Mantoura, C. 1987. Organic films at the halocline. *Nature* 328: 579-580.

- Hanekom, N., Baird, D. and Erasmus, T. 1988. A quantitative study to assess standing biomasses of macrobenthos in soft substrata of the Swartkops estuary, South Africa. *South African Journal of Marine Science* 6: 163-174.
- Heineken, T.J.E. 1981. Estuaries of the Cape. Part II: Synopses of available information on individual systems. CSIR Report No. 7: Gamtoos (CMS48). CSIR Research Report 406: 40 pp.
- Hill, M.O. 1973. Diversity and evenness: A unifying notation and its consequences. *Ecology* 54: 427-432.
- Jerling, H.L. and Wooldridge, T.H. 1995. Relatively negative $\delta^{13}\text{C}$ ratios of mesozooplankton in the Sundays River Estuary, comments on potential carbon sources. *South African Journal of Aquatic Science* 21: 71-77.
- Jezewski, W.A. and Roberts, C.P.R. 1986. Estuarine and lake freshwater requirements. Department of Water Affairs, Technical Report No TR129, Pretoria: 22 pp.
- Kalejta, B. and Hockey, P.A.R. 1991. Distribution, abundance and productivity of benthic invertebrates at the Berg River estuary, South Africa. *Estuarine, Coastal and Shelf Science* 33: 175-191.
- Koop, K., Bally, R. and McQuaid, C.D. 1983. The ecology of South African estuaries. Part 12: The Bot River, a closed estuary in the south-western Cape. *South African Journal of Zoology* 18: 1-10.
- Ludwig, J.A. and Reynolds, J.F. 1988. Statistical ecology. Wiley, New York: 337 pp.
- McLusky, D.S. 1981. The estuarine ecosystem. Blackie, Glasgow: 215 pp.
- McLusky, D.S. 1993. Marine and estuarine gradients - an overview. *Netherlands Journal Aquatic Ecology* 27: 489-493.
- Montagna, P.A. and Yoon, W.B. 1991. The effect of freshwater inflow on meiofaunal consumption of sediment bacteria and microphytobenthos in San Antonio Bay, Texas, U.S.A. *Estuarine, Coastal and Shelf Science* 33: 529-547.
- Morris, A.W., Mantoura, R.F.C., Bale, A.J. and Howland, R.J.M. 1978. Very low salinity regions of estuaries: important sites for chemical and biological reactions. *Nature* 274: 678-680.
- Paterson, A.W. and Whitfield, A.K. 1997. A stable carbon isotope study of the food web in a freshwater-deprived South African estuary, with particular emphasis on the ichthyofauna. *Estuarine, Coastal and Shelf Science* 45: 705-715.
- Reddering, J.S.V. and Esterhuysen, K. 1984. Sedimentation in the Gamtoos estuary. Department of Geology, University of Port Elizabeth, Rosie Report No 7: 85 pp.
- Reddering, J.S.V. and Scarr, N.G. 1990. The physical characteristics of the Gamtoos estuary. Institute for Coastal Research Report 23. University of Port Elizabeth: 39 pp.
- Riera, P., Richard, P., Grémare, G. and Blanchard, G. 1996. Food source of intertidal nematodes in the Bay of Marennes-Oléron (France), as determined by dual stable isotope analysis. *Marine Ecology Progress Series* 142: 303-309.
- Schlacher, T.A. and Wooldridge, T.H. 1995. Small-scale distribution and variability of demersal zooplankton in a shallow, temperate estuary: tidal and depth effects on species-specific heterogeneity. *Cahiers Biologie Marina* 36: 211-227.

- Schlacher, T.A. and Wooldridge, T.H. 1996a. Axial zonation patterns of subtidal macrozoobenthos in the Gamtoos estuary, South Africa. *Estuaries* 19(3): 680-696.
- Schlacher, T.A. and Wooldridge, T.H. 1996b. Patterns of selective predation by juvenile, benthivorous fish on estuarine macrofauna. *Marine Biology* 125: 241-247.
- Schlacher, T.A. and Wooldridge, T.H. 1996c. Origin and trophic importance of detritus – evidence from the stable isotopes in the benthos of a small, temperate estuary. *Oecologia* 106: 382-388.
- Schlacher, T.A. and Wooldridge, T.H. 1996d. Ecological responses to reductions in freshwater supply and quality to South Africa's estuaries: lessons for management and conservation. *Journal of Coastal Conservation* 2: 115-130.
- Schmidt-Nielsen, K. 1983. Animal physiology: adaptation and environment. Cambridge University Press, Cambridge: 619 pp.
- Schuchardt, B. and Schirmer, M. 1991. Phytoplankton maxima in the tidal freshwater reaches of two coastal plain estuaries. *Estuarine, Coastal and Shelf Science* 32: 187-206.
- Schumann, E.H. and Pearce, M.W. 1997. Freshwater inflow and estuarine variability in the Gamtoos estuary: Implications for watershed management. *Estuaries* 20(1): 124-133.
- Ter Morshuizen, L.D., Whitfield, A.K. and Paterson A.W. 1997. Distribution patterns of fishes in an Eastern Cape estuary and river with particular emphasis on the ebb and flow region. *Transactions of the Royal Society South Africa* 51: 257-280.
- Whitfield A.K. 1994. A review of the ichthyofaunal biodiversity in southern African estuarine systems. *Annales Sciences Zoologiques Musee Royal de l'Afrique Centrale* 275: 149-163.
- Whitfield, A.K. 1998. Biology and ecology of fishes in southern African estuaries. *Ichthyological Monographs of the J.L.B. Smith Institute of Ichthyology* No. 2: 223pp.
- Whitfield, A.K. and Blaber, S.J.M. 1976. The effects of temperature and salinity on *Tilapia rendalli* Boulenger 1986. *Journal of Fish Biology* 9: 99-104.

**Chapter 6: An investigation into the structure and functioning of the ichthyofauna
associated with the REI zone in two Eastern Cape estuaries
with differing freshwater inputs**

A.K. Whitfield, A.W. Paterson and L.D. Ter Morshuizen
J.L.B. Smith Institute of Ichthyology, Grahamstown

General summary

1. Fish assemblages in the REI region of the Great Fish and Kariega estuaries are described and compared to the catch composition in other reaches of each estuary.
2. The available evidence suggests that river flow has a major impact on the structure and functioning of fish communities within these two estuaries and the upper reaches/REI region in particular.
3. The high conductivity of riverine flow into the Great Fish Estuary promoted the utilisation of both the river and headwaters of the estuary by euryhaline marine and estuarine fish species.
4. Those marine species which are most dependant on South African estuaries as nursery areas tended to move furthest up the Great Fish system, often into the river itself.
5. Of particular significance were the large numbers of 0+ juveniles of the recreationally important spotted grunter *Pomadasys commersonnii* and dusky kob *Argyrosomus japonicus* that utilise the REI region of the Great Fish system as a primary nursery area.
6. During and immediately after flood conditions, salts in the Great Fish River catchment are diluted, freshwater conductivity decreases and the euryhaline marine species are no longer abundant in riverine samples. There is therefore an indirect link between river flow and marine fish distribution and abundance in the Great Fish system.
7. The geographical headwaters of the freshwater deprived Kariega Estuary were utilised by a range of fish species but, in contrast to the Great Fish Estuary, few important recreational or subsistence species (e.g. *P. commersonnii* and *A. japonicus*) were recorded.
8. The reduced riverine flow into the Kariega Estuary resulted in an extremely restricted REI area being made available to resident and migrant fish species.
9. Although salinity has been shown to be an important structuring force influencing fish assemblages in many South African estuaries, it is not the primary factor governing the distribution of fish in a freshwater deprived, marine dominated system such as the Kariega.

General introduction

Ichthyofaunal research in southern African estuaries has concentrated on the lower and middle reaches of these systems, thus resulting in a paucity of information on fish communities associated with the upper reaches (including the REI region). This lack of research is disturbing, as the abiotic and biotic processes occurring in the upper reaches of estuaries is likely to have an impact on the system lower down. The few South African studies that have examined the longitudinal distribution of fish have shown that the greatest species diversity occurs in the lower reaches where certain marine species usually have a higher representation than further up the system (Marais 1983a, Ter Morshuizen and Whitfield 1994).

The distribution of fish in South African estuaries has been related to a number of factors (e.g. salinity, turbidity and food distribution) but the interaction between abiotic and biotic components is still poorly understood. Traditionally, salinity has been viewed as one of the most important variables affecting the distribution of organisms in estuaries. Blaber (1978) stated that southern African estuaries are usually characterised by a longitudinal grading of fish species, with certain taxa being confined to areas of higher salinity near the mouth, whereas others are found predominantly in the less saline upper reaches. Other factors such as temperature, turbidity and dissolved oxygen may also affect fish distribution (Blaber and Blaber 1980). Research on the Humber Estuary in the United Kingdom established that salinity was the dominant factor affecting the distribution of species, with temperature also having a major influence (Marshall and Elliott 1998).

What are the principal factors influencing the distribution of fish species in South African estuaries, and the REI in particular? In order to answer these questions it was necessary to conduct studies in at least two of the four estuaries originally selected for the Eastern Cape REI project. Since work had already commenced in the headwaters of the freshwater 'rich' Great Fish Estuary, it was decided to continue focusing on this system and to broaden the study to include the nearby freshwater 'deprived' Kariega Estuary.

Using the global hypothesis as a basis (see Chapter 1), the prime objective of this study was to test the following hypothesis in the context of fish assemblages in Eastern Cape estuaries:

"Ichthyofaunal structure and function in the river-estuary interface region is governed primarily by the quantity, quality and supply pattern of freshwater received. Furthermore, riverine input has a major influence on fish assemblages in the entire estuary".

In order to fully explore the issues raised in the above hypothesis it was necessary to extend the research area beyond the REI region to include the middle and lower reaches of both the Great Fish and Kariega systems. In addition, comparative ichthyofaunal information from other Eastern Cape estuaries is used to highlight the new interpretations that arose from this study concerning the utilization of the headwater region.

6.1 Distribution patterns of fishes in a freshwater rich estuary, with particular emphasis on the REI region

6.1.1 Summary

Two and a half years of data were collected from the Great Fish River (< 1.0 ‰), head region (1.0 to 4.0 ‰) and remaining estuary (> 4.0 ‰) using seine and gill nets to determine the fish species composition within these regions. Euryhaline marine taxa belonging to the families Mugilidae and Sparidae dominated the catch in all three regions. *Gilchristella aestuaria* was the single most common estuarine resident, and several marine species were recorded in freshwater for the first time. The cichlid *Oreochromis mossambicus* was the most common freshwater species but was only abundant in riverine samples. Catches made with the gill nets and large seine net both indicated that the densities of fish were higher at the head than in either the riverine or estuarine regions. Since the highest levels of suspended particulate organic matter in the Great Fish Estuary are known to be in the head region, the highest densities of fishes would be expected from the same area. Physical parameters were also measured to ascertain some of the factors which may affect distribution and length frequencies of the most common taxa within this system. Salinity was shown to be an important factor affecting the distribution of species within the sampled area, with domination by euryhaline marine taxa being most pronounced in the estuary and decreasing towards the river region.

6.1.2 Introduction

Although there is a large amount of information available on fishes in South African estuaries (Wallace *et al.* 1984; Potter *et al.* 1990; Whitfield 1990), there is a lack of information on fish assemblages at the river-estuary interface (REI) or ebb and flow region. This is in spite of the warning by Rogers *et al.* (1984) that estuarine researchers need to give attention to the freshwater areas adjacent to estuaries due to their importance in the overall functioning of the estuary and their acute vulnerability to human activities. Although Bok (1983) recorded 11 fish species (six freshwater, two estuarine and three marine) in the ebb and flow region of the Kowie system, and Rayner (1993) found 15 species (four freshwater, two estuarine and nine marine) at the Keiskamma Estuary ebb and flow, no previous studies have been directed specifically at the REI region. It was therefore decided to undertake investigations of the ichthyofaunal assemblages in the head region of two Eastern Cape estuaries, one of which was the Great Fish system.

6.1.3 Study area

Arising in the Karoo, the 650 km long Great Fish River is a sixth order system that drains a catchment of more than 30 000 km² (O'Keeffe 1989). Prior to 1975, the river was known to have had a highly variable flow regime, with periods of zero flow frequently occurring (Reddering and Esterhuysen

1982). However, in 1975 the erratic flow of this system was stabilised by the transfer of water from the Orange River. For the period 1978 to 1991 the mean monthly flow rates at the Department of Water Affairs and Forestry Station Q9H018-A01 exceeded $23 \times 10^6 \text{ m}^3$, but the mean annual discharge into the estuary decreased, when compared to pre-1975 figures, mainly due to the abstraction of water for irrigation (O'Keeffe and de Moor 1988).

The Great Fish River has a low to average conservation status compared to its original form (O'Keeffe 1989). Watling and Watling (1983) regarded this river as being unpolluted in terms of metal contaminants, with those being present originating from natural rather than anthropogenic sources (Gardner *et al.* 1985). Salt levels in the river are high ($> 2000 \text{ mg l}^{-1}$) but these are diluted by water from the Orange River transfer scheme (132 mg l^{-1}) (O'Keeffe and de Moor 1988). In addition, the Great Fish River is extremely turbid due to the highly erodible Beaufort soils in the catchment (Laurenson 1984). Whitfield *et al.* (1994) showed that turbidity was high throughout the estuary but decreased towards the mouth of the system. Estuarine sediment originates mainly from the catchment area, with marine sediments seldom penetrating beyond the first kilometre from the mouth (Reddering and Esterhuysen, 1982). Due to persistent freshwater input, the estuary has a strong longitudinal salinity gradient between the mouth and head reaches (Whitfield *et al.* 1994). A preliminary study on the salinity regime of the Great Fish Estuary during 1993 showed that a salt wedge is often present in the system from the mouth to about 11.0 km upstream (Ter Morshuizen 1994).

The Great Fish system was selected for this project because it receives a regular freshwater input and is navigable by boat beyond the estuarine section. This facilitated comparable sampling in both the river and estuary regions. Bi-monthly sampling on the new-moon low tide was undertaken from November 1992 to January 1995, providing samples that spanned three summers and two winters. The sampling regime was divided into three components so that a wide size range of fish could be collected.

6.1.4 Materials and methods

6.1.4.1 Fish sampling

Small seine net

A seine net of 5 x 1 m and 0.5 mm bar mesh size was used from September 1994 to January 1995 at sites 03 to 14 (Figure 6.1.1). The small seine net, which targeted small, littoral fishes, was operated by two people in water less than 75 cm deep and covered an estimated 25 m² per haul.

Large seine net

On all 14 field trips, sampling was conducted over a two day period using a 30 x 2 m seine net with a 10 mm bar mesh size, fitted with a bag of 5 mm bar mesh size. From November 1992 until July 1994 samples were collected from sites 1 to 8. For the last three trips, September 1994 to January 1995, sampling occurred at all 17 sites (Figure 6.1.1). Sites 03 to 8 were sampled on the

to 14 on the subsequent day. The net swept an estimated 600 m² at each site, and was operated in water no deeper than 1.5 m. This net was used to sample the pelagic, demersal and benthic species in the littoral region over a predominantly mud bottom. The large size of the net facilitated the capture of small and medium size classes of most species.

Gill net

From January 1994 to January 1995 gill nets were used to sample the larger fish component in the channel. The gill nets were set overnight over a two day period in the same area that had been seined on the respective day, i.e. between sites 03 to 8 on the first night and 9 to 14 on the subsequent night (Figure 6.1.1). The nets were laid by boat at 18h00, checked at midnight and lifted at 06h00 the following morning. A total of six multifilament nets were used, each 20 m x 2 m with a 50 mm bar mesh size. These nets were set parallel to the shore and in pairs to compensate for possible microhabitat differences.

With the exception of the sandy bottom at site 14 (Figure 6.1.1), the three techniques were used exclusively over predominantly muddy substrata where rocks and submerged macrophytes were absent. Consequently, the catches obtained with the two seine nets are conservative, since the unconsolidated mud bottom made sampling very difficult. All samples were fixed on site in 10% formalin. In the laboratory, fish were identified to the lowest possible taxonomic level, measured to the nearest mm standard length (SL) and preserved in 60% isopropyl alcohol.

The grouping of species according to their life-history categories follows the system described by Whitfield (1994). Mugilids < 30.0 mm SL were not identified beyond family level. Only confirmed species were used in the life-history category analyses.

6.1.4.2 Physico-chemical sampling

A water sample was collected from midwater at each seine net site and adjacent to each gill net fleet whenever the nets were laid, checked or lifted. Temperature was measured on site with a mercury thermometer, and salinity and turbidity were determined in the laboratory using a Reichert Instruments Model 10419 optical salinometer and Hach Model 2100A turbidimeter respectively.

6.1.4.3 Statistical methods

Temperature, turbidity and salinity were compared between sites, and between sites and the adjacent channel, using a one-way analysis of variance (ANOVA) test. The ANOVA used was the Scheffe method, which gives considerable certainty to the level of confidence obtained. Species richness (R) (Margalef 1958), species diversity (H) and diversity evenness (E) (Krebs 1978) were calculated for the species assemblages in each of the three regions.

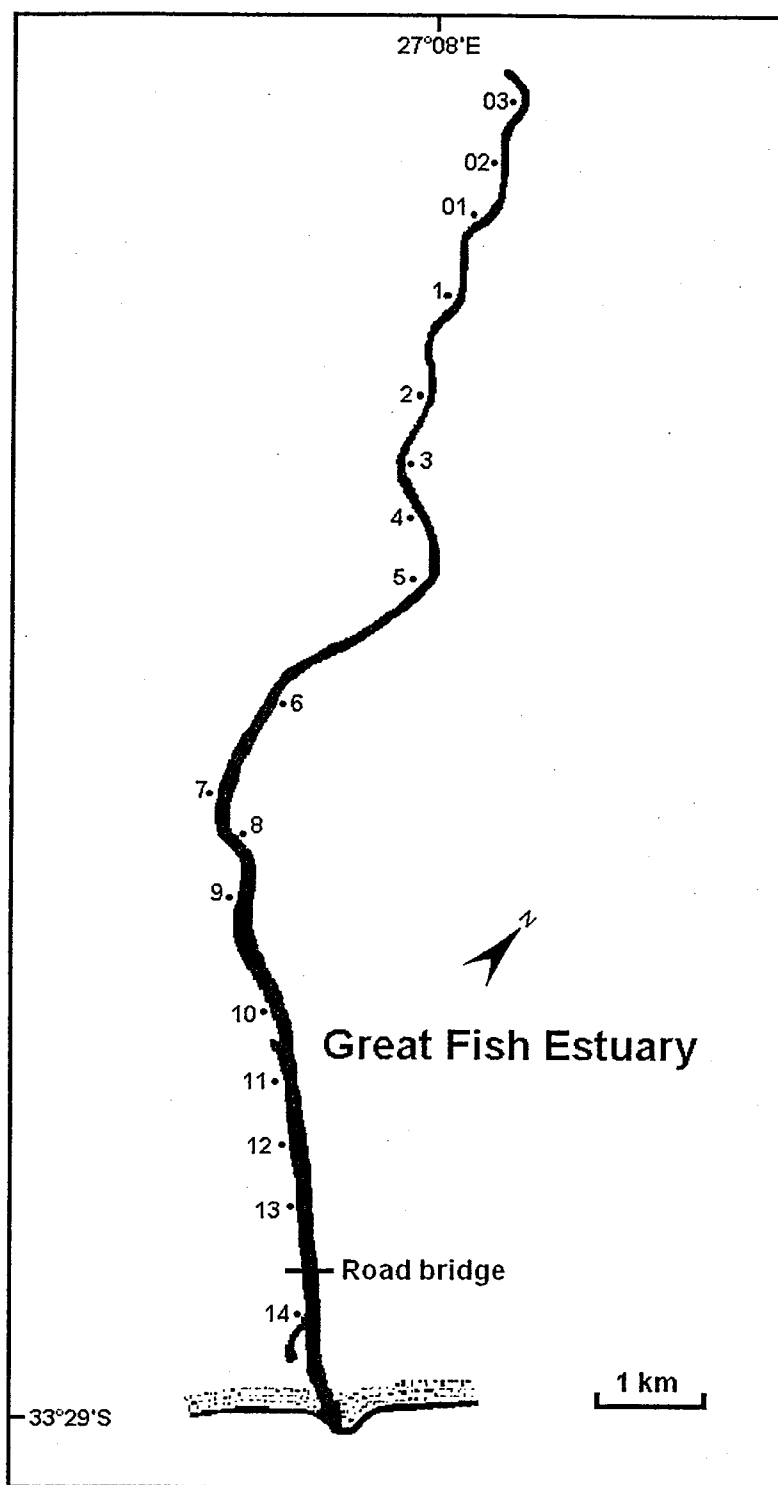


Figure 6.1.1: Map of the Great Fish River system showing sampling sites used in this study.

6.1.5 Results and discussion

6.1.5.1 Physical parameters

Estuarine researchers regard levels of > 80.0 NTU as very turbid (Cyrus 1988). Therefore the turbidities recorded in the Great Fish system, with means in the vicinity of 200.0 NTU (Figure 6.1.2), can only be described as highly turbid. Temperature varied between 12.0°C in winter and 29.0°C in summer. The ANOVA indicated no significant difference between sites, and between sites and the adjacent channel for either turbidity or temperature. There was a highly significant ($P < 0.01$) difference between sites for salinity, which was subsequently used as the factor for differentiating between the regions. Measurements of < 1.0 ‰ were regarded as riverine, 1.0 to 4.0 ‰ demarcated the head region, and > 4.0 ‰ the remainder of the estuary. Sites 03 to 3 were always riverine, sites 11 to 14 always estuarine and sites 4 to 10 varied according to the freshwater flow rate at the time of sampling.

