DPWS

NSW DEPARTMENT
OF PUBLIC WORKS
AND SERVICES

Manly Hydraulics Laboratory

Berowra Creek Estuary Processes Study Estuarine Water Quality

Report MHL928
December 1998

BEROWRA CREEK ESTUARY PROCESSES STUDY
WATER QUALITY PROCESSES

-

Report No. MHL928

NSW Department of Public Works and Services
Manly Hydraulics Laboratory

Report No. MHL928
DPWS Report No. 98040
ISBN 0 7313 0787 9
MHL File No. IN0101/1
First published (December) 1998

QA SYSTEM Manly Hydraulics Laboratory is Quality System Certified to AS/NZS ISO 9001:1994.

Summary

Water quality issues in the Berowra Creek estuary have been investigated through review of previous information and a set of field experiments conducted over the summer of 1997-98. Field investigations involved specific studies carried out by Manly Hydraulics Laboratory (MHL) - mixing, chlorophyll-a and light measurements; Environment Protection Authority (EPA) - sediment nutrient fluxes; Hornsby Shire Council (HSC) - water quality sampling; Department of Land and Water Conservation (DLWC) Estuaries - phytoplankton/zooplankton investigations. In addition, the results of previous modelling work were analysed and interpreted in terms of the underlying processes.

Flushing rates vary along the estuary and according to the topography. Upstream in the shallow and narrow reaches, water residence times of 30 hours due to the fresh water input preclude excessive algal blooms. Around the deep holes near Calabash Bay the residence time increases to around 8-15 days and downstream of Bujwa Bay the depth decreases and residence times are around 30 hours.

The downstream reaches are characterised by large areas of shallow mud flats comprised of fine material which is readily suspended by the tidal currents. The waters are generally more turbid in these reaches, leading to reduced light availability. The sandier sediments upstream of Berowra Ferry are less prone to resuspension and the water is clearer than the downstream reaches.

Nutrients are delivered to the system from catchment inflows that include discharges from two STPs. The STPs contributed more than 25% of the total phosphorus load, and 97% of total nitrogen load during the three months of the investigations. Concentrations of the bioavailable forms of nutrients (dissolved inorganic phosphorus and dissolved inorganic nitrogen) near the deep holes rarely dropped to levels that could limit the phytoplankton growth. Only during a major bloom (chlorophyll-a ~ 70 µg/L) nutrients decreased to levels that would limit growth.

The highly variable light field and spring to neap variations in dispersion are thought to control the algal biomass distribution in the estuary. The SALMON-Q model results also indicated that light and dispersion were more important than nutrient limitation for controlling algal biomass and possible blooms.

The sediment nutrient fluxes were found to be relatively small and did not contribute significantly to the overall nutrient budget.

Over the study period of December 1997 to February 1998, zooplankton biomass was highly responsive to perturbations in phytoplankton biomass. The species composition of the phytoplankton was such that zooplankton were able to consume them and make effective use of them as a nutrient source. Flushing times within the estuary were sufficiently slow to enable zooplankton to undergo increased productivity in terms of egg production (numbers)

and somatic growth (size). Zooplankton were therefore able to increase in size and abundance and were able to exert top down control on phytoplankton. Environmental conditions (nutrient availability, light intensity, temperature) were such that the combined effects of zooplankton and flushing were unable to reduce phytoplankton abundance below ~ 10 μg Chla/L.

The major processes affecting algal blooms in Berowra Creek estuary appear to be light availability and dispersion. At present there are sufficient nutrient inputs to sustain regular blooms. There is some evidence that grazing also contributes to the decrease in phytoplankton biomass.

Acknowledgements

This report details the findings of a number of component programs carried out for the Berowra Creek Estuary Processes Study. The following people contributed to specific sections of the report:

- Department of Land and Water Conservation for zooplankton data
- David Rissik for sections dealing with zooplankton and phytoplankton/zooplankton interactions
- AWT Science and Environment for SALMON-Q model review
- Dr Ian Fisher for Chapter 2.