No relationship was evident between either salinity and temperature or turbidity and temperature. A slight negative correlation was found between salinity and turbidity, which concurs with the findings of Whitfield *et al.* (1994), but this inverse relationship was not statistically significant. Estuarine water samples collected at the gill net sites showed a decrease in salinity over the low tide relative to the high tide period. Water temperature at all gill net sites dropped by an average 2.0°C overnight (18h00 to 06h00).

6.1.5.2 Small seine net fish assemblage

A total of 5510 fishes were captured during this phase of the study, representing at least 23 species (Table 6.1.1). Due to their small size, several of the specimens captured were not identified to species level. Riverine samples were dominated by mugilids, the head region by both mugilids and sparids, and the estuarine samples by sparids (Table 6.1.1).

6.1.5.2.1 Species analyses

The analyses for individual species most commonly captured using the small seine revealed some interesting trends. The sparid *Rhabdosargus holubi*, was common in the river and head regions but was most abundant at higher salinities in the estuary (Table 6.1.1). The length frequency graphs for *R. holubi* show little variation between the three zones sampled, the modal size class being 10.0 to 14.0 mm SL in all three regions (Figure 6.1.3).

Mugilidae < 30.0 mm SL were abundant throughout all the sampling regions (Table 6.1.1). *Liza richardsonii* was absent from the riverine samples, but among the most common species captured in the head and estuary regions. The head region was favoured by *Mugil cephalus*, with the modal size class being 40.0 to 59.0 mm SL in all three regions. Juveniles (< 30.0 mm FL) of this species are known to recruit into Eastern Cape estuaries during the winter months (Bok 1984). As the small seine net was only used during spring and summer, this smaller size class would not be expected to dominate catches.

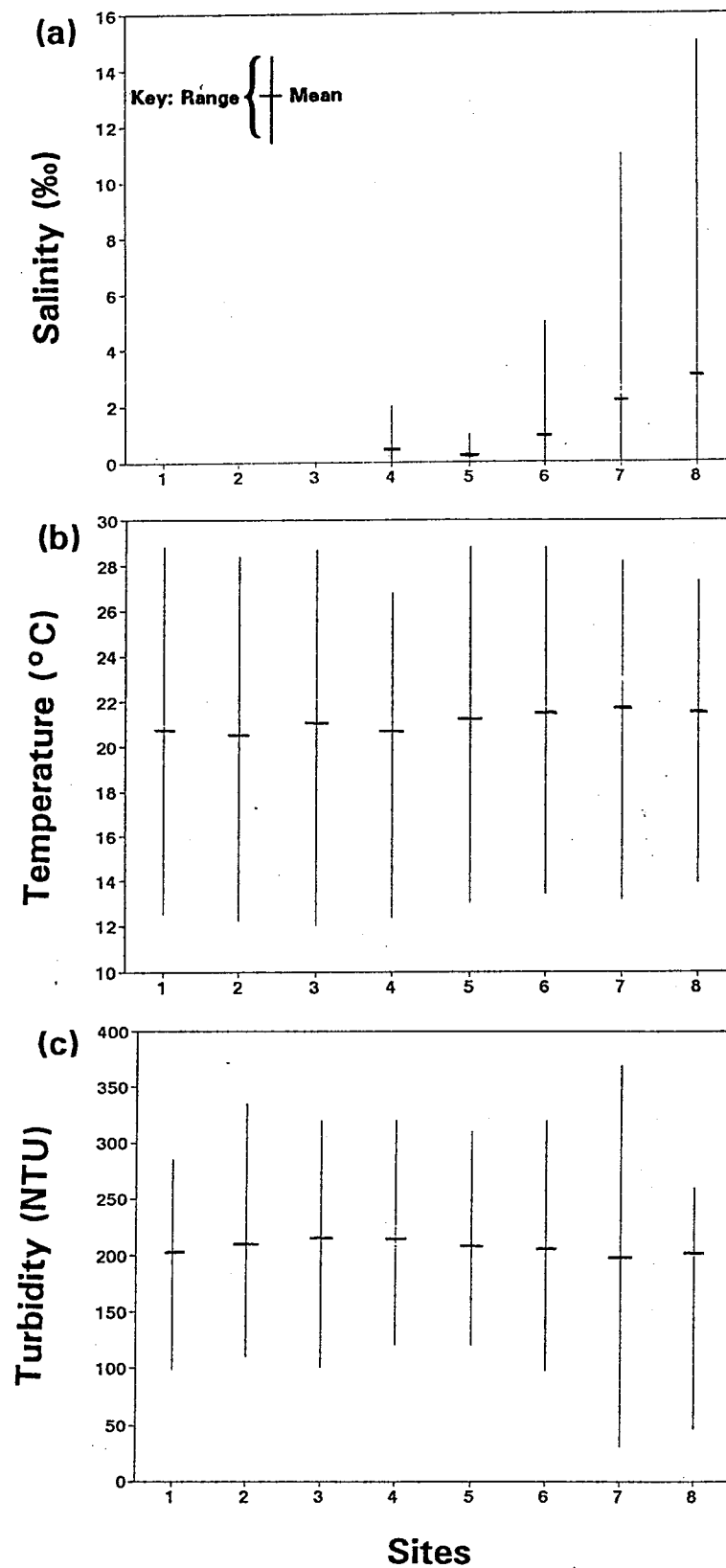


Figure 6.1.2: Mean and ranges of sampling site salinities (a), temperatures (b) and turbidities (c) recorded at sites 1 to 8 on the Great Fish River system from November 1992 to January 1995.

Pomadasys commersonnii showed a trend of increasing numbers from the river to the estuary region (Table 6.1.1). No specimens < 30.0 mm SL were sampled at any of the riverine sites where the modal size was 40.0 to 49.0 mm SL. The modal size class in the estuary was 10.0 to 19.0 mm SL, which is smaller than the size classes recorded by Wallace and van der Elst (1975) in KwaZulu-Natal estuaries. Although *Lithognathus lithognathus* was the fourth most abundant species captured with the small seine net, it was only common in the estuary (Table 6.1.1). Conversely, another sparid, *Acanthopagrus berda*, was recorded in the river but not in the estuary.

Monodactylus falciformis was more abundant in the riverine samples compared to those from either the head or estuary regions (Table 6.1.1). This marine species has been recorded from riverine areas elsewhere on the subcontinent (Skelton 1993), and studies on Eastern Cape estuaries have found this species to be most abundant in the upper reaches (Beckley 1984; Hanekom and Baird 1984). In addition, the 0+ age group *M. falciformis* are known to associate with vegetation cover and the riverine areas of the Great Fish system are characterised by inundated grass along the banks.

While *Gilchristella aestuaria* was the fourth most common species to be captured with the small seine net in the estuary, none were captured in the head region and few in the river (Table 6.1.1). These results suggest that *G. aestuaria* avoids shallow areas in the upper parts of the system.

Although the dominant freshwater species, *Labeo umbratus*, showed a clear preference for the river and head regions (Table 6.1.1), the modal size class for this species was constant in all three regions at 10.0 to 19.0 mm SL. The other major freshwater taxon, the cichlid *Oreochromis mossambicus*, appeared to be confined to the river region, where they were present in a wide range of sizes (9.0 to 169.0 mm SL).

6.1.5.2.2 Life-history analysis

River: Almost 66 % of the fishes caught with the small seine net in the river were unidentified Mugilidae. Based on the presence of larger identifiable size classes in the small seine net samples, and their known abundance in the Great Fish River (Bok 1983), the bulk of these unidentified specimens were probably *Mugil cephalus* and *Myxus capensis*. Euryhaline marine species, excluding unidentified mugilids, comprised two thirds of the small seine net catches in the river region (Figure 6.1.4 A). The most abundant species in this category from the riverine samples were 0+ *R. holubi*, *M. falciformis* and *Heteromycteris capensis* (Table 6.1.1). Freshwater species were the second most numerous in the riverine catches (Figure 6.1.4 A), with *Redigobius dewaali* and *L. umbratus* being the two most common species (Table 6.1.1). The estuarine category, comprising three species (*Glossogobius callidus*, *Psammogobius knysnaensis* and *G. aestuaria*), made up < 10 % of the fishes from the riverine samples (Figure 6.1.4 A).

Table 6.1.1: The mean number of fish per haul and ranking of fishes captured in each of the three regions of the Great Fish River system using a small seine net.

Family	Species	Life-history category	River (< 1‰)	Rank	Head (1 to 4‰)	Rank	Estuary (> 4‰)	Rank
Anguillidae	Unidentified	Va	0.03	23			0.09	15
Clupeidae	<i>Gilchristella aestuaria</i>	Ia	0.31	13			3.55	4
Elopidae	<i>Elops machnata</i>	Iia	0.69	11				
Cyprinidae	<i>Cyprinus carpio</i>	IV	0.09	19				
	<i>Labeo umbratus</i>	IV	1.09	7	1	6	0.09	15
Poeciliidae	<i>Gambusia affinis</i>	IV	0.28	16				
Haemulidae	<i>Pomadasys commersonnii</i>	Iia	0.78	9	2	3	3.45	5
Sparidae	<i>Acanthopagrus berda</i>	Iia	0.09	19				
	<i>Lithognathus lithognathus</i>	Iia	0.03	23	0.17	10	4.91	3
	<i>Rhabdosargus holubi</i>	Iia	7.28	2	19.5	2	226.45	1
Sciaenidae	<i>Argyrosomus japonicus</i>	Iib	0.19	17	0.17	10	0.09	15
Monodactylidae	<i>Monodactylus falciformis</i>	Iia	3.72	3			0.18	12
Cichlidae	<i>Oreochromis mossambicus</i>	IV	0.88	8				
Mugilidae	Juveniles < 30 mm SL		39.94	1	21.17	1	38.91	2
	<i>Liza dumerilii</i>	Iib	0.06	22	0.17	10	0.18	12
	<i>Liza richardsonii</i>	Iic			1.33	5	2.45	6
	<i>Mugil cephalus</i>	Vb	0.31	13	2	3	1.18	7
	<i>Myxus capensis</i>	Vb	0.72	10			0.64	10
Gobiidae	<i>Caffrogobius nudiceps</i>	Ib			0.17	10	0.82	8
	<i>Glossogobius callidus</i>	Ib	0.69	11			0.09	15
	<i>Psammogobius knysnaensis</i>	Ia?	0.31	13	0.33	9	0.82	8
	<i>Redigobius dewaali</i>	IV	2.81	4				
Soleidae	<i>Heteromycteris capensis</i>	Iib	1.72	6			0.18	12
	<i>Solea bleekeri</i>	Iib	0.19	17				
Unidentified			0.09	19	0.67	7	0.36	11
Number of seine hauls			32		6		11	
Mean number of fish per haul			64.8		49.4		284.4	
Species richness (R)			2.9		1.8		1.8	
Species diversity (H)			2.2		1.1		0.4	
Evenness (E)			0.7		0.5		0.2	

Rhabdosargus holubi

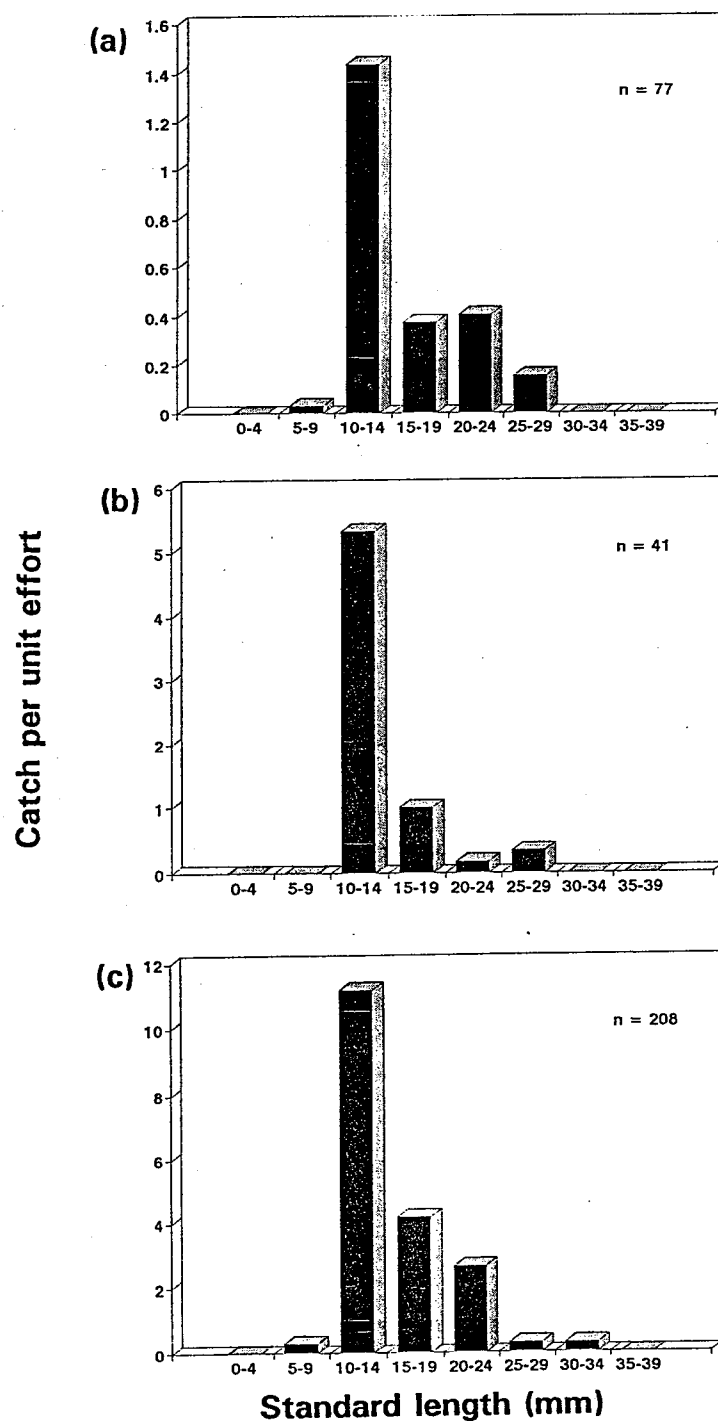


Figure 6.1.3: The length frequency distribution of *Rhabdosargus holubi* captured with a small seine net in the Great Fish River (a), head region (b) and estuary (c).

Head region: The mugilids were also the most abundant family captured with the small seine net in the head region (Table 6.1.1). *R. holubi* was the single most abundant species, contributing almost as much to the total catch as the mugilids. Juvenile euryhaline marine fish again dominated the catch (Figure 6.1.4 B), largely due to the contribution made by *R. holubi*, along with *P. commersonnii* and *L. richardsonii*. As with the riverine samples, the known catadromous species (made up entirely of *M. cephalus*) represented < 10 % of the overall total for the head region, but were the second most abundant category (Figure 6.1.4 B). *Caffrogobius nudiceps* and *P. knysnaensis* were the only estuarine species recorded and *L. umbratus* the only freshwater species (Table 6.1.1).

Estuary: The vast majority of fish captured were *R. holubi*, thus contributing to the dominance of euryhaline marine species from this region (Figure 6.1.4 C). Mugilids were the second most abundant family, with the second most abundant species being *L. lithognathus* (Table 6.1.1). Other juvenile euryhaline fish common in this area were *P. commersonnii* and *L. richardsonii*.

System: Overall trends indicate that 0+ euryhaline marine fish dominated the small seine net catches in all three sampled regions (Figure 6.1.4 D). Mugilidae < 30.0 mm SL were the most abundant taxa in both the riverine and head regions of the system, but could not be included in the classification due to the family having representatives from at least two of the life-history categories. The single most abundant species, common in all three regions, was *R. holubi* (Table 6.1.1), with *P. commersonnii* being the only other species regularly recorded throughout the sampled area. Several taxa were common in one or two of the regions but totally absent from the others. Freshwater species, with the exception of *L. umbratus*, were only captured in the riverine area.

On comparing the distribution of species to their life-history categories, a general trend of decreasing use of the riverine habitat with a decreasing dependence on estuaries, can be observed, i.e. those species which are most dependant on estuaries tend to move furthest up the system. Juvenile *Acanthopagrus berda*, *Elops machnata*, *L. lithognathus*, *P. commersonnii* and *M. falciformis* (category IIa) all utilise the riverine region freely, often more so than the head region or rest of the estuary. At the other end of the spectrum, *L. richardsonii* (category IIc) was one of the most commonly caught species in the estuary and head regions, but was absent from the riverine samples (Table 6.1.1).

The indices all indicated that diversity was highest in the river and lowest in the estuary (Table 6.1.1), due mainly to the lack of overwhelming dominance by any one taxon in this zone. Differences between the regions was greatest in the case of the species diversity index (H) and least when analysed using the species richness index (R).

6.1.5.3 Large seine net fish assemblage

A total of 36 172 fishes, representing 41 species and 24 families, were captured during this sampling programme (Table 6.1.2). Mugilidae dominated the samples throughout the system, comprising 44 % of the catch. Whitfield *et al.* (1994) found the Great Fish Estuary to have higher densities of mugilids than the adjacent Kowie system and suggested that this may be due to the large riverine input of organic material. Altogether 37 fish species were recorded in the river, 33 in the head region and 32 in the estuary. Species richness (R) was moderately high in all three sampled regions (3.24 to 3.58)

when compared to similar gear results from other Eastern Cape estuaries (Whitfield *et al.* 1989; Ter Morshuizen and Whitfield 1994), and decreased from the river to the estuary. This differs from the usual trend of decreasing species richness from the lower to the upper reaches of estuaries (Plumstead *et al.* 1989). Such a result may indicate that the Great Fish system differs significantly from other systems in the Eastern Cape, or that a lack of sampling in other systems has caused the relatively high species richness associated with the head regions and rivers above those estuaries to be overlooked.

6.1.5.3.1 Species analyses

In keeping with similar studies in other South African estuaries (Beckley 1984; Whitfield *et al.* 1989), the estuarine resident *G. aestuaria* was the most abundant species, comprising 22.3 % of the total catch. As was the case with the small seine net, the largest numbers of *G. aestuaria* captured with the large seine net were recorded in the estuary (Table 6.1.2). Since copepods have been shown to be the main prey item of *G. aestuaria* (Talbot and Baird 1985), and Grange *et al.* (2000) found the greatest abundance of copepods at salinities of approximately 8.0 ‰ in the Great Fish Estuary, it was to be expected that the highest densities of this species would be in the same region. Talbot (1982) determined that *G. aestuaria* suffers 99 % mortality by the age of two years in the Swartkops Estuary due to bird and fish predation. Thus it is not surprising that the modal class for all three regions on the Great Fish system was only 40.0 to 49.0 mm SL (Figure 6.1.5). Some large specimens (> 70.0 mm SL) were captured in the riverine section, including one specimen of 93.0 mm SL, which is greater than the maximum size of 60.0 mm SL for this species (Whitehead and Wongratana 1986). In addition to this, specimens from this system are deep bodied (depth in SL = 3.94) which is similar to those from Lake St Lucia (depth in SL = 3.92) (Blaber *et al.* 1981).

Rhabdosargus holubi was one of the most abundant species in the estuary (Table 6.1.2). A large number of individuals were recorded in 'freshwater', which contradicts the findings of Blaber (1973) who found freshwater to be lethal for this species at all temperatures between 5 and 35°C. The latter finding was based on fish transferred directly from sea water (35.0 ‰) to the experimental salinity and the abruptness of such a move, rather than the actual values themselves, may have caused the mortalities (Dallas and Day 1993). Fish < 80.0 mm SL were most common and no individuals > 120.0 mm SL were captured at any of the sites (Figure 6.1.6). Other workers have shown that this species is dependant on estuaries as nursery areas, and migrates to sea to join adult populations from about 120.0 mm TL (Beckley 1984).

The most common euryhaline marine species captured with the 30 m seine net was *Liza dumerilii*. This is one of the few species that was equally common at all sampled salinities (Table 6.1.2), with the majority measuring between 30.0 and 90.0 mm SL (Figure 6.1.7). Similarly, *L. richardsonii* was most abundant in the size range 30.0 and 89.0 mm SL, which probably reflects the selectivity of the seine net. Although plentiful throughout the sampling area, *M. cephalus* was most common in the estuary (Table 6.1.2).

Pomadasys commersonnii are attracted to turbid estuaries such as the Great Fish where the low visibility has little negative effect on the foraging efficiency of this species (Hecht and van der Lingen 1992). Juveniles were most abundant in the head region and were ranked fourth in the riverine section (Table 6.1.2). The modal length class for all sites on the Great Fish system was 30.0 to 59.0 mm SL (Figure 6.1.8), which is similar to the modal class of 40.0 to 80.0 mm TL for this species from KwaZulu-Natal estuaries (Wallace 1975).

Juvenile *A. japonicus* were most plentiful in the lower salinities of the head region (Table 6.1.2). Although abundant in the Great Fish system, juvenile *A. japonicus* are scarce in clearer estuaries (Hanekom and Baird 1984; Beckley 1983; Whitfield *et al.* 1994) and the surf zone (Bennett 1988; Whitfield 1989), suggesting that turbid estuaries are an important nursery area for first year juveniles. The modal size class for the river and head region was 30.0 to 59.0 mm SL, and that of the estuary 120.0 to 149.0 mm SL (Figure 6.1.9). This spatial segregation according to size may well reduce cannibalism within this predatory species.