The report was compiled by Dr David van Senden with support from the following MHL personnel: Roman Kadluczka, David Allsop, Megan Jensen and Kerrie Harrison. Georgina Sherwin from Environmental Sciences & Engineering is also acknowledged for her support in compiling the final product.

Table of Contents

List of Figures

List of Tables

1. Introduction

The Berowra Creek estuary is a tributary of the Hawkesbury River. The estuary is approximately 24 km long with the tidal limit at Rocky Fall Rapids. It is fed by a number of local tributaries that drain the steep catchments adjacent to the waterway. The catchment is 310 square kilometres and is comprised mainly of natural bushland with urban and semi-rural developments along the ridges of the upper catchment areas. The estuary has a waterway area of approximately 13 km^2. A number of small river settlements exist in areas of accessible foreshore along the creek (Figure 1.1).

The urban development has slowly evolved over the past 200 years with accelerated growth during the past 30 years associated with recent land releases. To service the population expansion two sewage treatment plants (STPs) were constructed near the urban areas on Calna Creek and Waitara Creek. These plants discharge treated effluent to their respective creeks that then feed into the Berowra Creek estuary. Measurements and model simulations indicate that on an annual basis about 83% of the total nitrogen load and 22% of the total phosphorus load to the estuary may be attributed to the STP discharges while the remainder is due to the catchment loads. During extreme wet weather events the plants become overloaded with stormwater runoff and the effluent may be bypassed directly to the creek.

The region is a popular recreational destination for a range of, mostly outdoor, activities including bushwalking, camping, boating and fishing. Recent changes in land use patterns and the increased pressure on the estuarine environment due to a range of human influences have resulted in a deterioration of the water quality in the Berowra Creek estuary. The need for an integrated management plan that addresses the needs of the users while recognising the desire to maintain the environment is now being addressed through estuary and catchment management programs.

The estuary has been classified as eutrophic, algal blooms being a regular occurrence. Previous water quality studies focused on the nutrient concentrations as the main factor leading to the algal blooms. The role of light, stratification, flushing and zooplankton grazing in determining the size and extent of the algal blooms has not previously been described although their relevance has been alluded to.

As part of the estuary management process, the estuary processes study aims to provide an assessment of the key processes that determine the ecological structure of function of the estuary. This information then provides a basis for the assessment of likely impacts due to possible management actions and hence forms an important component of the management planning process. The study has aimed to assess the short term variability associated with the spring/neap cycle and the influence of stratification and light attenuation on the algal blooms. A series of experiments was conducted over the summer of 1997-98 to assess the algal bloom processes and their interactions. A detailed review and assessment of the SALMONQ model results was also carried out to assess the dominant processes leading to the simulated results.

1.1 Previous Relevant Studies

Previous studies of the water quality in Berowra Creek estuary have highlighted the role of nutrient inputs from the catchment and also the potential for sediment nutrient release. Modelling studies conducted as part of an assessment of the effects of the nutrient inputs from sewage treatment plants demonstrated that reduction of the nutrient loads from the STPs would likely result in a reduction in the algal blooms. From this work it was not clear which processes in the model were causing the model response and whether these controlling processes were applicable to the environment.

1.1.1 Ohfunato Bay and Kurihama Bay, Japan

Tsuruya and Hibino (1998a and 1998b) conducted investigations in Ohfunato Bay and Kurihama Bay, two enclosed bays on the Japanese east coast containing deep holes. These studies found that water within the bays, particularly in the holes, can become highly anoxic and accumulate high levels of pollutants due to poor flushing. These studies found that flushing can be influenced by:

- seasonal variations in ocean water level gradients and salinity associated with large-scale meteorologic and oceanographic phenomena. In particular, flushing was enhanced when ocean waters were highly saline. It was found that these high saline waters would intrude into the deepest sections of the bays, displacing the waters previously there. Conversely, low salinity ocean waters were found to enhance stratification;

- inflows of freshwater from the upstream catchment. These caused a reduction in salinity of the upper waters of the bay and enhanced stratification;

- an increase in salinity stratification was found to be associated with the incidence of severe storms. Apparently the impacts of fresh water inflows to the bays were greater than the effects of mixing due to wind.