Small seine net

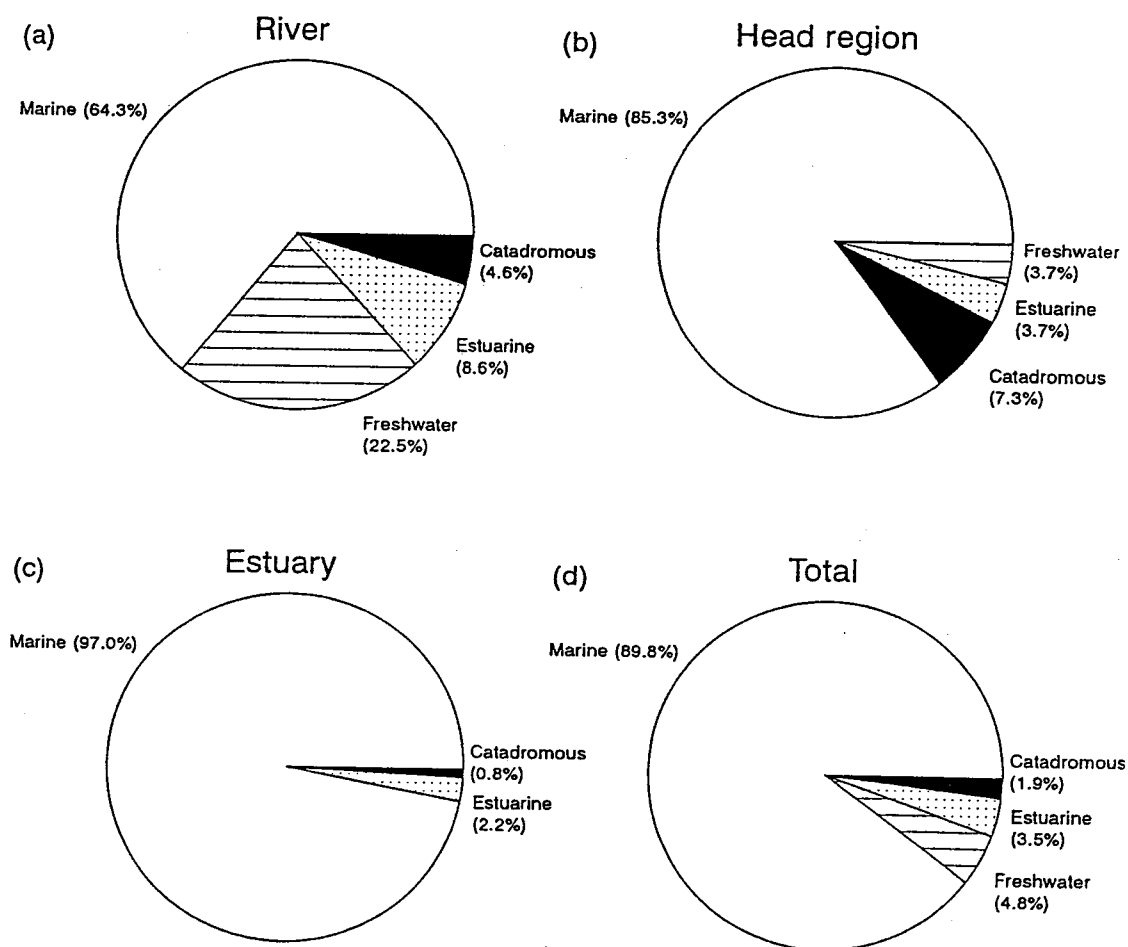


Figure 6.1.4: The proportion contributed by each life history category (Whitfield 1994) to the small seine net catch in the Great Fish River (a), head region (b), estuary (c) and total area sampled (d).

Gilchristella aestuaria

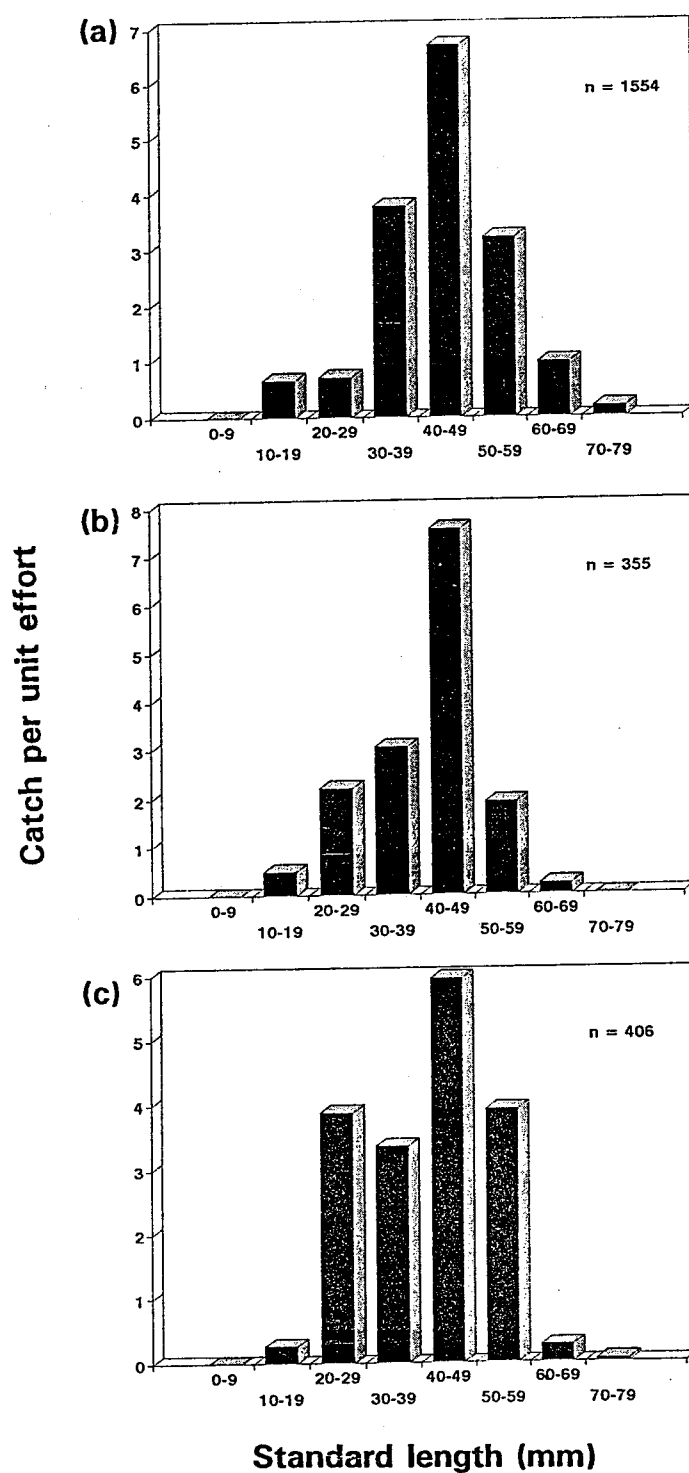


Figure 6.1.5: The length frequency distribution of *Gilchristella aestuaria* captured with a large seine net in the Great Fish River (a), head region (b) and estuary (c).

Table 6.1.2: The mean number of fish per haul and ranking of fishes captured in each of the three regions of the Great Fish River system using the large seine net.

Family	Species	Life-history category	River (< 1 ‰)	Rank	Head (1 to 4 ‰)	Rank	Estuary (> 4 ‰)	Rank
Clupeidae	<i>Gilchristella aestuaria</i>	Ia	42.26	3	79.52	2	96.96	1
Elopidae	<i>Elops machnata</i>	IIa	2.21	10	5.7	11	0.13	24
Cyprinidae	<i>Cyprinus carpio</i>	IV	1.24	13	2.7	14	0.04	29
	<i>Labeo umbratus</i>	IV	0.91	18	0.52	21	1.04	14
Clariidae	<i>Clarias gariepinus</i>	IV	0.23	23				
Ariidae	<i>Galeichthys feliceps</i>	IIb	0.47	21	2.43	16	0.09	25
Hemiramphidae	<i>Hyporhamphus capensis</i>	Ia	0.01	34				
Atherinidae	<i>Atherina breviceps</i>	Ib			0.3	24		
Platycephalidae	<i>Platycephalus indicus</i>	IIc	0.01	34	0.17	27	0.09	25
Ambassidae	<i>Ambassis gymnocephalus</i>	Ib?	1.24	13	6.26	9	26.25	5
Teraponidae	<i>Terapon jarbua</i>	IIa	0.06	26	0.22	25		
Carangidae	<i>Caranx sexfasciatus</i>	IIb	0.06	26	0.83	19	0.09	25
	<i>Lichia amia</i>	IIa	0.03	28	0.09	30	0.22	20
Leiognathidae	<i>Leiognathus equula</i>	IIc			0.04	31		
Gerreidae	<i>Gerres acinaces</i>	IIb	0.03	28				
Haemulidae	<i>Pomadasys commersonnii</i>	IIa	11.25	4	65	4	10.52	8
	<i>Pomadasys olivaceum</i>	III			0.04	31		
Sparidae	<i>Acanthopagrus berda</i>	IIa	1.42	12	1.17	18	0.3	18
	<i>Lithognathus lithognathus</i>	IIa	0.6	20	3.87	13	6.65	9
	<i>Rhabdosargus holubi</i>	IIa	5.25	7	104.87	1	77.13	2
Sciaenidae	<i>Argyrosomus japonicus</i>	IIb	5.15	8	8.61	7	1.39	13
	<i>Johnius dussumieri</i>	IIc	0.02	31				
Monodactylidae	<i>Monodactylus falciformis</i>	IIa	1.2	16	0.35	23	0.26	19
Cichlidae	<i>Oreochromis mossambicus</i>	IV	5.56	6	0.39	22	0.04	29
Mugilidae	Juveniles < 30 mm SL		48.21	1	62.04	5	16.57	6
	<i>Crenimugil crenilabis</i>	IIc	0.03	28	0.13	28	0.13	21
	<i>Liza dumerilii</i>	IIb	42.78	2	70.35	3	56.2	3
	<i>Liza richardsonii</i>	IIc	1.22	15	2.74	15	31.17	4
	<i>Liza tricuspidens</i>	IIb	0.07	24			0.13	21
	<i>Mugil cephalus</i>	Vb	7.22	5	12.61	6	10.57	7
	<i>Myxus capensis</i>	Vb	4.17	9	0.3	24	4.83	10
Polynemidae	<i>Polydactylus plebius</i>	III?			0.13	28		
Gobiidae	<i>Caffrogobius gilchristi</i>	Ib	0.07	24	0.22	25	0.7	16
	<i>Caffrogobius nudiceps</i>	Ib	0.02	31	1.65	17	0.91	15
	<i>Eleotris fusca</i>	Ia?	0.02	31				
	<i>Glossogobius callidus</i>	Ib	1.15	17	0.78	20	0.04	29
	<i>Oligolepis keiensis</i>	Ia?					0.09	25
	<i>Psammogobius knysnaensis</i>	Ia?	0.72	19	6.7	8	2.09	12
	<i>Redigobius dewaali</i>	IV	0.03	28				
Soleidae	<i>Heteromycteris capensis</i>	IIb	0.26	22	6.22	10	2.61	11
	<i>Solea bleekeri</i>	IIb	1.94	11	4.43	12	0.65	17
Tetraodontidae	<i>Amblyrhynchotes honckenii</i>	III					0.09	25
Unidentified							0.04	29

Rhabdosargus holubi

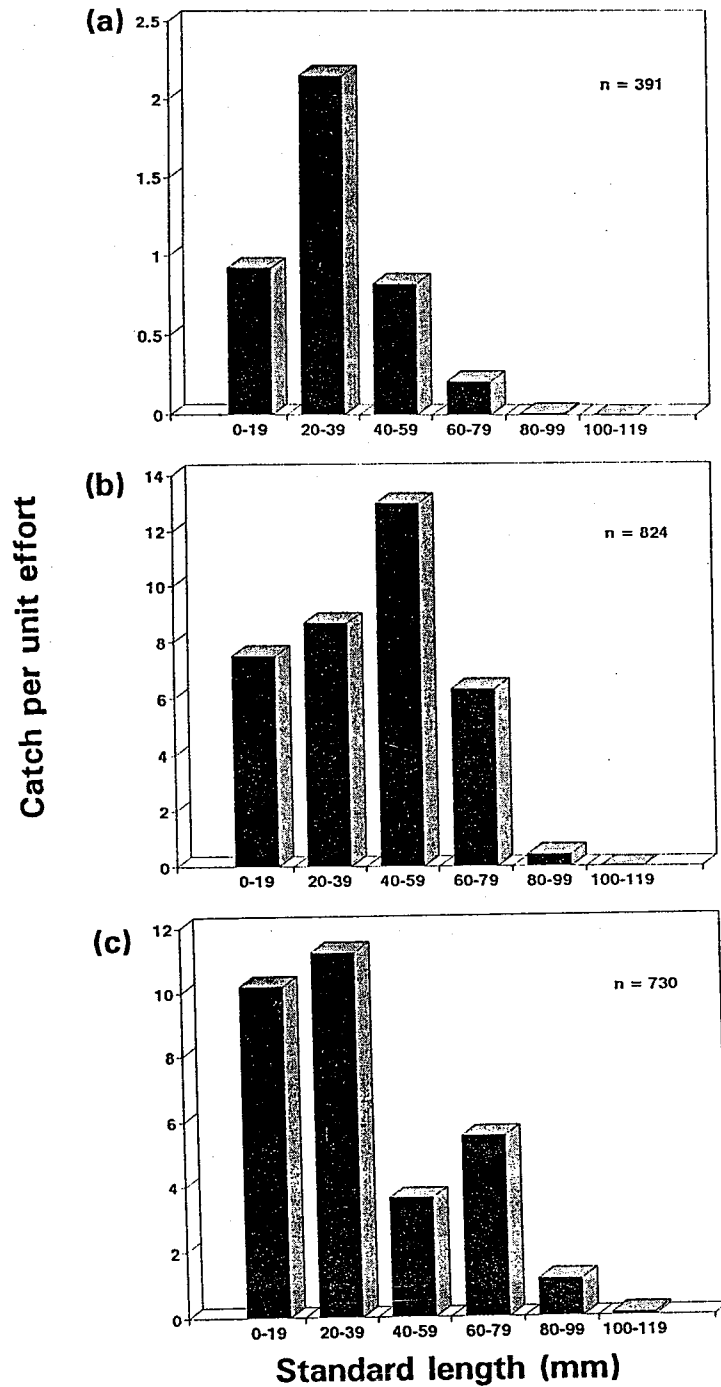


Figure 6.1.6: The length frequency distribution of *Rhabdosargus holubi* captured with a large seine net in the Great Fish River (a), head region (b) and estuary (c).

Liza dumerilii

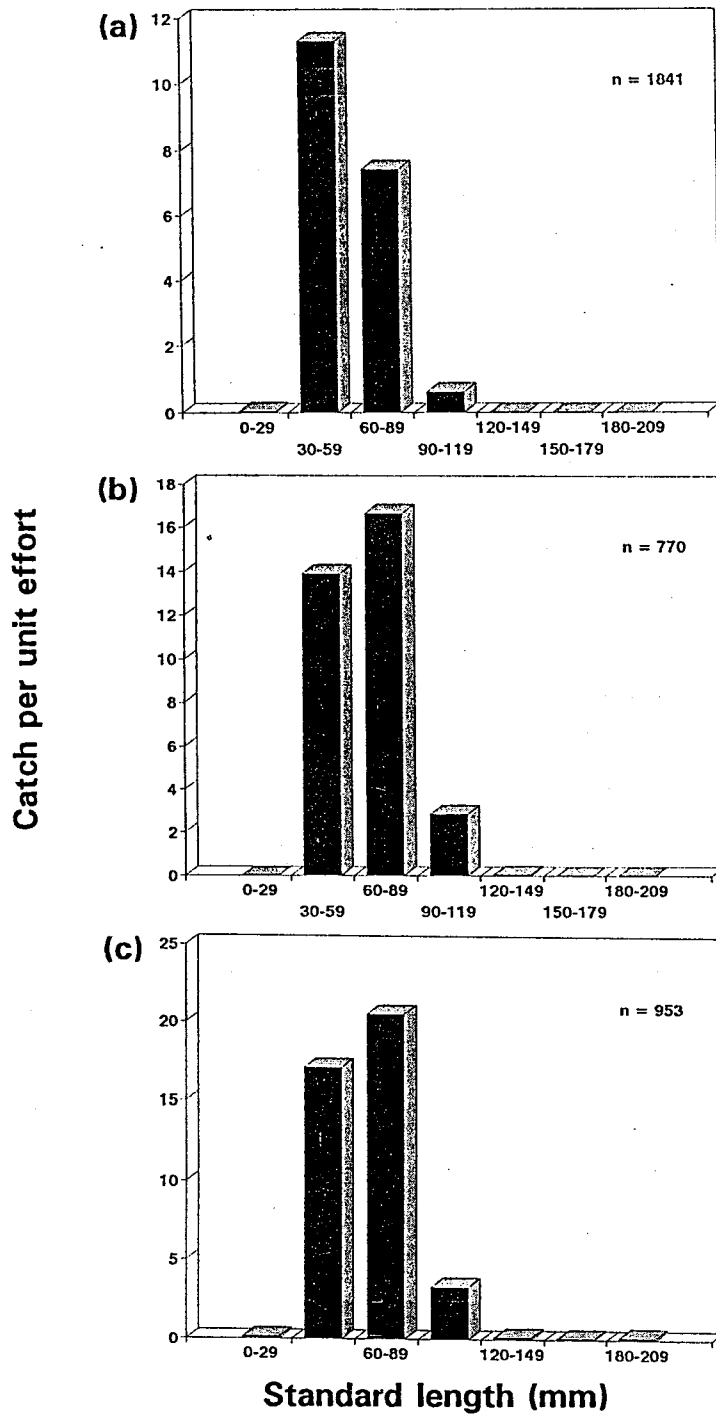


Figure 6.1.7: The length frequency distribution of *Liza dumerilii* captured with a large seine net in the Great Fish River (a), head region (b) and estuary (c).

The only abundant freshwater species, *O. mossambicus*, was most common in the riverine samples (Table 6.1.2) despite its ability to survive highly saline conditions in estuaries (Whitfield *et al.* 1981). This study has confirmed that *O. mossambicus* tend to avoid estuaries during their open phase, and remain in the river above the estuarine zone until such time as they close (Whitfield and Blaber 1978).

6.1.5.3.2 Life-history analysis

River: Almost 54 % of the large seine net catch from the river were euryhaline marine fish (Figure 6.1.10 A), with *L. dumerilii* being the most abundant species (Table 6.1.2). The other marine taxa common in this area were *P. commersonnii*, *R. holubi*, *A. japonicus* and *E. machnata* (Table 6.1.2). Estuarine taxa accounted for about one third of the riverine catch (Figure 6.1.10 A), with *G. aestuaria* being the only abundant species in this category (Table 6.1.2). The two catadromous mugilids, *M. cephalus* and *M. capensis*, were both common in the riverine samples and together comprised almost 10 % of the total catch. However, a large proportion of the unidentified Mugilidae (< 30.0 mm SL) are likely to be either *M. cephalus* or *M. capensis*. The only abundant freshwater species in the riverine section was *O. mossambicus* (Table 6.1.2).

Head region: Euryhaline marine species dominated the large seine net catch at the head of the Great Fish Estuary (Figure 6.1.10 B), with *R. holubi*, *P. commersonnii* and *L. dumerilii* being very abundant (Table 6.1.2). Estuarine taxa were the second most abundant category, with *G. aestuaria* being the most common species followed by *P. knysnaensis* and *A. gymnocephalus*. Of the two catadromous species, only *M. cephalus* was found to be common (Table 6.1.2), although *M. capensis* may have been well represented in the unidentified juvenile mugilid samples. Freshwater species were seldom recorded at the head of the estuary (Figure 6.1.10 B).

Estuary: Estuarine samples were also dominated by euryhaline marine species (Figure 6.1.10 C), with *R. holubi*, *L. dumerilii*, *L. richardsonii* and *P. commersonnii* being the main contributors (Table 6.1.2). The estuarine category was the second most abundant, mainly due to *G. aestuaria* and *A. gymnocephalus*. Both *M. cephalus* and *M. capensis* were fairly common in the estuary, while freshwater species were uncommon (Figure 6.1.10 C) although *L. umbratus* was recorded in low numbers (Table 6.1.2).

System: The overall analysis of species captured with the large seine net, according to life-history categories, shows similar trends to those observed for small seine net catches. In both instances the euryhaline marine category was the most abundant in all three regions (Figures 6.1.4 and 6.1.10), and the common taxa captured were Mugilidae (< 30.0 mm SL), *G. aestuaria*, *R. holubi*, *P. commersonnii*, *L. dumerilii* and *M. cephalus* (Tables 6.1.1 and 6.1.3). The indices for the large seine net also showed diversity to be highest in the river ($R = 3.58$, $H = 2.09$) and lowest in the estuary ($R = 3.24$, $H = 1.98$), although evenness was similar throughout the sampled area ($E = 5.8 - 6.0$). Many of the species captured during the present study showed an increase in body length from the river to the estuary. This trend supports the hypothesis of Rogers *et al.* (1984) that many juvenile fish utilise estuarine headwaters first and move down the estuary as they grow.

Pomadasys commersonnii

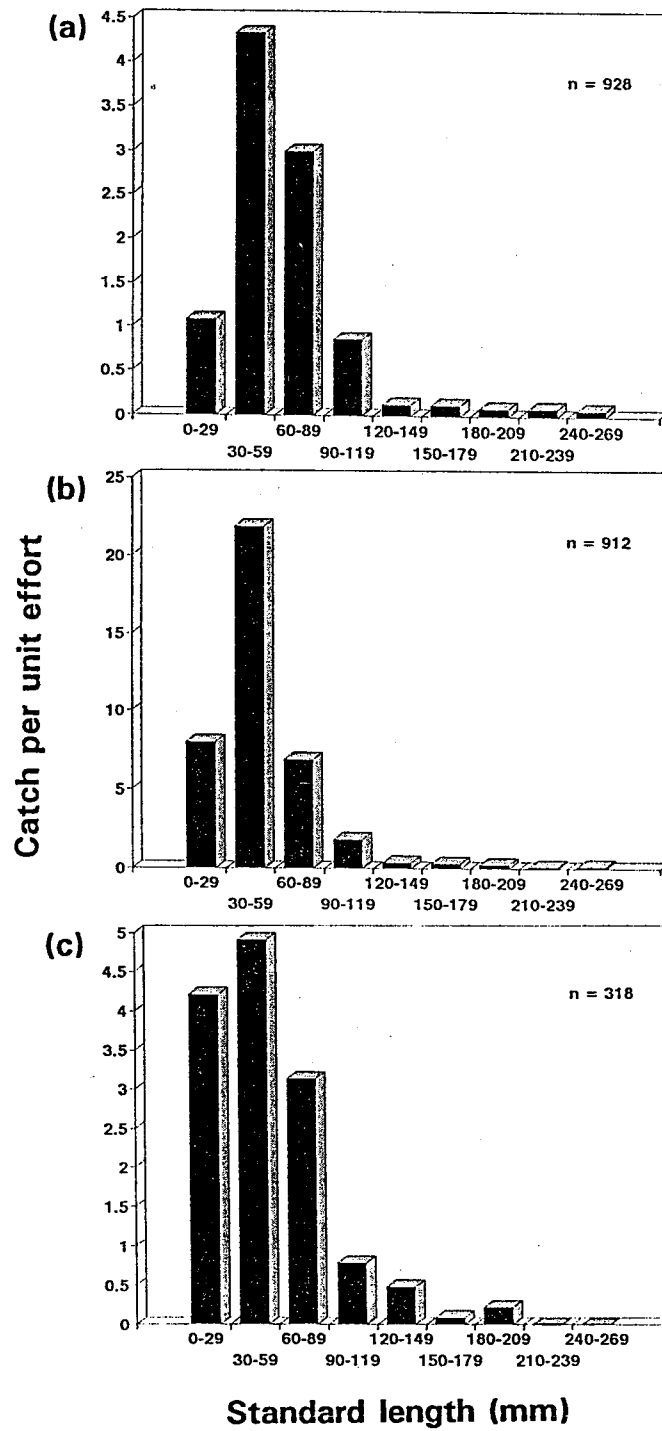


Figure 6.1.8: The length frequency distribution of *Pomadasys commersonnii* captured with a large seine net in the Great Fish River (a), head region (b) and estuary (c).

Argyrosomus japonicus

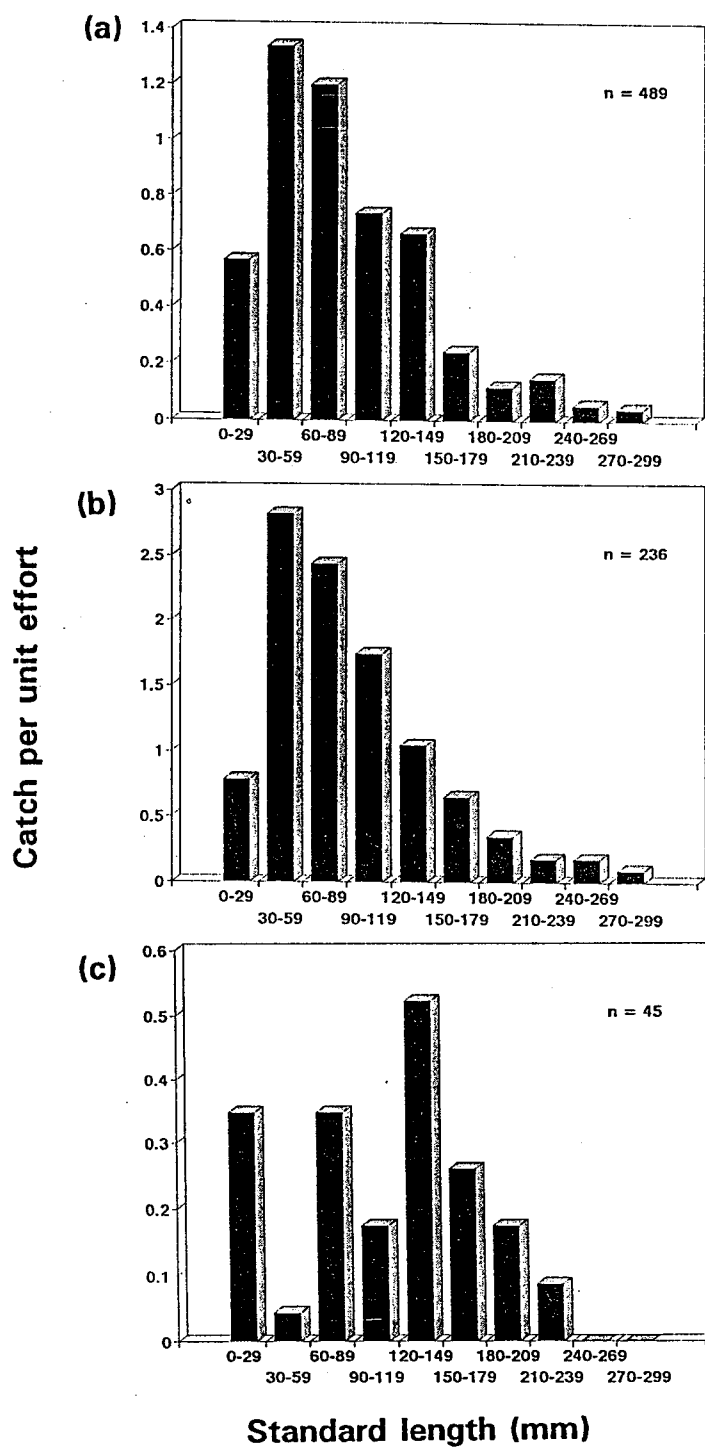


Figure 6.1.9: The length frequency distribution of *Argyrosomus japonicus* captured with a large seine net in the Great Fish River (a), head region (b) and estuary (c).

6.1.5.4 Gill net fish assemblage

A total of 511 fishes representing 15 species were captured by gill netting during this study (Table 6.1.3). *Pomadasys commersonnii* was the most abundant species, followed by *A. japonicus* and *M. cephalus*. Whitfield *et al.* (1994), using a range of gill net mesh sizes in the Great Fish Estuary, found *L. richardsonii* to be the most abundant species, followed by *M. cephalus* and *P. commersonnii*.

During this study *P. commersonnii* was common throughout the sampled area, but was most abundant in the upper estuarine and head regions. In the Gamtoos Estuary they were also most abundant in the upper estuary (Marais 1983b), whereas they were rarely recorded at the head of the Sundays (Marais 1981) or Great Fish estuaries (Whitfield *et al.* 1994). *Pomadasys commersonnii* was most frequently recorded in the highly saline upper reaches of the freshwater deprived Kromme Estuary (Marais 1983a), indicating that the lower salinity recorded at the head of the Great Fish system is unlikely to be the sole reason for this species being abundant there during the present study. The relatively high densities of *P. commersonnii* in the Swartkops Estuary and lower abundance in the Sundays Estuary was attributed to the relatively high biomass of the mudprawn *Upogebia africana* in the former system (Marais 1981, 1983b). The abundance of *P. commersonnii* in the Great Fish Estuary, where large *U. africana* populations are present, was therefore to be expected.

Argyrosomus japonicus was also most abundant at the estuary head during the present study, thus confirming the findings of Whitfield *et al.* (1994) in the Great Fish, Marais and Baird (1980) in the Swartkops, and Marais (1981) in the Sundays systems. In the clear Kromme Estuary, which lacks a normal axial salinity gradient, this species was evenly distributed throughout the system (Marais 1983a), whereas in the turbid Gamtoos and Great Fish estuaries, which both have a strong riverine input, they were most abundant in the upper estuary (Marais 1983b; Whitfield *et al.* 1994).

In the Gamtoos (Marais 1983b) and Great Fish systems, *M. cephalus* was far more common in the river, where large numbers were caught on several occasions. In addition, Marais (1981) found this species to be most abundant in the upper reaches of the Sundays Estuary. Furthermore, in 1982 this species was uncommon in the Kromme system (Marais 1983a) which had a poorly developed axial salinity gradient and an average salinity in excess of 30 ‰ at all sampling sites.

The overall catch per unit effort (CPUE) for the present study indicates that the largest catches were made in the head region, followed by the river, and finally the estuary. Grange and Allanson (1995) found that as a result of strong freshwater inflow to the Great Fish Estuary, a gradient in food resource availability existed, with highest values being recorded towards the upper reaches of the estuary. It is to be expected therefore that the highest densities of fish should occur in the head region where food resources are greatest. When the CPUE results of the gill netting phase of the present study are converted to the same units used in other studies (Marais and Baird 1980; Marais 1981, 1983a,b), the catches from this study are similar to those from the Sundays and Gamtoos systems (Table 6.1.3).

Large seine net

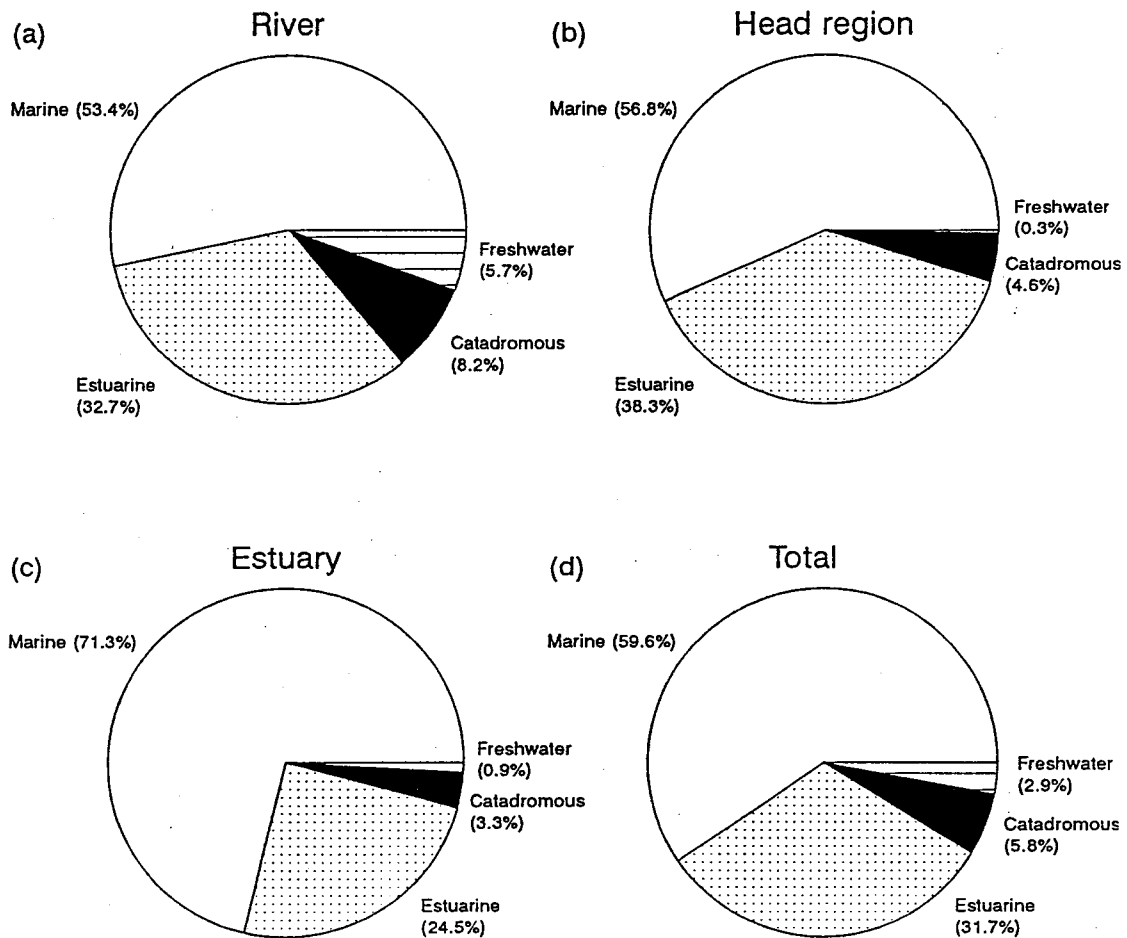


Figure 6.1.10: The proportion contributed by each life history category (Whitfield 1994) to the large seine net catch in the Great Fish River (a), head region (b), estuary (c) and total area sampled (d).

6.1.5.5 Salinity and fish distribution

Although other studies have shown that salinity and turbidity are both major factors influencing the distribution of fish species within South African estuaries (Cyrus and Blaber 1987b; Ter Morshuizen and Whitfield 1994), the lack of a distinct turbidity gradient during the present study precluded the possibility of assessing turbidity influences on fish species distribution within the sampled area.

South African estuaries are usually characterised by a longitudinal distributional pattern of fish species. The stenohaline marine species are typically confined to regions of higher salinity, whereas euryhaline species are also abundant in the less saline upper reaches (Hanekom and Baird 1984; Whitfield *et al.* 1981). In the Great Fish system, strongly euryhaline marine species dominated the estuary (> 4.0 ‰), head region (1.0 to 4.0 ‰) and sampled area of the river (< 1.0 ‰) (Figures 6.1.4 and 6.1.10). Several of these species have seldom, or never been recorded from freshwater (e.g. *R. holubi*, *P. commersonnii*, *L. dumerilii*, *L. richardsonii* and *L. lithognathus*).

Table 6.1.3: Catch per unit effort (number of fish caught per 80 m² of gill net over 6 hours) in each of the three sampled regions of the Great Fish River system.

Family	Species	River	Rank	Head	Rank	Estuary	Rank
Elopidae	<i>Elops machnata</i>	0.08	4				
Cyprinidae	<i>Cyprinus carpio</i>	0.05	6	0.11	7		
	<i>Labeo umbratus</i>	0.03	9				
Clariidae	<i>Clarias gariepinus</i>	0.03	9				
Ariidae	<i>Galeichthys feliceps</i>	0.05	6	0.22	6	0.36	3
Platycephalidae	<i>Platycephalus indicus</i>	0.03	9				
Haemulidae	<i>Pomadasys commersonnii</i>	1.58	2	4.44	1	2.82	1
Sparidae	<i>Acanthopagrus berda</i>	0.03	9				
Sciaenidae	<i>Argyrosomus japonicus</i>	0.75	3	3.22	2	1.73	2
Mugilidae	<i>Liza richardsonii</i>	0.03	9	0.11	7		
	<i>Liza tricuspidens</i>	0.08	4	0.33	5	0.36	3
	<i>Mugil cephalus</i>	5.95	1	2.22	3	0.18	5
	<i>Myxus capensis</i>			0.44	4	0.09	6
	<i>Valamugil buchanani</i>			0.11	7		
CPUE (fish.6 hrs ⁻¹ 80 m ⁻²)		8.7		11.3		5.6	
CPUE (kg.6 hrs ⁻¹ 80 m ⁻²)		9.4		12.4		5.2	
CPUE (kg.12 hrs ⁻¹ 150 m ⁻²)		35.4		46.7		19.5	
CPUE (kg.12 hrs ⁻¹ 150 m ⁻²) Sundays River Estuary						20.4	
CPUE (kg.12 hrs ⁻¹ 150 m ⁻²) Gamtoos River Estuary						33.3	

The presence of a salt wedge in the estuary is unlikely to be the major factor governing marine fish species occurrence in freshwater, since the wedge only extends as far as the head region (Ter Morshuizen 1994), and these taxa were also recorded in the river above the estuary. The high conductivity levels of the Great Fish River are a possible factor enabling these species to utilise freshwater areas within this system. Conductivity levels ranged from 150.0 to 625.0 mS m⁻¹ in the sampled region of the Great Fish River during this study, which is high when compared to other Eastern Cape rivers, e.g. Kromme River (41.0 mS m⁻¹), Kat River (43.0 mS m⁻¹) and Blaaukrantz River (10.0 mS m⁻¹). Although the tolerance of most species to freshwater conditions was surprising, it is consistent with the opinion of Whitfield *et al.* (1981) that estuarine-associated fish taxa are usually more tolerant of low rather than high salinities.

6.2 Distribution patterns of fishes in a freshwater deprived estuary, with particular emphasis on the geographical headwaters

6.2.1 Summary

Fish assemblages in the headwater region of the Kariega Estuary are described and compared to the catch composition in other parts of the system. The available evidence suggests that river flow has a major impact on the structure and functioning of fish communities within estuaries and the upper reaches in particular. The geographical headwaters of the freshwater deprived Kariega Estuary were utilised by a range of fish species but, in contrast to the nearby freshwater 'rich' Great Fish Estuary, few important angling species (e.g. spotted grunter *Pomadasys commersonnii* and dusky kob *Argyrosomus japonicus*) were recorded. The reduced riverine flow into the Kariega Estuary resulted in an extremely restricted river-estuary interface (REI) zone being made available to resident and migrant fish species. Despite the low freshwater flows recorded, the catadromous freshwater mullet *Myxus capensis* was abundant within the REI zone and headwater reach, possibly due to the lack of riverine habitat available for the juveniles of this species. Although salinity has been shown to be an important structuring force influencing ichthyofaunal assemblages in many South African estuaries, it is not the primary factor governing the distribution of fish in a freshwater deprived, marine dominated system such as the Kariega.

6.2.2 Introduction

Biological research in South African estuaries has been concentrated on the lower and middle reaches of systems, with little of information being available from the geographic headwaters or upper reaches. Research on fish communities in South African estuaries has followed a similar pattern, with few studies examining the entire length of an estuary, the notable exception being a recent study on the Great Fish River Estuary (Ter Morshuizen *et al.* 1996). This lack of information on the longitudinal distribution of fish within estuaries is compounded by the paucity of studies that rigorously examine the factors, both abiotic and biotic, which structure these communities. Advances in multivariate statistics have enhanced numerical ecology, but the majority of studies examining spatial and temporal variation in fish communities have been analysed using conventional univariate statistics (Ter Morshuizen and Whitfield 1994). Research into the forces that structure fish communities have also focused on the relationship between fishes and a single abiotic variable (Cyrus and Blaber 1987a,b), with only a few using multivariate techniques to link fish species to a range of environmental parameters (Morin *et al.* 1992; Thiel *et al.* 1995; Marshall and Elliott 1998). This study aims to use multivariate techniques to examine the longitudinal variation in Kariega Estuary fish assemblages and the factors that influence them.

6.2.3 Study area

The permanently open Kariega Estuary is situated on the east coast of South Africa (33° 41'S, 26° 42'E) and is approximately 18.0 km long (Figure 6.2.1). The channel in the upper reaches is narrow (40 to 60 m) while in the lower reaches the estuary widens (100.0 m) and is bordered by sand flats and salt marshes (Grange 1992). The estuary has an average midstream depth of between 2.5 and 3.5 m and was formed by the drowning of a river valley following a rise in sea level (Reddering and Rust 1990).

The Kariega Estuary is a marine dominated system with very little riverine influence. The system is often hypersaline in the upper reaches and, apart from episodic freshwater inputs, river inflow is negligible for extended periods (Hodgson 1987; Allanson and Read 1995; Grange *et al.* 2000). This low freshwater input into the system is due to the Eastern Cape being relatively arid, which is exacerbated by a very poor rainfall to runoff conversion. In addition, the catchment of the Kariega Estuary is small (686 km²) and highly regulated by three dams and numerous farm weirs. In contrast, the marine environment has a major influence on the estuary, which is demonstrated by the 106:1 ratio of tidal prism volume to river volume (Grange *et al.* 2000). The mean spring tidal range along the Eastern Cape coast is 1.6 m and the Kariega Estuary may therefore be classified as being microtidal. Even without a strong freshwater input, scouring by tidal currents is sufficient to maintain a permanent connection with the sea.

With a strong marine influence and negligible freshwater input, the salinity is usually uniformly marine (35.0 ‰) along most of the length of the estuary. The system has a low turbidity (< 10.0 NTU) and is well mixed with almost no salinity or thermal stratification of the water column at any stage of the tidal cycle (Grange and Allanson 1995). As a consequence of the low riverine input and the resulting poor nutrient status of the system (Allanson and Read 1995; Grange and Allanson 1995), the phytoplankton stocks are low and the water column can be regarded as being oligotrophic. *Zostera capensis* occurs mostly as a littoral band just above and below the low-water spring tide level and the mean width of the beds varies from 5.2 m in the lower reaches to 3.3 m in the upper reaches (Ter Morshuizen and Whitfield 1994).