1.1.2 Algal Blooms, USA

Anderson et al. (1993) and Anderson (1994) suggested that marine ecosystems and fisheries resources are increasingly under threat from the effects of algal blooms, however the ecology, toxicology and oceanography associated with algal blooms is still only poorly understood.

In the USA, toxic and other harmful effects from algal blooms have been significant enough to prompt initiation of significant research programs into the organisms causing harmful algal blooms, and the effects these organisms have on coastal ecology and human health risks. Past impacts include:

- mass mortalities of wild and farmed fish and shellfish,
- human illness and death from consumption of fish and shellfish,
- death of marine mammals, seabirds and other animals, and
- significant alteration of marine habitats and trophic structures.

US experience shows that whereas harmful algal blooms (HABs) previously occurred only occasionally in scattered locations, they now frequently threaten almost the entire US coastline.

In the US, a national research program has been established entitled Ecological and Oceanography of Harmful Algal Blooms (ECOHAB). This has three program elements focusing on the research into organisms themselves, the role of environmental factors in the ecology of harmful algal species and their competitors, and the ways in which HABs impact on marine food webs.

1.2 Data Collection – Nutrients, Algae, Chlorophyll-a

A data collection program was carried out over summer between November 1997 and February 1998. The aim of this exercise was to provide information on the spatial and temporal variability of the algal blooms in Berowra Creek and to investigate the influence of nutrients, light and stratification on the bloom dynamics.

The data collection program involved regular field sampling exercises and deployment of in situ recorders. Nutrient and water samples for phytoplankton identification were collected by Hornsby Council, physico-chemical, meteorologic variables and thermistor chain data were collected by MHL, DLWC personnel carried out zooplankton tows and the EPA conducted nutrient sampling and sediment nutrient release experiments. These experiments are described in Section 3.

1.2.1 Phytoplankton and Zooplankton

Algal blooms have been a feature of the Berowra Creek estuary for several years. Temporal and spatial variability of phytoplankton biomass and any associated or related responses of zooplankton in the estuary are not well understood. The investigations carried out by DLWC have therefore been fairly exploratory.

It has been recognised that in order to effectively manage the estuary it is necessary to develop an understanding of the processes leading to excessive phytoplankton production and to investigate the trophic interactions within the system. Understanding the natural control mechanisms acting within the system can assist with the selection of appropriate management options.

Species and size composition of planktonic copepods have been reported to change in response to eutrophication. Subsequently changes in the zooplankton community and size structures have the potential to affect feeding by juvenile and larval fish.

Zooplankton responses to phytoplankton blooms can be indicated by increased biomass of zooplankton in response to the bloom, or by declining phytoplankton biomass in response to grazing by increased zooplankton biomass.