6.2.4 Materials and methods

This study was divided into two main sections, namely a headwater fish component and an overall fish distribution component. The headwater study primarily examined the fish community found in the headwaters of the Kariega Estuary and focused on species composition, diversity, life-history stages and seasonality. The fish distribution study examined fish assemblages along the entire length of the estuary to establish whether there were any relationships between a range of abiotic and biotic parameters and the spatial variation in fish assemblages.

6.2.4.1 Headwater fish study

Sampling protocol

Sampling was conducted monthly from January to December 1999. The littoral zone was sampled over high tide at sites 31 to 40 (Figure 6.2.1) using a 5 x 1.5 m anchovy seine net (10 mm stretch mesh). Each seine net haul swept an estimated 35.0 m² area. All fish were fixed in 10 % formalin on site and transported back to the laboratory for analysis.

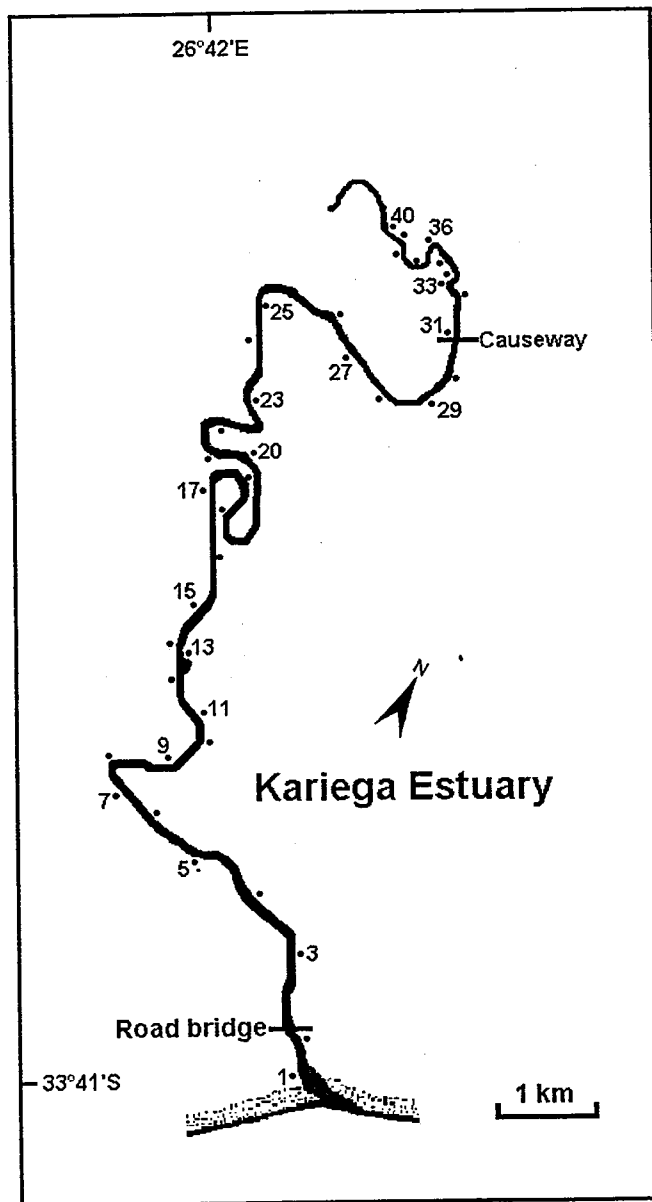


Figure 6.2.1: Map of the Kariega Estuary indicating sampling sites.

Data analysis

In the laboratory all fish were counted, identified to species level where possible and measured to the nearest mm standard length (SL). Mullet < 20.0 mm SL were recorded as Mugilidae. Average CPUE (average number of fish captured per haul), percentage composition and frequency of occurrence, was calculated for each species. The estuarine-association categories assigned to each species conform to those of Whitfield (1998) and the percentage that each category contributed to the total catch was calculated. The size at which the different fish species were utilizing the headwaters was determined by the construction of length-frequency histograms.

6.2.4.2 Fish distribution study

Sampling protocol

Samples were collected during January, May, July and October 1999. Within each season all 40 sites (Figure 6.2.1) were seined over two consecutive days using the sampling protocol described above. In addition, the percentage vegetation cover was visually assessed and water temperature, salinity, turbidity, percentage oxygen concentration and pH were measured at each site.

Data analysis

The mean ichthyofaunal density data collected during each season ($n = 40$ samples/sites) were root-root transformed to weight the contributions of common and rare species. An association matrix was produced using the Bray-Curtis similarity measure, from which classification and ordination procedures were conducted. The similarity matrix was classified using hierarchical agglomerative clustering with group average linking (Clarke and Warwick 1994). Average CPUE and percentage composition were calculated for each species in each of the six groupings identified at the 60% similarity level in the association matrix. Species richness (Margalef index) was calculated for the fish assemblages found in each of the six groupings (Washington 1984).

6.2.5 Results and discussion

6.2.5.1 Headwater fish study

6.2.5.1.1 Composition and abundance

A total of 9 362 fishes were captured, representing 19 species and 13 families. The dominant species were *Glossogobius callidus*, *Gilchristella aestuaria*, *Myxus capensis* and *Rhabdosargus holubi*, which together comprised 78.6 % of the headwaters' ichthyofauna (Table 6.2.1). The dominant families in terms of species number and percentage composition were the Mugilidae (3 species, 26.0 %), Gobiidae (4 species, 25.0 %) and Sparidae (2 species, 14.0 %). The number of species sampled in the Kariega headwaters was higher than those recorded in the adjacent Kowie Estuary (11 species, Bok 1983) and the Keskamma Estuary (15 species, Rayner 1993). In contrast to the Kariega headwaters, 32 species have been recorded in the headwaters of the Great Fish Estuary, with the

juveniles of a number of recreational angling species (e.g. *Pomadasys commersonnii*) being abundant in this reach (Ter Morshuizen *et al.* 1996). In the Kariega Estuary headwaters, juveniles of important angling species (e.g. *Argyrosomus japonicus*, *Lithognathus lithognathus* and *P. commersonnii*) were either scarce or absent (Table 6.2.1).

6.2.5.1.2 Life history analysis

Numerically the ichthyofauna was dominated by estuarine species (Category I, 53.0 %) with marine species (Category II, 31.0 %) and catadromous species (Category V, 16.0 %) also contributing significantly (Table 6.2.1). No marine stragglers (Category III) and very few freshwater fish (Category IV, 0.5 %) were caught. The vast majority of fishes captured were juveniles (Figure 6.2.2), which conforms to the proposed nursery function of estuaries (Whitfield 1998), with the headwaters of these systems being particularly important to certain species (Rogers *et al.* 1984). The major marine species *R. holubi* and *M. capensis* were dominated by young of the year with no adult fish being recorded, while the primary estuarine species *G. aestuaria* and *G. callidus* were represented by both juveniles and adults (Figure 6.2.2).

6.2.5.1.3 Seasonality

Overall fish abundance did not show any clear trends (Figure 6.2.3) but certain of the dominant species did display some seasonality. *Rhabdosargus holubi* exhibited a marked decrease in abundance during the winter months while *Mugil cephalus* was also most common in winter (Figure 6.2.4). In contrast, studies in other habitats within the Kariega Estuary (e.g. *Zostera capensis* beds in the lower and middle reaches) exhibited clear peaks in overall abundance of fish in spring and summer months (Paterson and Whitfield 2000).

6.2.5.2 Fish distribution study

6.2.5.2.1 Physico-chemical analysis

The average salinity in the Kariega Estuary was fairly constant (34.0 to 35.0 ‰) for the first 30 sites after which it dropped gradually to 23.0 ‰ at site 39. The only site that recorded salinities < 5.0 ‰ (oligohaline conditions) was site 40 that had an average salinity of 3.0 ‰. Turbidities were generally low, ranging from 5.0 to 21.0 NTU. There were no clear trends, and the lowest turbidities were generally found in the mouth and headwater regions of the estuary. The pH decreased slightly from site 1 (pH 8.0) to site 39 (pH 7.6) but then increased to a pH of 8.2 at site 40. The average temperature increased up the length of the system from 17.2°C at site 1 to 22.7°C at site 39 and then declined to 20.2°C at site 40. All sites were well oxygenated, with a minimum dissolved oxygen value of 90.0% saturation. No vegetation was recorded in the mouth region (sites 1 and 2) or the headwaters (sites 32 to 39). *Zostera capensis* was recorded from sites 3 to 31 and the trend exhibited was one of decreasing plant cover, ranging from > 90.0 % in the lower stations to < 50.0 % in the upper stations. Site 40 had a 50.0 % covering of *Potamogeton pectinatus*. While *Z. capensis* is common in the lower reaches of permanently open estuaries in the Eastern Cape, the low turbidity

and marine dominance of the system has resulted in eelgrass beds extending into the upper reaches of the Kariega system. This increased range of *Z. capensis* is characteristic of Eastern Cape estuaries that have had long-term reductions in freshwater inputs (Adams and Talbot 1992).

6.2.5.2.2. Composition and abundance

A total of 14 011 fish comprising 26 species, were sampled. The dendrogram constructed using root-root transformed abundance data clearly demonstrates that there are distinct ichthyofaunal assemblages along the length of the estuary (Figure 6.2.5). At the 25 % similarity level sites 1 to 3 are separated from the remaining sites, indicating that the mouth region of the estuary has a very different ichthyofaunal community. At the 60 % similarity level there are six clear groupings, namely the mouth (sites 1 to 3), lower (sites 4 to 6), middle (sites 5 to 16), upper (sites 17 to 30), headwaters (sites 31 to 39) and the REI region (site 40). The trend is robust with no outlying sites and the same trend is found if the dendrogram is constructed using percentage composition data.

The ichthyofaunal assemblages in the different reaches (defined at the 60 % similarity level) were % whereas the headwaters had a number of important species, *G. aestuaria* (27.0 %), *G. callidus* (23.0 %) and *M. cephalus* (16.0 %). The REI region was dominated by *M. capensis* (36.0 %), *M. falciformis* (32.0 %) and *G. callidus* (26.0 %). Very few recreationally important marine species (e.g. *P. commersonnii* and *A. japonicus*), which were common in the Great Fish Estuary (Ter Morshuizen *et al.* 1996), were found in the Kariega system. The concentration of the juveniles of these species in the REI region of a freshwater rich estuary, in conjunction with their scarcity or

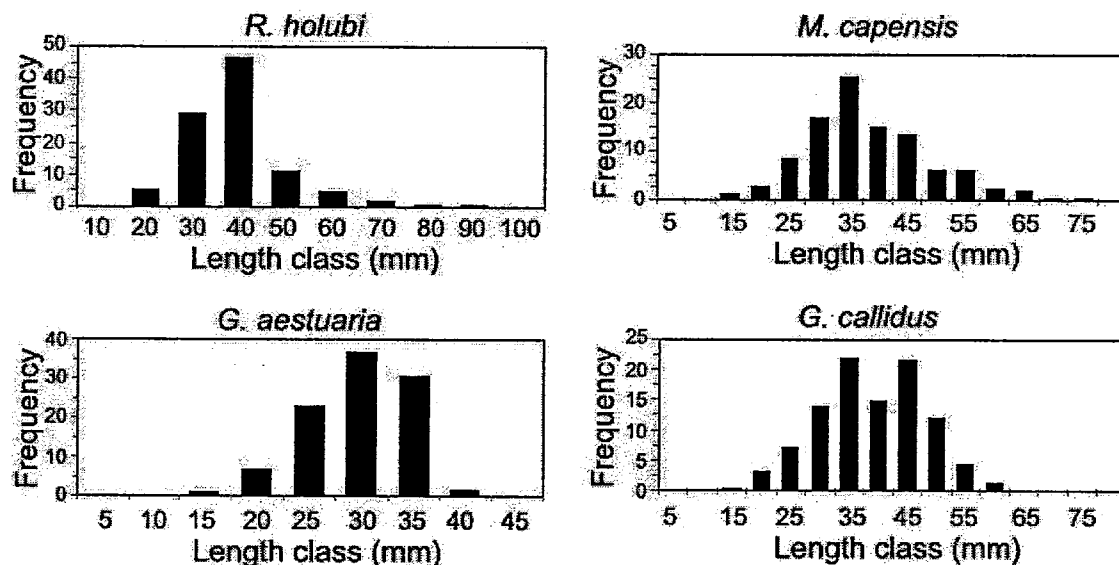


Figure 6.2.2: Percentage length (SL) frequency distributions of the dominant marine (*R. holubi* : maturity = 200 mm; *M. capensis* : maturity = 190 mm) and estuarine (*G. aestuaria* : maturity = 30 mm; *G. callidus* : maturity = 35 mm) species caught in the headwaters of the Kariega Estuary.

absence along the entire length of a freshwater deprived estuary, has major implications for fish stocks which utilise the REI region of estuaries as nursery areas. While further research is needed, it may be postulated that if freshwater is abstracted from a system and it results in a reduction of the REI region, there will be a concomitant reduction in suitable nursery areas available to species such as *P. commersonnii* and *A. japonicus*.

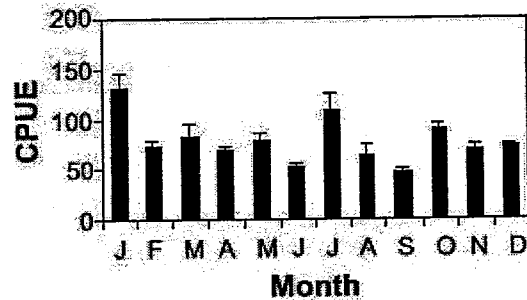


Figure 6.2.3: Mean CPUE (+1 S.E.) of the fishes sampled in the headwaters of the Kariega Estuary from January to December 1999.

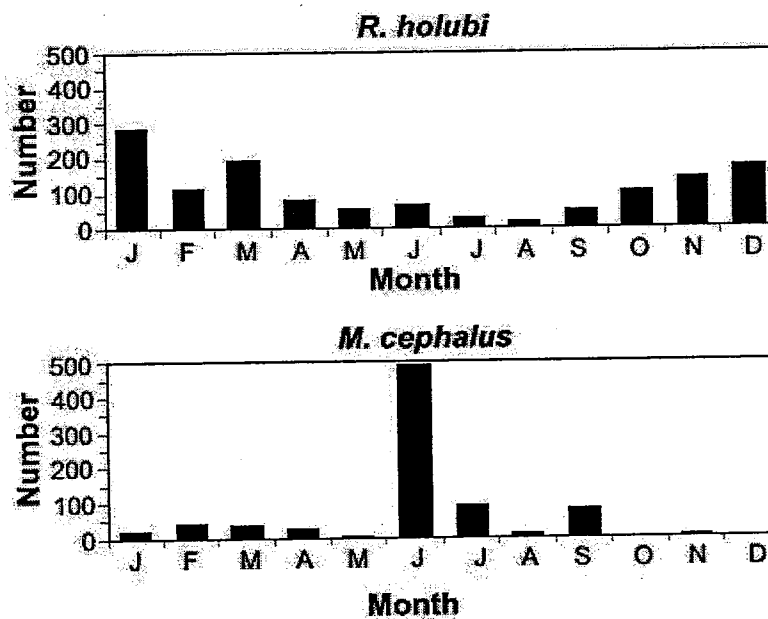


Figure 6.2.4: Monthly abundance values of *R. holubi* and *M. cephalus* caught in the headwaters of the Kariega Estuary from January to December 1999.

Table 6.2.1: Species composition, CPUE, estuarine-association categories and frequency of occurrence of fishes caught in the headwaters of the Kariaga Estuary from January to December 1999).

Species	Association category	Total (n)	Average CPUE	SD	Percentage composition	Frequency of occurrence
<i>G. callidus</i>	Ib	2314	19.3	26.5	24.7	100
<i>G. aestuaria</i>	Ia	2226	18.6	40.1	23.7	100
<i>M. capensis</i>	Vb	1518	12.7	26.0	16.2	100
<i>R. holubi</i>	IIa	1313	10.9	15.7	14.0	100
<i>M. cephalus</i>	IIa	784	6.5	41.6	8.4	83.3
<i>M. falciformis</i>	IIa	644	5.4	14.1	6.8	83.3
<i>A. breviceps</i>	Ib	405	3.4	16.7	4.3	75
Mugilidae (< 20mm)	-	78	0.7	2.7	0.8	33.3
<i>O. mossambicus</i>	IV	34	0.3	1.1	0.4	33.3
<i>G. acinaces</i>	IIb	18	0.2	1.0	0.2	33.3
<i>B. pallidus</i>	IV	10	0.1	0.4	0.1	50
<i>C. gilchristi</i>	Ib	6	0.1	0.3	<0.1	33.3
<i>L. tricuspidens</i>	IIb	4	<0.1	0.3	<0.1	8.3
<i>S. bleekeri</i>	IIb	2	<0.1	0.1	<0.1	16.6
<i>S. acus</i>	Ib	1	<0.1	0.1	<0.1	8.3
<i>C. nudiceps</i>	Ib	1	<0.1	0.1	<0.1	8.3
<i>P. commersonii</i>	IIa	1	<0.1	0.1	<0.1	8.3
<i>A. gymnocephalus</i>	Ib	1	<0.1	0.1	<0.1	8.3
<i>P. knysnaensis</i>	Ib	1	<0.1	0.1	<0.1	8.3
<i>L. lithognathus</i>	IIa	1	<0.1	0.1	<0.1	8.3
Total		9362				

The extent of estuarine association of the fish assemblages in the various reaches exhibited some clear trends (Figure 6.2.6). The estuarine species (Category I) were more prevalent in the middle reaches, while the percentage of marine species (Category II) decreased from the mouth towards the headwaters and then increased slightly in the REI. Very few marine stragglers (Category III) and freshwater fish (Category IV) were captured, with the marine stragglers being restricted to the mouth and the freshwater fish to the REI region. The catadromous species (Category V) were found predominantly in the headwaters and REI region.

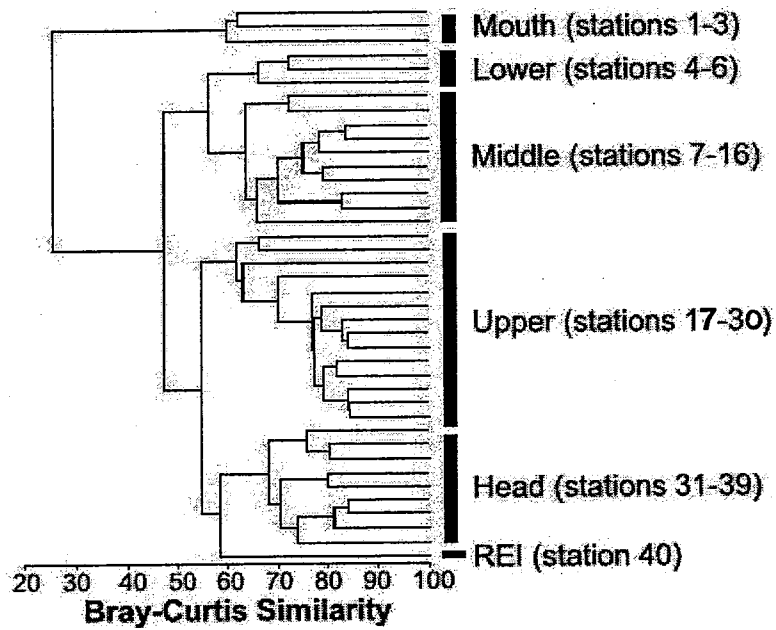


Figure 6.2.5: Dendrogram showing classification of the 40 seining sites in the Kariaga Estuary.

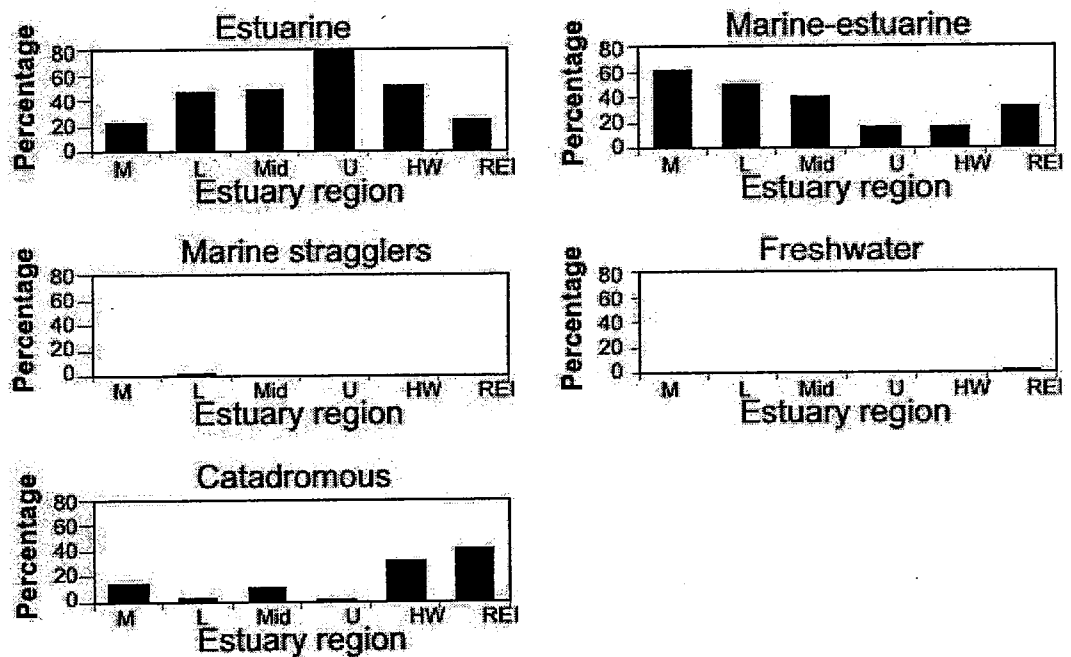


Figure 6.2.6: Percentage composition of the different estuarine-association categories in the Kariaga Estuary.

The species richness of the fish assemblages varied between the six reaches, with the highest richness being found in the mouth, lower and middle reaches (Table 6.2.3). This trend of decreasing richness and diversity in an upstream direction is similar to that found in a previous study on the Kariëga eelgrass beds (Ter Morshuizen and Whitfield 1994) as well as a study on the Elbe Estuary in Germany (Thiel *et al.* 1995). In contrast, the Great Fish system had a high ichthyofaunal diversity in the river and lower diversities towards the mouth (Ter Morshuizen *et al.* 1996).

Many of the marine stragglers (e.g. *Diplodus cervinus hottentotus* and *Pomadasys olivaceum*) that are found in the mouth and lower reaches of the Kariëga Estuary are restricted to that region even though full seawater conditions extend for another 10 km upstream. The restriction of these marine stragglers and absence of other common marine inshore species (e.g. *Pachymetopon grande*, *Oplegnathus conwayi* and *Chirodactylus brachydactylus*) indicates that while the Kariëga often has full seawater for over 80 % of its length, it is not an arm of the sea, but rather functions as a freshwater deprived estuary.

The bio-environmental analysis, which correlates the results of the ichthyofauna classification (Figure 6.2.5) to a range of abiotic and biotic variables, indicated that the best correlation for individual variables was temperature ($r = 0.57$) and percentage vegetation cover ($r = 0.35$). Salinity, oxygen concentration and turbidity were all poorly correlated to the ichthyofauna groupings in the Kariëga Estuary. In contrast, salinity and turbidity were found to be the best individual correlates with the fish assemblages in the freshwater enhanced Great Fish River Estuary (Ter Morshuizen *et al.* 1996). Turbidity is known to attract certain fish species (Cyrus and Blaber 1987a,b) but the lack of freshwater input into the Kariëga Estuary has resulted in low turbidities (< 20.0 NTU) throughout the system. Low oxygen concentrations have been shown to influence fish distribution in some KwaZulu-Natal estuaries (Begg 1984; Blaber *et al.* 1984), but the oxygen saturation in the Kariëga Estuary is consistently high (> 90 % saturation). The consistently clear and well oxygenated nature of the water in the littoral reaches of the Kariëga Estuary exclude these factors as structuring forces influencing fish distribution.

The best combination of variables accounting for changes in fish assemblage structure was temperature, salinity and percentage vegetation cover ($r = 0.62$). Salinity is generally regarded as one of the major abiotic forces involved in structuring fish communities in estuaries, e.g. fish assemblages in the Elbe Estuary were found to be predominantly influenced by salinity and temperature with day time and tidal cycle also being important (Thiel *et al.* 1995). Similarly, salinity and temperature were the dominant environmental influences on the fish assemblage of the Humber Estuary (Marshall and Elliott 1998). In contrast, salinity in the Kariëga Estuary cannot be a dominant structuring force, as there were distinct fish communities associated with the mouth, lower and middle reaches where there was no longitudinal salinity gradient. An earlier study on the fishes associated with *Z. capensis* beds in the Kariëga Estuary (Ter Morshuizen and Whitfield 1994), established that distributional patterns of most fish species associated with *Z. capensis* were similar to other estuaries in the Eastern Cape with normal axial salinity gradients, indicating that salinity was not an important ichthyofaunal structuring force in this system.

Table 6.2.2: Seine net catch per unit effort and percentage composition of fishes sampled in the Kariega Estuary during this study.

Species	Mouth (Sites 1 to 3)			Lower (Sites 4 to 6)			Middle (Sites 7 to 16)			Upper (Sites 17 to 30)			Headwater (Sites 31 to 39)			REI (Site 40)	
	CPUE	%		CPUE	%		CPUE	%		CPUE	%		CPUE	%		CPUE	%
<i>L. dumerilii</i>	35.8	57.7	-	-	-	-	0.2	0.3	0.3	1	1	-	-	-	-	-	-
<i>M. cephalus</i>	8.8	13.9	1.6	1.9	1.9	6.2	6.2	9.7	9.7	0.4	0.4	16.2	16.2	16.9	16.9	3.7	3.1
<i>G. aestuaria</i>	5.5	8.8	9.3	10.9	10.9	3.1	3.1	4.9	4.9	5.7	5.6	25.9	25.9	26.8	26.8	2.5	2
<i>A. breviceps</i>	4	6.6	5.3	6.2	6.2	9	9	14	14	16	15.8	2	2	2	2	-	-
<i>P. knysnaensis</i>	3.8	6.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>D. sargus</i>	1.8	2.8	3.3	3.9	3.9	0.2	0.2	0.4	0.4	-	-	-	-	-	-	-	-
<i>R. globiceps</i>	1	1.6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>S. salpa</i>	0.7	1.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>P. olivaceum</i>	0.4	0.6	-	-	-	<0.1	<0.1	<0.1	<0.1	-	-	-	-	-	-	-	-
Mugilidae (<20mm)	0.3	0.5	-	-	-	0.3	0.3	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.4	1.5	1.3
<i>R. holubi</i>	-	-	38.3	44.9	44.9	25.2	25.2	39.2	39.2	15.8	15.6	14	14	14.5	14.5	-	-
<i>C. superciliosus</i>	-	-	21.7	25.6	25.6	3	3	4.7	4.7	-	-	-	-	-	-	-	-
<i>C. nudiceps</i>	-	-	2.9	3.4	3.4	0.7	0.7	1.1	1.1	<0.1	<0.1	-	-	-	-	-	-
<i>D. cervinus</i>	-	-	1	1.3	1.3	<0.1	<0.1	<0.1	<0.1	-	-	-	-	-	-	-	-
<i>S. acus</i>	-	-	0.8	1	1	0.3	0.3	0.5	0.5	-	-	-	-	-	-	-	-
<i>C. natalensis</i>	-	-	0.8	1	1	1.3	1.3	2.1	2.1	-	-	-	-	-	-	-	-
<i>G. callidus</i>	-	-	-	-	-	12.4	12.4	19.2	19.2	61.1	60.4	22	22	22.8	22.8	26.7	22.2
<i>C. gilchristi</i>	-	-	-	-	-	1.3	1.3	2	2	-	-	-	-	-	-	-	-
<i>M. capensis</i>	-	-	-	-	-	0.4	0.4	0.6	0.6	0.3	0.3	13.8	13.8	14.2	14.2	44.2	36.8
<i>S. bleekeri</i>	-	-	-	-	-	<0.1	<0.1	<0.1	<0.1	-	-	-	-	-	-	-	-
<i>S. sutor</i>	-	-	-	-	-	<0.1	<0.1	<0.1	<0.1	-	-	-	-	-	-	-	-
<i>M. falciformis</i>	-	-	-	-	-	<0.1	<0.1	<0.1	<0.1	-	-	1.7	1.7	1.7	1.7	38.7	32.2
<i>A. immaculatus</i>	-	-	-	-	-	<0.1	<0.1	<0.1	<0.1	-	-	-	-	-	-	-	-
<i>P. commersonnii</i>	-	-	-	-	-	-	-	-	-	<0.1	<0.1	-	-	-	-	-	-
<i>H. capensis</i>	-	-	-	-	-	-	-	-	-	<0.1	<0.1	-	-	-	-	-	-
<i>O. mossambicus</i>	-	-	-	-	-	-	-	-	-	-	-	0.3	0.3	0.4	0.4	1.7	1.5
Total	62	100	85	100	100	64	64	100	100	101	100	96	96	100	100	119	100

Table 6.2.3: Species number and richness of the different fish assemblages identified at the 60% similarity level (cf. Figure 6.2.6).

	Mouth (Sites 1 to 3)	Lower (Sites 4 to 6)	Middle (Sites 7 to 16)	Upper (Sites 17 to 30)	Head-water (Sites 31 to 39)	REI (Site 40)
Average species number	9.0	12.3	11.8	6.6	7.4	8.0
Species richness (Margalef index)	2.2	2.6	2.6	1.3	1.4	1.4

The bio-environmental analysis indicated that percentage vegetation cover may be an important environmental factor. This result is supported by other studies, which have shown that eelgrass influences fish abundance and community composition in estuaries (Branch and Grindley 1979; Beckley 1983). Submerged aquatic vegetation provides shade, reduced water currents and increased surface area, while the rhizome mat may provide further habitat complexity (Bell and Pollard 1989; Orth 1992). These attributes, together with high primary productivity, are presumed to provide an abundant and varied supply of food, as well as protection from predators.

While a combination of temperature, percentage vegetation cover and salinity best correlates with the various fish assemblages, this correlation is not exceptionally high, indicating that other factors may well be important. Marshall and Elliott's (1998) study on the Humber Estuary also found that the range of variables measured only partly explained the variance displayed by the fish assemblages, and they concluded that other factors such as predator prey interactions, as well as prey distribution may also affect distribution patterns. From the above it becomes apparent that although salinity has been shown to be an important structuring force influencing fish assemblages in many estuaries, it is not the primary factor governing the distribution of fish in a freshwater deprived and marine dominated system such as the Kariega.

6.3 Freshwater flow regime and fish assemblages in the Great Fish REI region

6.3.1 Summary

Two and a half years of data were collected from the lower Great Fish River, head and estuary regions to determine the fish species composition within these areas. *Gilchristella aestuaria*, *Liza dumerilii*, *Rhabdosargus holubi* and *Pomadasys commersonnii* were the four most abundant species captured, with riverine flow rate having an important effect on both species composition and numbers of fishes in the different regions. Most marine species displayed a strong inverse relationship between catch per unit effort and elevated freshwater inputs. Euryhaline marine species dominated the catches at all sampling sites during low flows but were less common during high flow periods when catadromous species were most abundant. Based on the available evidence it is suggested that for most marine species in the river this decline in abundance is related to reduced conductivity levels following floods rather than avoidance of elevated flows. The impact of elevated suspended concentrations and lowered dissolved oxygen concentrations on freshwater and estuarine fish populations during major river flooding is also discussed.

6.3.2 Introduction

Despite the large amount of ichthyological research that has been conducted in South African estuaries (Potter *et al.* 1990; Whitfield 1994) there is a paucity of information on fish assemblages at the river-estuary interface (REI). Bok (1983) recorded 11 fish species in the head region of the Kowie system and Rayner (1993) found 15 taxa at an ebb and flow site on the Keiskamma Estuary. A recent study (Ter Morshuizen *et al.* 1996) provided the first detailed investigation into the species composition and distribution of fishes in the head region of a South African estuary.

A change in riverine flow rate is accompanied by altered water volumes and velocities, and sometimes by fluctuating levels of nutrients, suspended particulate matter, dissolved organic matter, temperature, conductivity, dissolved oxygen and turbidity (Drinkwater and Frank 1994). Therefore, major alterations in freshwater flow regime would be expected to cause changes in both the riverine and estuarine physico-chemical environments. In this section, the possible influence of river flow volumes on fishes in the lower Great Fish system, with particular emphasis on conductivity level alterations associated with different flow regimes, is investigated.

6.3.3 Study area

The Great Fish River (33° 28'S and 27° 10'E) enters the Indian Ocean about half way between Port Elizabeth and East London. Prior to 1975 the river was known to have had a highly variable flow regime. Periods of zero surface flow frequently occurred, causing the river to form a series of discrete pools. When these conditions persisted long enough, closure of the estuary mouth resulted (Reddering and Esterhuysen 1982). However, in 1975 the erratic flow of this system was stabilised by

the provision of water from the Orange River via an 85 km long tunnel. The geology of the catchment includes the highly erodible Beaufort and Ecca groups (Reddering and Esterhuysen 1982) resulting in considerable amounts of salts being leached from ancient marine sediments by runoff (O'Keeffe and de Moor 1988). Consequently the salinity levels in the river can be as high as 2000.0 mg l⁻¹ but these are diluted by water from the Orange River (132.0 mg l⁻¹) (O'Keeffe and de Moor 1988). In addition, the river is extremely turbid (average 200.0 NTU) due to the highly erodible soils present in the catchment area (Laurenson 1984).

In 1980 the tidal head was about 11 km from the Great Fish mouth and the furthest extent of estuarine water was only 5 to 7 km from the mouth (Reddering and Esterhuysen 1982). For the period from 1992 to 1995 the tidal head was approximately 15 km from the mouth and the furthest extent of saline water (1.0 ‰) at spring high tide ranged from 3 to 10 km from the mouth, varying according to river flow rate.

6.3.4 Materials and methods

Bi-monthly sampling on the new-moon low tide occurred from November 1992 until January 1995, providing 14 sampling occasions that spanned three summers and two winters. Between November 1992 and July 1994 eight samples were collected per trip, increasing to 17 samples for each of the last three trips. Sampling was conducted using a 30 m x 2 m seine net with a 10 mm bar mesh size, fitted with a bag of 5 mm bar mesh size. The net swept an estimated 600 m² at each site, and was operated in water no deeper than 1.5 m. With the exception of the sandy bottom at site 14 (Figure 6.3.1) all the sites had a predominantly muddy substratum, with rocks and submerged macrophytes being absent. Each site was seine netted once on each sampling occasion and fish samples were immediately fixed in 10% formalin.

In the laboratory, fish were identified, counted and preserved in 60% isopropyl alcohol. The grouping of species according to their life-history styles follows the categories outlined by Whitfield (1994). Mugilids < 30.0 mm SL were not identified beyond family level. All specimens that could not be identified to species level have been omitted from the life-history analyses as their categories are uncertain. Catch per unit effort values are based upon the average number of fish captured per seine net haul.

Species richness (R) (Margalef 1958) and species diversity (H) (Krebs 1978) indices were calculated for the fish assemblages in each of the three regions using confirmed species only. The behaviour of the former index is determined primarily by the number of species recorded in relation to the total number of fish captured, whereas the latter index reflects the relative abundance of individual species within the overall community.

A water sample was collected from midwater at each site. Temperature was measured on site by means of a mercury thermometer, and salinity and turbidity were determined in the laboratory using a Reichert Model 10419 optical salinometer and Hach Model 2100A turbidimeter respectively. Temperature, turbidity and salinity were compared between sites using the Scheffe one-way analysis of variance (ANOVA) test.

River flow volumes were calculated from unpublished information provided by the Department of Water Affairs and Forestry for Station Q9H018. For the sake of comparison, monthly flow was divided into low ($< 10 \times 10^6 \text{ m}^3$), medium ($10 \text{ to } 20 \times 10^6 \text{ m}^3$) and high volumes ($> 20 \times 10^6 \text{ m}^3$). The Spearman rank correlation and Pearson's correlation coefficient were used to determine the relationship between fish abundance and river flow volume. The electrical conductivity (mS m^{-1} at 25.0°C) of the river water under different flow regimes (Figure 6.3.2) was obtained from unpublished Department of Water Affairs and Forestry measurements made at Station Q9M18.

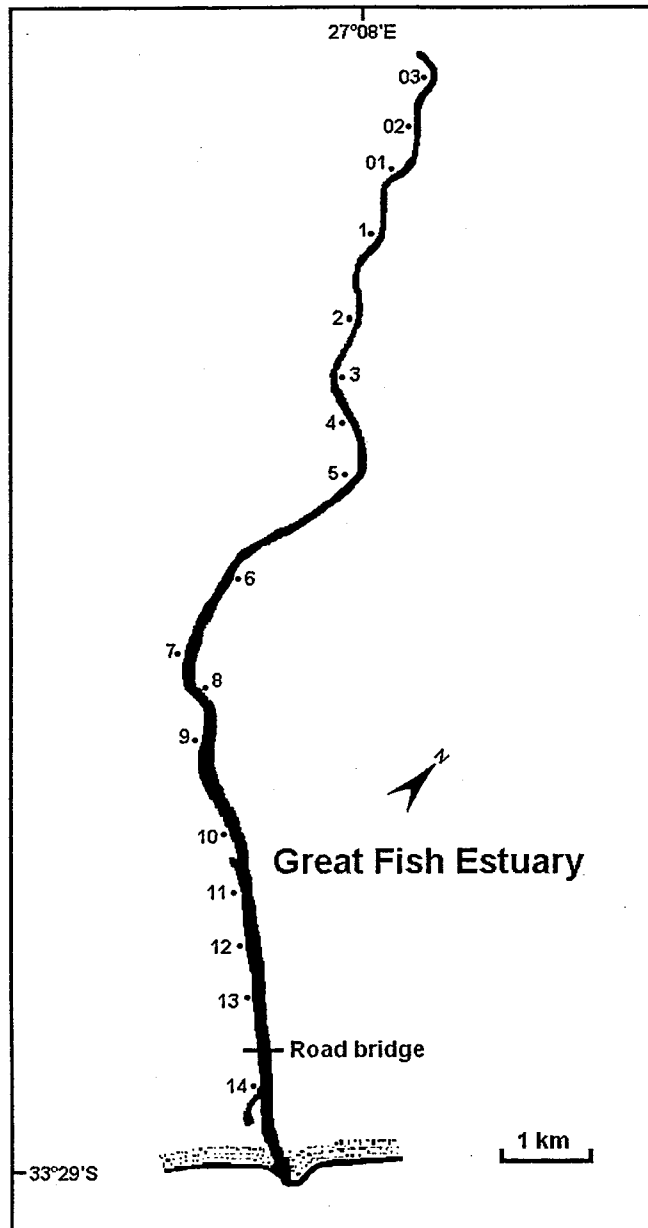


Figure 6.3.1: Map of the Great Fish River system showing the seine net sampling sites.

6.3.5 Results and discussion

6.3.5.1 Physical parameters

Estuarine researchers have regarded levels of > 80.0 NTU as very turbid (Cyrus 1988). Therefore turbidities recorded in the Great Fish River and upper estuary, with means in the vicinity of 200 NTU, can only be described as highly turbid. Temperature varied between a minimum of 12.0°C in winter and a maximum of 29.0°C in summer. The ANOVA indicated a lack of significant differences between sites 1 to 8 for either turbidity or temperature. Salinity, however, showed a significant difference ($P < 0.01$) between sites and was therefore used as the parameter for differentiating between the regions. Measurements of < 1.0 ‰ were regarded as riverine, from 1.0 to 4.0 ‰ as the head of the estuary and > 4.0 ‰ as estuarine.

Salinity, rather than physical locality, was used to facilitate comparisons between fish assemblages captured during periods of differing river flow. Under river flood conditions the 'estuary' was reduced in size, as the increased volume of freshwater discharged was more substantial than the incoming tide. Conversely, the estuary expanded upstream during low flow periods. Sites 03 to 3 were always riverine, sites 11 to 14 were always estuarine and sites 4 to 10 varied according to the river flow rate at the time of sampling.

No correlation was evident between either salinity and temperature or turbidity and temperature. A strongly negative trend existed between conductivity and monthly river flow volume (Figure 6.3.2).

6.3.5.2 River flow volumes and fish response

Linking fish catches to Great Fish River volumes for the month prior to each sampling trip revealed distinct trends between flow rate and the species assemblage within the sampling area. Both Spearman's and Pearson's tests indicated a weak negative correlation between overall fish abundance and the riverine flow rate. Fish abundance in both the river and head regions decreased with increasing riverine flow, whereas the highest CPUE of fishes in the estuary were during medium flow conditions (Figure 6.3.3). Under low flow regimes the greatest abundance of fishes was recorded in the head region (Figure 6.3.3). With an increase in freshwater input the majority of fish species, especially the marine taxa, become scarce in the river (Figure 6.3.4). A major increase in river flow results in an overall decline in fish abundance throughout the system (Figure 6.3.3) with river flooding sometimes causing extensive fish mortalities in the estuary (Whitfield and Paterson 1995).

In the river and head regions, species richness (R) was almost identical during the different flow regimes. However, the estuary region R value was highest during periods of low flow (Figure 6.3.5) when large numbers of euryhaline marine species were present in that part of the system (Table 6.3.1). Under low flow conditions species diversity (H) was similar in all three regions, with the highest value recorded for the estuary. During periods of high flow, diversity (H) was greatest in the river due to the decline in abundance of marine and estuarine dominants in this region (Figure 6.3.5).

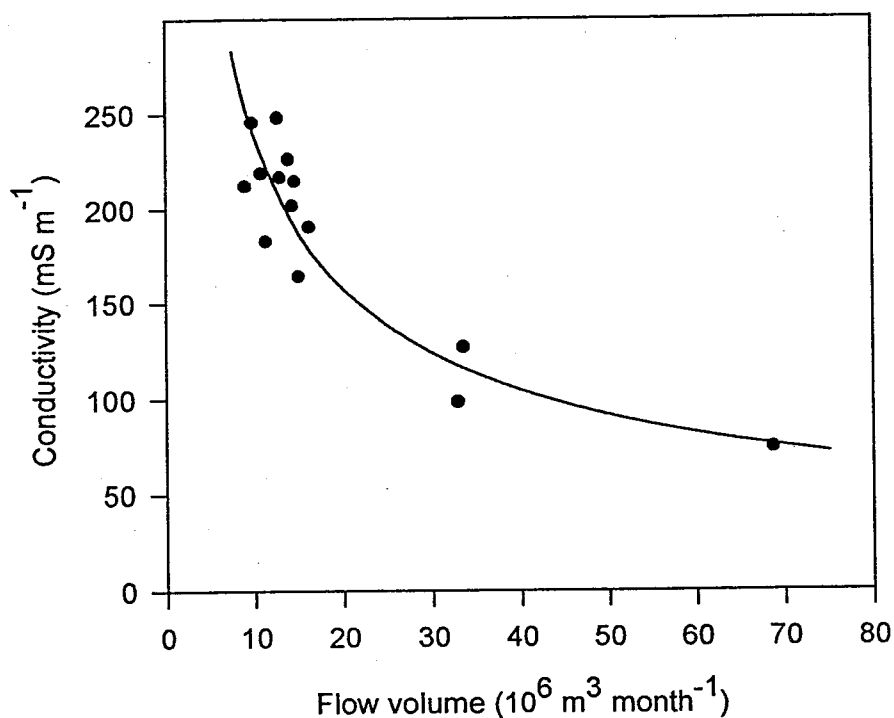


Figure 6.3.2: Regression curve ($y = 936.7 x^{-0.595}$, $r = 0.94$) for monthly river flow volume versus conductivity in the Great Fish River system (data supplied by the Department of Water Affairs and Forestry).