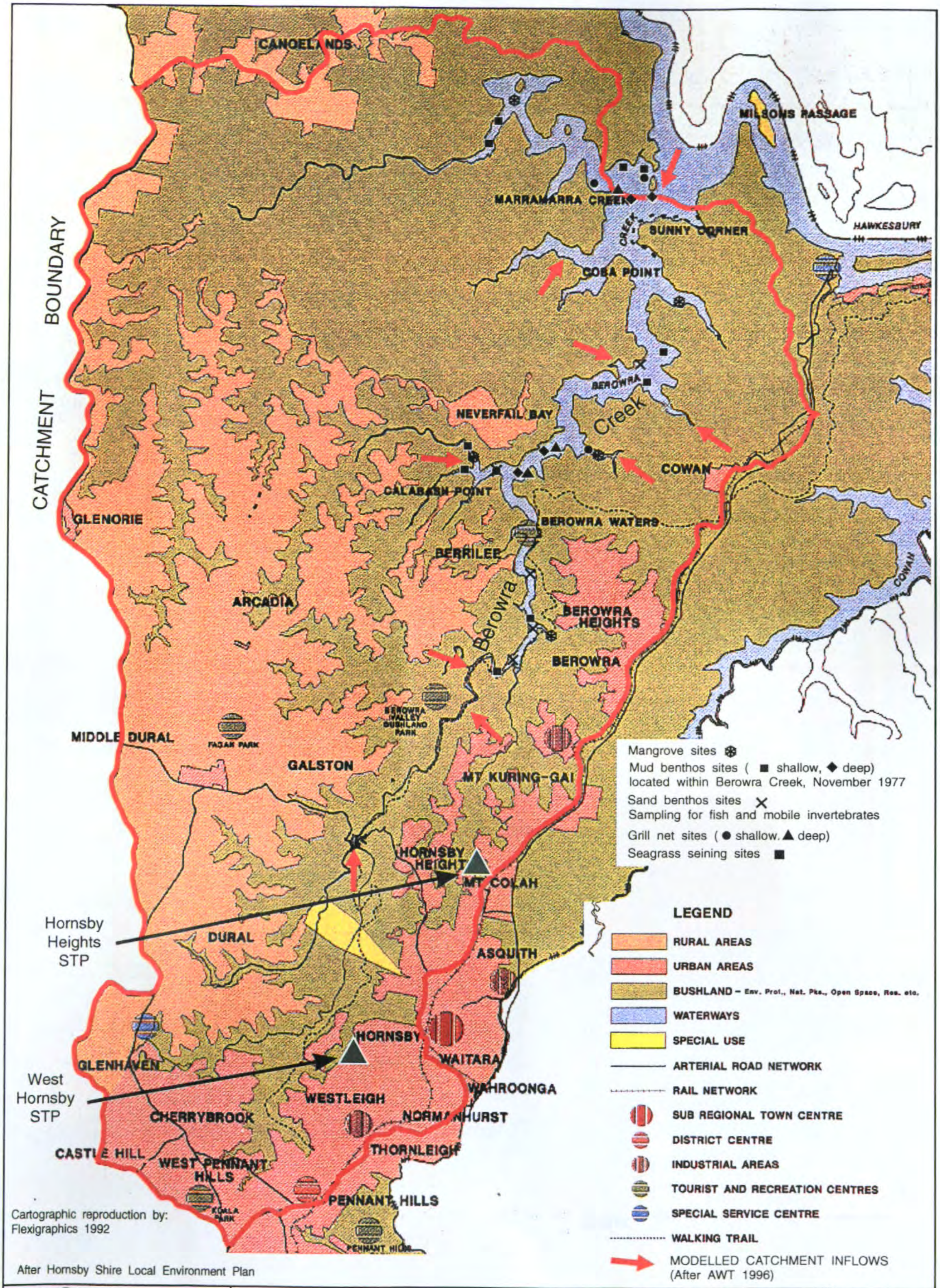
Zooplankton size and biomass are important attributes to consider when assessing zooplankton responses to shifts in phytoplankton biomass and in the succession of dominant phytoplankton taxa.

1.3 EPA Work

The EPA conducted studies within Berowra Creek focusing on sediment nutrient fluxes. As phytoplankton blooms generally occur in the deeper section of the estuary between Berowra Ferry and Oaky Point, it was assumed that they might be sustained by the release of nutrients from bottom sediments within deep holes. As the concentrations of nitrate are relatively high in Berowra Creek waters, the study focused on potential benthic release of phosphate from the sediment. Benthic nutrient fluxes under oxic and hypoxic/anoxic conditions were compared whilst the potential for nutrient release from these sediments was also investigated. Sediment-water fluxes of nutrients were investigated by incubating sediment cores.

Three field trips were carried out between September and December 1997, deep (16-17 m) and shallow (7-8 m) sites were sampled. Surface and bottom water samples were collected for nutrient concentration analysis and water column temperature, salinity and dissolved oxygen concentration were recorded during each field trip. Sediment cores were also collected during each trip.

In addition to the above field trips, sediment cores were collected from a deep and a shallow site on 23 January 1998, and a sediment slurry experiment was then carried out.



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

CATCHMENT, LANDUSE AND ESTUARINE ECOLOGY

MHL Report 928

Figure 1.1

DRAWING 928BER11.CDR

2. Salmon-Q Model Processes

2.1 Model Overview and Results

The SALMON-Q model is a suite of programs developed by HR Wallingford (UK). It is used to model mathematically the hydraulic and water quality phenomena in one-dimensional channel networks. The model is a time-dependent, fixed element (Eulerian) model that simulates fluvial flow, tidal propagation, saline intrusion and water quality processes in water bodies where vertical and lateral variation in water quality are small compared with longitudinal variations for most of the time. The model simulates the oxygen and nutrient balance and primary productivity including phytoplankton, macrophytes and benthic algae, using an implicit six-point finite difference scheme that is well suited to solving the hydrodynamic advection-dispersion equations.