The marine species contribution to the catch in the river and estuary decreased from > 60.0% under low flow conditions to < 30.0 % during periods of elevated riverine flow (Figure 6.3.6). Marine stragglers were absent from the river, and were only present in small numbers in the estuary (< 0.1% of CPUE) during low or medium river flow conditions (Table 6.3.1). The estuarine species remained common under all flow regimes, whereas the catadromous and most of the freshwater species were more common during the elevated flow periods (Figure 6.3.6).

The translocated Orange River water has a major diluting effect on the Great Fish River water, with the latter having extremely high conductivity levels prior to 1975 (O'Keeffe 1989). Despite this dilution, conductivity levels in the system remain high (up to 625.0 mS m^{-1}), except during river flooding (Figure 6.3.2). It is therefore likely that the abundance of marine and estuarine fishes in the river is closely related to conductivity, with declining fish abundance associated with reduced conductivity levels. Since most estuarine-associated fish species can tolerate freshwater conditions for short periods, the duration of flooding events has a major influence on the ichthyofauna (Marais 1982). However, osmotic stress is not the only factor influencing estuarine fish survival during floods. For example, the extremely high suspensoid levels together with lowered dissolved oxygen concentrations during flash flooding in the Great Fish Estuary may have been responsible for the January 1995 fish kill in this system (Whitfield and Paterson 1995). Since the definition of the head region and estuary was based on salinity and not geography, conductivity cannot be a factor affecting the abundance of fishes in these saline regions.

Table 6.3.1: Mean number of fish per seine haul (CPUE) for each species from the river (R), head (H) and estuary (E) of the Great Fish system under low, medium and high river flows.

Flow regime	Low			Medium			High		
Mean flow (10 ⁶ m ³ .month ⁻¹)	9.4			13.5			45.0		
Region/Taxa	R	H	E	R	H	E	R	H	E
Estuarine									
<i>Ambassis gymnocephalus</i>	1.3	17.7	0.2	0.8	4.9	43.5	2.3	0.8	
<i>Atherine breviceps</i>		1.7						0.3	0.3
<i>Caffrogobius gilchristi</i>			0.8	0.1	0.1	1.0		0.3	
<i>C. nudiceps</i>			0.6		0.2	3.8			
<i>Eleotris fusca</i>							0.1		
<i>Gilchristella aestuaria</i>	87.8	493.3	98.4	48.1	5.2	133.6	4.8	11.3	20.8
<i>Glossogobius callidus</i>	0.2	0.3	0.2	1.6	0.4	0.2	1.0	1.8	
<i>Oligolepis keiensis</i>			0.4						
<i>Psammogobius knysnaensis</i>	0.1	0.3	1.8	2.3	4.5	8.2			
Marine									
<i>Acanthopagrus berda</i>	0.9		0.6	2.2	1.4	0.4	0.4	0.5	
<i>Argyrosomus japonicus</i>	5.4	19.7	2.4	7.6	7.4	2.3	0.6		0.3
<i>Caranx sexfasciatus</i>	0.3	6.3	0.4						
<i>Crenimugil crenilabis</i>			0.6	0.1	0.2				
<i>Elops machnata</i>	8.3	36.0	0.6	0.6	1.2	0.1	1.5		
<i>Galeichthys feliceps</i>	0.2	0.7		0.8	3.2	0.1			0.3
<i>Gerres acinaces</i>	0.1								
<i>Heteromycteris capensis</i>	0.1		1.0	0.5	2.8	10.8			
<i>Leiognathus equula</i>					0.1				
<i>Lichia amia</i>	0.1	0.7	0.6			0.1			
<i>Lithognathus lithognathus</i>	0.1		9.4	1.1	1.1	12.5		0.5	
<i>Liza dumerilii</i>	146.3	48.0	206.4	28.6	64.1	41.4	6.0	8.8	8.3
<i>L. richardsonii</i>	0.1		48.8	2.3	0.3	36.6			4.8
<i>L. tricuspidens</i>	0.4		0.4			0.1			
<i>Monodactylus falciformis</i>	1.1	2.0	0.4	1.6	0.1	0.3	0.6		0.3
<i>Platycephalus indicus</i>					0.2	0.2			
<i>Pomadasys commersonnii</i>	13.5	74.7	13.8	14.6	62.1	19.4	4.4	28.0	1.5
<i>Rhabdosargus holubi</i>	4.2	69.7	65.0	7.5	56.2	188.8	0.1	5.5	2.8
<i>Solea bleekeri</i>	2.5	2.7	0.2	2.5	4.1	2.4	0.7	1.3	
<i>Terapon jarbua</i>				0.1	0.3				
Marine stragglers									
<i>Amblyrhynchotes honckenii</i>			0.2			0.1			
<i>Polydactylus plebius</i>					0.2				
<i>Pomadasys olivaceum</i>						0.1			
Freshwater									
<i>Cyprinus carpio</i>	1.6	2.7	0.2	1.4	3.2		0.8		
<i>Clarias gariepinus</i>	0.2			0.3					
<i>Oreochromis mossambicus</i>	4.7			4.8	0.5	0.1	8.2		
<i>Labeo umbratus</i>	0.1			0.1			3.2	2.0	7.0
Catadromous (*A proportion of the mugilids <30 mm SL belong to the marine group)									
<i>Mugil cephalus</i>	12.4	2.0	4.8	2.9	7.1	15.4	13.2	20.8	20.8
<i>Myxus capensis</i>	0.1			2.7	0.2	8.1	10.4		0.5
Mugilidae (< 30 mm SL)*	83.9	1.0	32.2	61.0	49.8	43.9	4.1	15.3	30.5
Mean CPUE	375.9	779.5	490.6	196.2	281.1	573.5	62.4	97.2	98.2

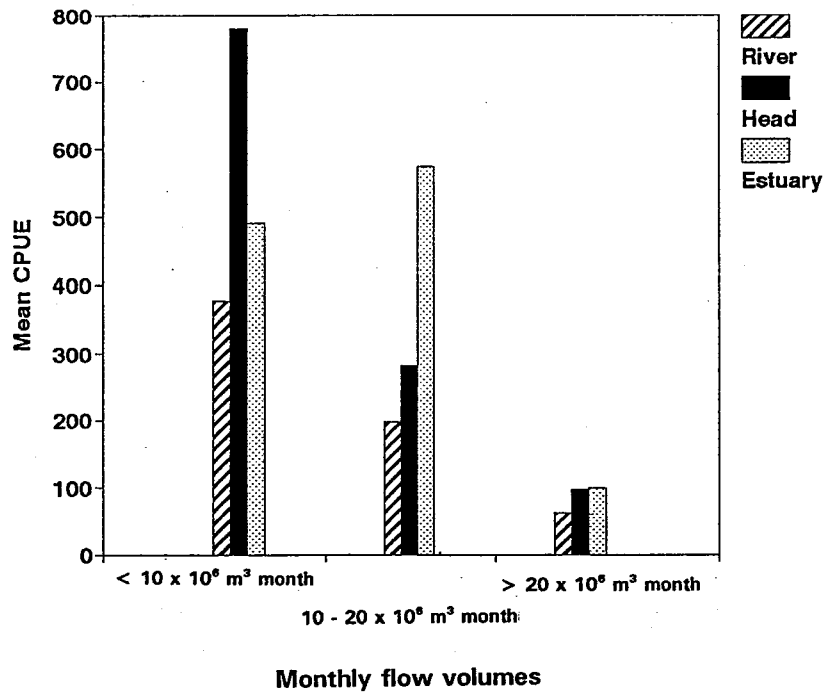


Figure 6.3.3 Fish catch per unit effort (CPUE) in the three regions of the Great Fish River system under conditions of low ($< 10 \times 10^6 \text{ m}^3 \text{ month}^{-1}$), medium ($10 \text{ to } 20 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) and high ($> 20 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) riverine flow.

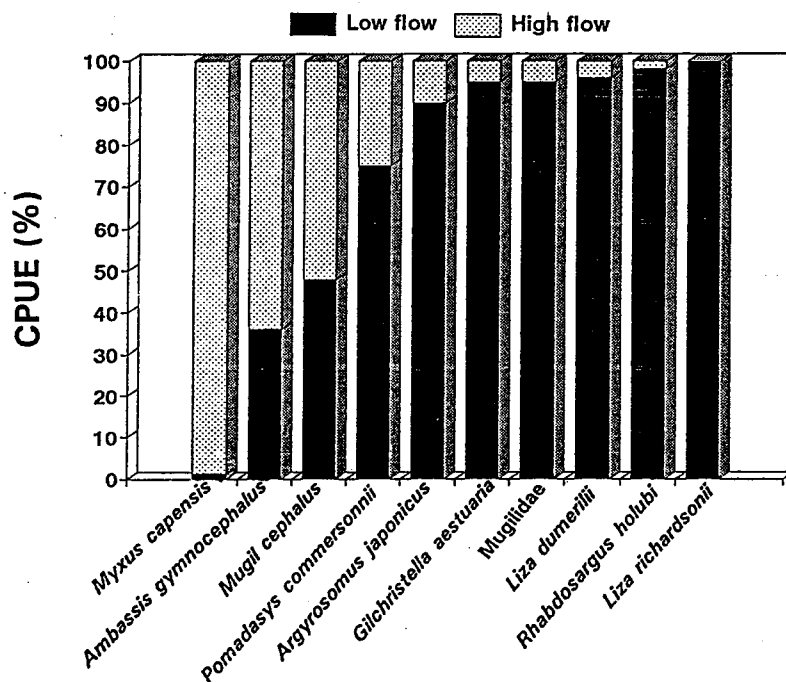


Figure 6.3.4: Relative catch per unit effort (CPUE) of the most common fish species captured in the Great Fish River system under conditions of low ($< 10 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) and high ($> 20 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) riverine flow.

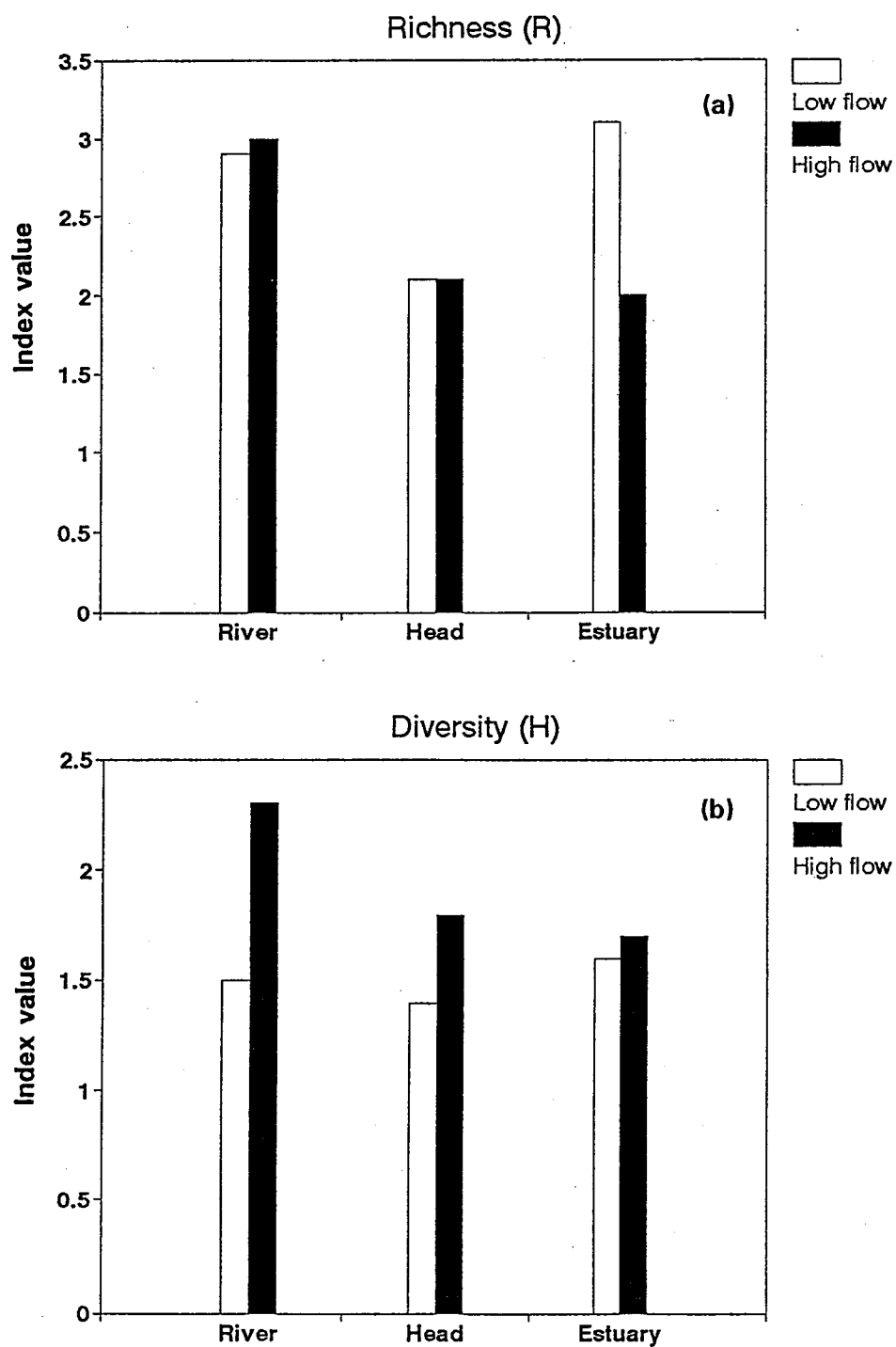


Figure 6.3.5: Fish species richness (a) and diversity (b) indices for three regions of the Great Fish River system under conditions of low ($< 10 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) and high ($> 20 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) riverine flow.

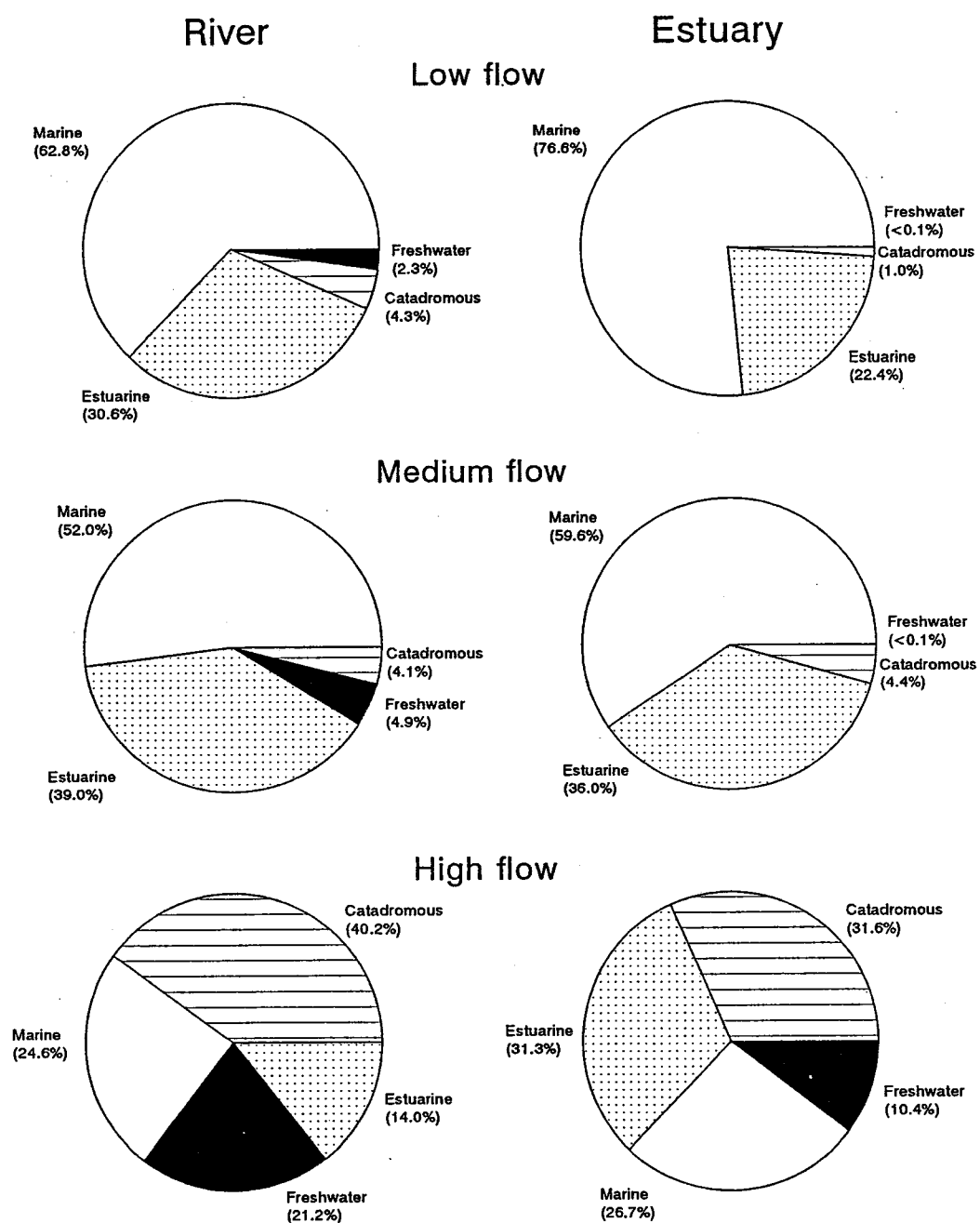


Figure 6.3.6: Life-history composition of fish assemblages in the Great Fish River and Estuary under conditions of low ($< 10 \times 10^6 \text{ m}^3 \text{ month}^{-1}$), medium ($10 \text{ to } 20 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) and high ($> 20 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) riverine flow. Marine stragglers always comprised $< 0.1\%$ of the fish assemblage and are therefore excluded from the analysis.

In a review of fish 'kills' in South African estuaries, Whitfield (1995) indicated that a combination of factors usually cause these mass mortalities. The above synthesis, together with the results from the present study, point to river flooding causing a decline in the abundance of most marine fish species in Eastern Cape systems, with changes in conductivity, dissolved oxygen levels, silt loads, rate of freshwater input and duration of the increased flow all playing a possible role.

An inverse relationship is evident between river flow volume and conductivity (Figure 6.3.2). During periods of low flow, salts leaching into the river cause an increase in conductivity, whereas high flow rates dilute this effect. Since sodium and chloride are two of the most prevalent elements in the Great Fish River, they also elicit the strongest response to dilution (Anonymous 1995). It is probable that the reduction in their concentrations have an effect on the ability of euryhaline marine fishes to utilise freshwater areas of the system. Under low and medium flows the conductivity of the river water remains high (Figure 6.3.2), thus indicating the presence of significant amounts of salts. It is also under these conditions that euryhaline marine species dominate the catches (> 60.0%) in all three of the sampled regions (Table 6.3.1). During and immediately after river flood conditions the salts are diluted, conductivity decreases and the euryhaline marine species are no longer abundant in riverine samples (Figure 6.3.6). There is therefore an indirect link between river flow and marine fish distribution and abundance in the Great Fish River system.

6.4 Acknowledgements

We thank Professor S. Radloff of the Department of Statistical Sciences at Rhodes University for statistical advice and the Department of Water Affairs and Forestry for use of unpublished river flow and water conductivity data. This study was funded by the Water Research Commission and the National Research Foundation.

6.5 References

- Adams, J.B. and Talbot, M.M. 1992. The influence of river impoundment on the estuarine seagrass *Zostera capensis* Setchell. *Botanica Marina* 35: 69-75.
- Allanson, B.R. and Read, G.H.L.. 1995. Further comment on the response of Eastern Cape Province estuaries to variable freshwater inflows. *South African Journal of Aquatic Science* 21(1/2): 56-70.
- Anonymous 1995. Rivers of southern Africa: The Great Fish River. *African Wildlife* 49(5): 15.
- Beckley, L.E. 1983. The ichthyofauna associated with *Zostera capensis* Setchell in the Swartkops Estuary, South Africa. *South African Journal of Zoology* 18: 15-24.
- Beckley, L.E. 1984. The ichthyofauna of the Sundays Estuary, South Africa, with particular reference to the juvenile marine component. *Estuaries* 7: 248-258.
- Begg, G.W. 1984. The comparative ecology of Natal's smaller estuaries. Natal Town and Regional Planning Report 62: 182 pp.

- Bell, J.D. and Pollard, D.A. 1989. Ecology of fish assemblages and fisheries associated with seagrasses. In: Biology of seagrasses (ed. A.W.D. Larkum, A.J. McComb, and S.A. Shepard). Elsevier, Amsterdam: pp. 565-609.
- Bennett, B.A. 1988. The fish community of a moderately exposed beach on the southwestern Cape coast of South Africa and an assessment of this habitat as a nursery for juvenile fish. *Estuarine, Coastal and Shelf Science* 28: 293-305.
- Blaber, S.J.M. 1973. Temperature and salinity tolerance of juvenile *Rhabdosargus holubi* (Steindachner) (Teleostei: Sparidae). *Journal of Fish Biology* 5: 593-598.
- Blaber, S.J.M. 1978. Fishes of the Kosi system. *Lammergeyer* 24: 28-41.
- Blaber, S.J.M. and Blaber, T.G. 1980. Factors affecting the distribution of juvenile estuarine and inshore fish. *Journal of Fish Biology* 17: 143-162.
- Blaber, S.J.M., Cyrus, D.P. and Whitfield, A.K. 1981. The influence of zooplankton food resources on the morphology of the estuarine clupeid *Gilchristella aestuarius* (Gilchrist 1914). *Environmental Biology of Fishes* 6: 351-355.
- Blaber, S.J.M., Hay, D.G., Cyrus, D.P. and Martin, T.J. 1984. The ecology of two degraded estuaries on the north coast of Natal, South Africa. *South African Journal of Zoology* 19: 224-240.
- Bok A.H. 1983. The demography, breeding biology and management of the two mullet species (Pisces: Mugilidae) in the eastern Cape, South Africa. Ph.D. Thesis, Rhodes University, Grahamstown.
- Bok, A.H. 1984. Extensive culture of two species in freshwater impoundments in the eastern Cape. *South African Journal of Zoology* 19: 31-36.
- Branch, G.M. and Grindley, J.R. 1979. Ecology of southern African estuaries. Part XI. Mngazana: a mangrove estuary in Transkei. *South African Journal of Zoology* 14: 149-170.
- Clarke, K.R. and Warwick, R.M. 1994. Change in marine communities: an approach to statistical analysis and interpretation. Natural Environment Research Council, Plymouth: 144 pp.
- Cyrus, D.P. 1988. Turbidity and other physical factors in Natal estuarine systems. Part 1: Selected estuaries. *Journal of the Limnological Society of Southern Africa* 14: 60-71.
- Cyrus, D.P. and Blaber, S.J.M. 1987a. The influence of turbidity on juvenile marine fish in the estuaries of Natal, South Africa. *Continental Shelf Research* 7: 1411-1416.
- Cyrus, D.P. and Blaber, S.J.M. 1987b. The influence of turbidity on juvenile fishes in estuaries. Part 2. Laboratory studies, comparisons with field data and conclusions. *Journal of Experimental Biology and Ecology* 109: 71-91.
- Dallas, H.F. and Day, J.A. 1993. The effect of water quality variables on riverine ecosystems: a review. Water Research Commission Report No. TT 61/93.
- Drinkwater K.F. and Frank, K.T. 1994. Effects of river regulation and diversion on marine fish and invertebrates. *Aquatic Conservation Marine and Freshwater Ecosystems* 4: 135-151.
- Gardner, B.D., Connell, A.D., Eagle, G.A., Moldan, A.G.S. and Watling, R.J. 1985. South African Marine Pollution Survey Report 1979-1982. South African National Scientific Programmes Report No. 115.

- Grange, N. 1992. The influence of contrasting freshwater inflows on the feeding ecology and food resources of zooplankton in two Eastern Cape estuaries, South Africa. Ph.D. Thesis, Rhodes University, Grahamstown: 230 pp.
- Grange, N. and Allanson, B.R. 1995. The influence of freshwater inflow on the nature, amount and distribution of seston in estuaries in the eastern Cape, South Africa. *Estuarine, Coastal and Shelf Science* 40: 403-420.
- Grange, N., Whitfield, A.K., De Villiers, C.J. and Allanson, B.R. 2000. The response of two South African estuaries to altered river flow regimes. *Aquatic Conservation: Marine and Freshwater Ecosystems* 10: 155-177.
- Hanekom, N. and Baird, D. 1984. Fish community structures in *Zostera* and non-*Zostera* regions of the Kromme Estuary, St Francis Bay. *South African Journal of Zoology* 19: 295-301.
- 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: 95-107.
- Hodgson, A.N. 1987. Distribution and abundance of the macrobenthic fauna of the Kariega Estuary. *South African Journal of Zoology* 22: 153-162.
- Krebs C.J. 1978. Ecology: The experimental analyses of distribution and abundance. New York, Harper and Row Publishers, Inc.
- Laurenson L.J.B. 1984. Colonisation theory and invasive biota: the Great Fish River, a case history. M.Sc. Thesis, Rhodes University, Grahamstown.
- Marais, J.F.K. 1981. Seasonal abundance, distribution, and catch per unit effort using gill nets, of fishes in the Sundays estuary. *South African Journal of Zoology* 16: 144-150.
- Marais, J.F.K. 1982. The effects of river flooding on the fish populations of two eastern Cape estuaries. *South African Journal of Zoology* 17: 96-104.
- Marais, J.F.K. 1983a. Seasonal abundance, distribution and catch per unit effort of fishes in the Krom estuary, South Africa. *South African Journal of Zoology* 18: 96-102.
- Marais, J.F.K. 1983b. Fish abundance and distribution in the Gamtoos Estuary, with notes on the effect of floods. *South African Journal of Zoology* 18: 103-109.
- Marais, J.F.K. and Baird, D. 1980. Seasonal abundance, distribution and catch per unit effort of fishes in the Swartkops Estuary. *South African Journal of Zoology* 15: 66-71.
- Margalef R. 1958. Information theory in ecology. *General Systematics* 3: 36-71.
- Marshall, S. and Elliott, M. 1998. Environmental influences on the fish assemblage of the Humber Estuary, UK. *Estuarine, Coastal and Shelf Science* 46:175-184.
- Morin, B., Hudon, C. and Whoriskey, F.G. 1992. Environmental influences on seasonal distribution of coastal and estuarine fish assemblages at Wemindji, eastern James Bay. *Environmental Biology of Fishes* 35: 219-229.
- O'Keeffe, J.H. 1989. Conserving rivers in southern Africa. *Biological Conservation* 49: 255-274.
- O'Keeffe, J.H. and de Moor, F.C. 1988. Changes in the physico-chemistry and benthic invertebrates of the Great Fish River, South Africa, following an interbasin transfer of water. *Regulated Rivers Research and Management* 2(1): 39-55.

- Orth, R.J. 1992. A perspective on plant-animal interactions in seagrasses: physical and biological determinants influencing plant and animal abundance. In: Plant-animal interactions in the marine benthos (ed. D.M. John, S.J. Hawkins and J.H. Price). *Systematics Association, Special Volume* 46: 147-164.
- Paterson, A.W. and Whitfield, A.K. 2000. The ichthyofauna associated with an intertidal creek and adjacent eelgrass beds in the Kariega Estuary, South Africa. *Environmental Biology of Fishes* 58: 145-156.
- Plumstead, E.E., Prinsloo, J.F. and Schoonbee, H.J. 1989. A survey of the fish fauna of Transkei estuaries. Part Two. The Mbashe Estuary. *South African Journal of Zoology* 24: 273-281.
- Potter I.C., Beckley L.E., Whitfield A.K. and Lenanton, R.C.J. 1990. Comparisons between the roles played by estuaries in the life cycles of fishes in temperate Western Australia and South Africa. *Environmental Biology of Fishes* 28: 143-178.
- Rayner, J.L. 1993. A study on the ichthyofauna of the Kieskamma River ebb and flow region with emphasis on diversity and seasonality of their migrations B.Sc. Project, Rhodes University, Grahamstown.
- Reddering J.S.V. and Esterhuysen, K. 1982. Fluvial dominated sedimentation in the Great Fish estuary. Department of Geology, University of Port Elizabeth, ROSIE Report No. 4: 62 pp.
- 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.
- Rogers, S.G., Targett, T.E. and Van Sant, S.B. 1984. Fish-nursery use in Georgia salt-marsh estuaries: The influence of springtime freshwater conditions. *Transactions of the American Fisheries Society* 113: 595-606.
- Skelton, P.H. 1993. A complete guide to the freshwater fishes of southern Africa. Halfway House, Southern Book Publishers (Pty) Ltd.
- Talbot, M.M.J-F. 1982. Aspects of the ecology and biology of *Gilchristella aestuarius* (G & T) in the Swartkops Estuary. M.Sc. Thesis, University of Port Elizabeth, Port Elizabeth.
- Talbot, M.M.J-F. and Baird, D. 1985. Feeding of the estuarine round herring *Gilchristella aestuarius* (Stolephoridae). *Journal of Experimental Marine Biology and Ecology* 87: 199-214.
- Ter Morshuizen, L.D. 1994. A study of two Eastern Cape estuaries, the Great Fish River estuary and the Kariega River estuary, relating salinity regimes to freshwater inflow. B.Sc. Project, Rhodes University, Grahamstown.
- Ter Morshuizen, L.D. and Whitfield, A.K. 1994. The distribution of littoral fish associated with eelgrass *Zostera capensis* beds in the Kariega Estuary, a southern African system with a reversed salinity gradient. *South African Journal of Marine Science* 14: 95-105.
- Ter Morshuizen, L.D., Whitfield, A.K. and Paterson, A.W. 1996. Influence of freshwater flow regime on fish assemblages in the Great Fish River and estuary. *Southern African Journal of Aquatic Sciences* 22: 52-61.
- Thiel, R., Sepulveda, A., Kafemann, R. and Nellen, W. 1995. Environmental factors as forces structuring the fish community of the Elbe Estuary. *Journal of Fish Biology* 46:47-69.

- Wallace, J.H. 1975. The estuarine fishes of the east coast of South Africa. I. Species composition and length distribution in the estuarine and marine environments. II. Seasonal abundance and migrations. *Institute Investigational Report of the Oceanographic Research* 40: 1-72.
- Wallace, J.H., Kok, H.M., Beckley, L.E., Bennett, B., Blaber, S.J.M. and Whitfield, A.K. 1984. South African estuaries and their importance to fishes. *South African Journal of Science* 80: 203-207.
- Wallace, J.H. and van der Elst, R. 1975. The estuarine fishes of the east coast of South Africa. *Investigational Report of the Oceanographic Research Institute* 42: 1-63.
- Washington, H.G. 1984. Diversity, biotic and similarity indices. A review with special relevance to aquatic ecosystems. *Water Research* 18: 653-694.
- Watling, R.J. and Watling, H.R. 1983. Metal surveys in South African estuaries VII Bushmans, Kariëga, Kowie and Great Fish Rivers. *Water S.A.* 9: 66-70.
- Whitehead, P.J.P. and Wongratana, T. 1986. Family No. 54: Clupeidae. In: Smith's sea fishes (ed. M.M. Smith and P.C. Heemstra). CTP Book Printers, Cape Town: pp. 199-204.
- Whitfield, A.K. 1989. Ichthyoplankton in a southern African surf zone: nursery area for the postlarvae of estuarine associated fish species? *Estuarine, Coastal and Shelf Science* 29: 533-547.
- Whitfield, A.K. 1990. Life-history styles of fishes in South African estuaries. *Environmental Biology of Fishes* 28: 295-308.
- Whitfield A.K. 1994. A review of the ichthyofaunal biodiversity in southern African estuarine systems. *Annales Sciences Zoologiques Musee Royal de l'Afrique Centrale* 275: 149-163.
- Whitfield, A.K. 1998. Biology and ecology of fishes in southern African estuaries. *Ichthyological Monographs of the J.L.B. Smith Institute of Ichthyology* No. 2: 223pp.
- Whitfield, A.K., Beckley, L.E., Bennett, B.A., Branch, G.M., Kok, H.M., Potter, I.C. and van der Elst, R.P. 1989. Composition, species richness and similarity of ichthyofaunas in eelgrass *Zostera capensis* beds of southern Africa. *South African Journal of Marine Science* 8: 251-259.
- Whitfield, A.K. and Blaber, S.J.M. 1978. The distribution of the freshwater cichlid *Sarotherodon mossambicus* in estuarine systems. *Environmental Biology of Fishes* 4: 77-81.
- Whitfield, A.K., Blaber, S.J.M. and Cyrus, D.P. 1981. Salinity ranges of some southern African fish species occurring in estuaries. *South African Journal of Zoology* 16: 151-155.
- Whitfield A.K. and Paterson, A.W. 1995. Flood-associated mass mortality of fishes in the Sundays Estuary. *Water S.A.* 21(4): 385-389.
- Whitfield, A.K., Paterson, A.W., Bok, A.H. and Kok, H.M. 1994. A comparison of the ichthyofaunas in two permanently open eastern Cape estuaries. *South African Journal of Zoology* 29: 175-185.

Glossary

Abiotic: Non-living.

Aeolian: Applied to the act of the wind at or near the Earth's surface.

Algal bloom: A sudden increase in the density of phytoplankton within an area.

Allochthonous: Material originating from outside a system and which is transported into the system.

Anoxic: Without oxygen.

Anthropogenic: Human induced.

Assemblage: A predictable and particular collection of species within a biogeographic unit.

Autochthonous: Produced within a given habitat, community or system.

Autotroph: Organism that uses carbon dioxide as its main or sole source of carbon.

Bacterial mineralisation: Conversion of organic tissues to an inorganic state as a result of decomposition by bacteria.

Baseflow: Low flow remaining in a river system, even after a prolonged drought.

Benthic: Pertaining to the sea bed, river/estuary bed or lake floor.

Benthivores: Species that feed on the fauna and/or flora found in the benthic habitat.

Benthos: Those organisms living on, in or near the sea bed, river/estuary bed or lake floor.

Biodiversity: The variety of organisms considered at all levels, from genetic variants belonging to the same species through arrays of species to arrays of genera, families and still higher taxonomic levels; includes the variety of ecosystems, which comprise both communities of organisms within particular habitats and the physical conditions under which they live.

Biogeochemical cycle: Movement of chemical elements from organism to physical environment to organism in a more or less circular pathway.

Biomass: Any quantitative estimate of the total mass of organisms comprising all or part of a population or any other specified unit, or within a given area at a given time.

Biota: The total fauna and flora of a given area.

Biotic: Pertaining to life or living organisms.

Carbon isotopes: Naturally occurring isotopes of carbon.

Catadromous: Species that spawn in marine habitats and migrate to freshwater to mature.

Catchment: A natural drainage basin that channels rainfall into a single outflow.

Chlorophyll: Green pigment in plants that functions in photosynthesis.

Chlorophyta: Division of plants that includes the green algae.

Community: Any group of organisms belonging to a number of different species that co-occur in the same habitat or area and interact through trophic and spatial relationships; typically characterized by reference to one or more dominant species.

Conservation status: Assessment of the status of ecological processes and of the viability of species populations in an ecoregion.

Conspecific: Belonging to the same species.

Cyanophytes: Blue-green alga.

Denitrification: The release of gaseous nitrogen or the reduction of nitrates to nitrites and ammonia by the breakdown of nitrogenous compounds, typically by microorganisms when the oxygen concentration is low.

Deposit feeder: Any organism feeding on fragmented particulate organic matter in or on the substratum.

Detritus: Fragmented organic matter derived from the decomposition of plant and animal remains.

Diatom: Alga that belongs to the class Bacillariophyceae.

Dinoflagellate: Motile (flagellated) member of the class Dinophyceae.

Diversity: The absolute number of species in an assemblage, community or sample; a measure of the number of species and their relative abundance in a community.

Dominant: The highest ranking individual in a dominance hierarchy; an organism exerting considerable influence upon a community by its size, abundance or coverage.

Ebb and flow region: The region demarcating the headwaters of an estuary.

Ebb tide: Outgoing cycle of the tide.

Ecosystem: A community of organisms and their physical environment interacting as an ecological unit.

Ecotone: The boundary or transitional zone between adjacent communities or biomes.

Energetics: The study of energy transformations within a community or system.

Ephemeral: Short-lived or transient.

Estuary: A semi-enclosed coastal body of water which is either permanently or periodically open to the sea and within which there is a variation of salinity due to the mixture of sea water with fresh water derived from land drainage.

Euhaline: Salinities between 30‰ and 39.9‰.

Euryhaline: The ability of an organism to tolerate wide-ranging salinities.

Eutrophication: Over enrichment of a water body with nutrients, resulting in excessive growth of organisms and depletion of oxygen concentration.

Filter-feeder: Feeds by filtering suspended particulate organic matter from the water.

Flocculation: The settlement or coagulation of river-borne particles when they come into contact with sea water.

Flood: A flood is a strong increase in river flow, normally lasting for a relatively short period, and plays a major role in 'resetting' physical and environmental conditions.

Flood tide: Incoming cycle of the tide.

Fluvial: Pertaining to rivers and river action.

Food web: The network of interconnected food chains of a community.

Freshet: A temporary increase in river flow due to increased catchment run-off.

Freshwater: Water having a salinity of less than 0.5‰.

Groundwater: All the water that has percolated through the surface soil into the bedrock.

Habitat: The locality, site and particular type of local environment occupied by an organism.

Headwater: The source of a river or ebb and flow region of an estuary.

Heterogeneous: Having a non-uniform structure or composition.

Heterotroph: Obtains nourishment from exogenous organic material.

Hydrodynamics: Pertaining to the action of water, waves or tides.

Hypersaline: Water that has a salt concentration greater than the normal seawater average of 35‰.

Ichthyofauna: The fish fauna of a given region.

Impoundments: Artificially enclosed bodies of water.

Indicator species: A species, the presence or absence of which is indicative of a particular habitat, community or set of environmental conditions.

Infauna: The total animal life within a sediment.

Inorganic: Pertaining to, or derived from, non-biological material.

Interface: The contact surface between two contiguous substances or zones.

Intertidal: Pertaining the region between the spring tide low water mark and the spring tide high water mark.

Isosmotic: Having the same osmotic pressure.

Isotopes: Elements that occupy the same place in the periodic table but have a different atomic weight.

Life history: The significant features of the life cycle through which an organism passes, with particular references to strategies influencing survival and reproduction.

Limnetic: Pertaining to lakes or to other bodies of standing fresh water.

Macrobenthos: General term for larger organisms associated with bottom sediments.

Macrophytes: Large macroscopic plants.

Marine stragglers: Marine fish species that are occasionally recorded in estuaries.

Meiofauna: Microscopic or semi-microscopic animals that inhabit sediments (often interstitial).

Mesohaline: Salinities between 5‰ and 17.9‰.

Microalgae: Unicellular plants.

Microbial: Of or pertaining to micro-organisms.

Microphytobenthos: Benthic microalgae.

Mineralisation: The conversion of organic tissues to an inorganic state as a result of decomposition by sediment associated micro-organisms.

Morphological: Pertaining to the form and structure of an organism, with special emphasis on external features.

Neap tide: Smallest tidal range experienced due to the sun's gravitational forces acting against those of the moon.

Nutrient: Any substance required by organisms for normal growth and maintenance.

Oligohaline: Salinities between 0.5‰ and 4.9‰.

Oligotrophic: Having low primary productivity.

Organic: Pertaining to or derived from living organisms, or to compounds containing carbon as an essential component.

Organism: Any living entity.

Osmoregulation: The control of osmotic processes (water/salt balance) within an organism.

Pelagic: Pertaining to the water column.

pH: The negative logarithm of the hydrogen ion concentration giving a measure of acidity on a scale of 0 (acid) through 7 (neutral) to 14 (alkaline).

Phylogenetic: Pertaining to evolutionary relationships within and between groups.

Phytoplankton: Microscopic plants drifting in the plankton.

Piscivorous: Feeding on fish.

Plankton: Organisms unable to maintain their position or distribution independent of the movement of water masses.

Polyhaline: Salinities between 18‰ and 29.9‰.

Porewater: Water completely surrounded by soil materials.

Primary production: The synthesis of complex organic substances from simple inorganic substrates by an autotrophic organism.

Redox potential: Quantitative measure of the ease with which a substance is oxidised or reduced.

Retention time: The period in which the volume of water in an estuary, or in a part of the estuary, is being replaced by river and/or tidal flows.

River-estuarine interface (REI) region: The REI region is that part of an estuary where river and estuarine waters mix, and where the vertically integrated salinity is usually less than 10‰.

Runoff: That part of precipitation that is not held in the soil but drains freely away.

Salinity: A measure of the total quantity of dissolved ions in water.

Species: A group of organisms recognized as distinct from other groups.

Spring tide: Maximum tidal amplitude during the new and full moon periods.

Stenohaline: Species that cannot tolerate a wide range of salinities during their life histories.

Stratification: A word used to describe a layered water body.

Subtidal: Pertaining to the region below the spring tide low water mark.

Suspensoids: Fine particles of an insoluble substance uniformly dispersed throughout the water column.

Sympatric: Describes populations, species or taxa occurring together and occupying the same or different habitats in the same geographical area.

Taxon: A taxonomic group of any rank, including all the subordinate groups

Terrigenous: Originating from land.

Tidal head: The uppermost region of an estuary affected by sea tides.

Topography: All natural and man-made surface features of a geographical area.

Trophic: Pertaining to nutrition.

Trophic levels: The sequence of steps in a food chain or food pyramid from producer to primary, secondary or tertiary consumer.

Turbidity: Describes levels of suspended matter in the water column.

Zoobenthivorous: Feeding primarily on zoobenthos.

Zoobenthos: Bottom-dwelling animals.

Zooplanktivorous: Feeding primarily on zooplankton.

Zooplankton: Minute animals drifting in the water column.