Sydney Water and AWT EnSight have developed SALMON-Q models of the Hawkesbury-Nepean River and some of its major tributaries (including Berowra Creek), during the 1990s. Both the flow and quality in these waterways are strongly affected by the flow and quality generated from the catchments surrounding them (including point sources). To account adequately for these catchment influences, another modelling package (HSPF, developed by Aqua Terra Inc.) has been used to simulate the inflows and quality received from these catchments via more minor tributaries.

A detailed SALMON-Q model of the Berowra Creek estuary was developed by AWT EnSight for the specific purpose of evaluating options for the upgrading of Sydney Water Corporation's West Hornsby and Hornsby Heights sewage treatment plants (STPs). Both these STPs discharge treated effluent into tributaries of Berowra Creek. The Berowra Creek catchment was subdivided into 27 sub-catchments, see Figure 2.1, that were individually modelled using the HSPF package. The flow and concentration from these sub-catchments were aggregated into ten time varying inputs to the SALMON-Q estuary model. Figure 2.2 shows the input locations and associated aggregated sub-catchment areas.

In 1997, these models were calibrated against flow and water quality data (including nutrient and algal counts) collected over the period July 1994 to January 1996. This period includes two consecutive summer seasons that exhibited quite different algal growth, as well as a wide range of inflows from the catchment. In addition, calibration of the hydrodynamic components of the SALMON-Q model was carried out against the water level and other physical data collected during May to November 1995 by MHL.

The physical data collected in 1995 (MHL 1996) showed that the estuary could be continuously stratified (vertically) for periods of a few weeks after a major fresh water inflow from the catchment. The flows obtained from a one-dimensional model would be at variance with those actually occurring during those periods. Similarly, oxygen levels at depth will be much lower than the modelled levels. However, the consequences of these errors for prediction of nutrient and algal levels were tested and found to be small, especially as algae are generally washed out of the estuary during such events. From later data collection in the

1997-98 summer, the middle estuary was shown to be continuously weakly stratified in dry periods (MHL 1998a), but the tidal action generated sufficient vertical mixing to ensure that there was no substantial vertical variation in water quality variables. Consequently, the use of the one-dimensional SALMON-Q model of this estuary was still considered to be justified.

The initial environmental criterion for evaluation of options for STP upgrading was the restriction of algal concentrations to levels below those specified in the national guidelines for estuaries (ANZECC 1992). Inspection of the algal count data showed that high levels of algae were generally dominated by either diatom or dinoflagellate taxa. Each taxon dominated under different environmental conditions. Consequently, diatoms and dinoflagellates were represented as two distinct groups of algae in the SALMON-Q model of Berowra Creek estuary. Neither of these taxa is known to be capable of fixing atmospheric nitrogen, so that it is feasible for reductions in soluble nitrogen inputs to the estuary to restrict algal growth. This contrasts with the situation in the Hawkesbury River where the major algal blooms occur in the tidal, but fresh, section of the river. In that case, the blooms of most concern are composed of cyanobacteria (blue-green algae), of which one taxa (Anabaena) is known to fix atmospheric nitrogen.

In this section, the major processes in the SALMON-Q model are first described. Then the relative significance of the factors that control algal growth in the calibrated model are discussed, in terms of their effect on algal blooms in Berowra Creek estuary simulated during the three-year period 1992-1994.

2.2 Description of SALMON-Q Model Processes

The following description has largely drawn upon the information provided by Wallingford Software (1992).

2.2.1 Hydrodynamics and Dispersion

The channel network is considered to be composed of a number of reaches, each of which is a continuous channel bounded by a node at either end. A node may be:

- a land (discharge) or sea (level) boundary
- a hydraulic structure, or
- a junction of up to five channels.

Each reach is subdivided into a number of fixed elements and the partial differential equations for conservation of momentum and mass (continuity) are solved to obtain the water level and flow rate within each element at successive points in time.

The concentration of each water quality constituent in each element is similarly obtained by solving the respective partial differential equation for conservation of mass. These equations include the effects of advection, dispersion, lateral loadings and internal sources/sinks. The advection term is usually considered to be well defined as a result of hydrodynamic calibration against water level and discharge data.

To allow for the major factors that affect dispersion, SALMON-Q uses a dispersion coefficient that is a sum of three terms:

- a constant
- a term proportional to the ratio of local discharge to peak discharge at the downstream boundary, and
- a term proportional to the ratio of local longitudinal salinity gradient to total longitudinal salinity gradient.

These appropriate values of coefficients in these three terms are obtained by regarding salinity as a conservative tracer, so that its distribution along the estuary is due only to advection and dispersion by flow and salinity gradient. The coefficients can then be obtained by calibrating the model against salinity measurements during a dry period in the estuary of interest.

2.2.2 Sediment

Although SALMON-Q treats the water column as well mixed (i.e. depth-averaged), it defines two additional layers in the vertical, to enable representation of sediment-water interactions. The bed is composed of consolidated mud. A less dense layer, termed fluffy mud, overlies the bed. Particulate material settling out of the water column first enters the fluffy mud layer, until a predefined thickness is exceeded. As more particulates settle out subsequently, an equal mass is transferred from the fluffy mud to the bed. As the fluffy mud consolidates into the bed, water is also trapped within the bed pores. Dissolved substances are transported into the pores in proportion to the mass of mud transferred.

The contents of the fluffy layer continue to interact directly with the water column biochemically, e.g. its BOD consumes oxygen from the water column. Similarly, the bed contents interact with the pore water, but are isolated from the water column, unless resuspension occurs.

Material from both layers can be resuspended into the water column. The entire fluffy mud layer is resuspended whenever its critical shear stress is exceeded. The bed is usually assumed to have a higher critical shear stress and its rate of erosion is proportional to the amount by which the shear stress exceeds the critical level. Substances dissolved in the pore water are also returned to the water column during bed erosion.

2.2.3 Light and Temperature

Light and temperature play a major role in determining the growth rate of algae. This is discussed in detail below, along with other possible limitations to algal growth. The SALMON-Q model accepts hourly, daily or constant incoming solar radiation as an input for the purpose of assessing light limitation.

Most of the biochemical and biological processes included in SALMON-Q are temperature dependent. The temperature can be input as a constant value applicable throughout time and space. It can alternatively be treated as a transportable variable, to account for differences between tributary, river, estuary and sea temperatures. Tributaries and other discharges can have thermal loads associated with their flow. Heat transfer to or from the atmosphere is proportional to the difference between current and base temperature.

2.2.4 Oxygen

In the SALMON-Q model, the equation for conservation of mass for oxygen is governed by the transport and interactions of over 15 variables. Oxygen enters the channel network from minor tributaries (HSPF models), with rainfall, from surface reaeration, and as a product from photosynthesis by aquatic plant life. The amount of oxygen produced is stoichiometrically related to the production of algal carbon.

Oxygen is consumed by respiration of algae and in the oxidation of carbonaceous material (which is represented as BOD). Oxygen is also consumed in the stage of the nitrogen cycle where organic nitrogen is hydrolysed to ammonia, which is in turn oxidised to nitrate. If the level of oxygen in the water is too low to allow oxidation of carbonaceous material, then some of the available nitrate (and eventually sulphate if the nitrate supply is exhausted) will be reduced to provide the necessary oxygen.

The decay of organic materials and the nitrification process are all assumed to follow first order kinetics. The associated reaction rate 'constants' are assumed to be Arrhenius functions of temperature.

2.2.5 Nutrient Sources and Sinks

There are three macronutrients necessary for the growth of algae and hence are of importance in their modelling. These are nitrogen, phosphorus and silica. The latter is important only in the case of diatoms, which require substantial amounts to build into their cell walls. Before considering the nutrient cycling processes contained in the SALMON-Q model, it should be recognised that there are generally major inputs of these nutrients to the modelled waterways from the surrounding catchment areas, particularly during storm events. For waterways in the Hawkesbury-Nepean basin, these time varying loadings are generated by the HSPF dynamic models of the relevant sub-catchments.

The SALMON-Q model includes three forms of nitrogen – organic, ammoniacal and oxidised. Organic nitrogen is modelled as two dissolved substances that are converted by bacterial action to ammoniacal nitrogen at significantly different rates (in assumed first order reactions). This formulation enables adequate representation of STP discharges. Ammoniacal nitrogen is in turn nitrified to oxidised nitrogen at a rate non-linearly dependent on temperature, salinity and suspended solids concentration. Complete oxidation to nitrate is assumed, rather than a two-stage process with nitrite as an intermediate product. The model also assumes that algae can take up only the nitrate form, unless they are capable of fixing dissolved nitrogen derived from the atmosphere. The nitrate taken up is released back to the water column during decay of algal detritus. The interaction of nitrogen cycling processes with oxygen is described above.

Phosphorus is also present in the water column in several forms. Algae are assumed to be able to take up only the soluble inorganic form (phosphate). However, the only other forms in the water column that are represented in the model are that adsorbed to suspended sediment and that contained in algae. It is usual to regard the total soluble form as available to algae (i.e. includes soluble organic phosphorus), so that an estimate of total phosphorus (TP) can be obtained by adding the three model components together for comparison with the usually available TP measurements. The phosphorus contained in the algae is assumed to be released back to the water column in soluble form, during the decay of algal detritus.
Under oxic conditions, the amount of phosphorus that can be adsorbed to unit mass of sediment is assumed to be an asymptotic function of the soluble phosphorus concentration

known as a Langmuir adsorption isotherm. If the actual amount adsorbed is less than the Langmuir estimate, then the rate of adsorption from solution is directly proportional to the difference between the actual and Langmuir amounts.

Desorption is assumed to occur only when the surrounding water effectively becomes anoxic (i.e. dissolved oxygen is less than 5% of saturation). Then all adsorbed phosphorus is instantaneously released to the water column as soluble phosphorus, if the sediment is either in the water column or fluffy mud. However, this anoxic condition is most likely to occur in the bed, in which case the phosphorus is released into the pore water, rather than the water column.

The SALMON-Q model also assumes that soluble silica is a limiting nutrient for diatoms. This soluble form is derived from breakdown of silicate polymers that accompany catchment runoff. Consequently, its representation firstly includes advection and dispersion of soluble silica inputs from the catchment. As well as being taken up by algae from the water column, soluble silica is also assumed to be released back to the water column from the detrital compartment, due to the decay of the diatom component.

2.2.6 Primary Production

Primary production is the process by which plants (including algae) convert inorganic materials (carbon dioxide, water and nutrients) into additional plant biomass by photosynthesis. Oxygen is a by-product. Nutrients and carbon dioxide are released to the water column and/or atmosphere as the plants respire and as dead plants (detritus) decay. The SALMON-Q model contains three types of primary producers:

- planktonic algae – single-celled plants that live suspended in the water column
- benthic algae – single-celled plants that live on the bed surface
- rooted macrophytes – large plants that are rooted in the bed.

To simulate the temporal succession of dominant algal taxa often observed that is a result of different responses to a given set of environmental conditions, the SALMON-Q model includes four groups of planktonic algae that have their growth limited by the following factors:

- green algae –light, temperature, nitrogen and phosphorus
- diatoms – additionally limited by silica
- non-nitrogen fixing blue-green algae (cyanobacteria) – as for green algae
- nitrogen fixing blue-green algae (cyanobacteria) – as for green algae, except no nitrogen limitation.

The biomass of each planktonic algal group is increased in each time step by production (growth) and decreased by respiration and natural mortality (to create detritus). Grazing by higher trophic levels can be included only as additional mortality.

Growth rates for different algal groups are calculated according to the following equation:

$$P(t) = P_{max} \cdot \mu_L \cdot \mu_T \cdot \text{Min}(\mu_N, \mu_P, \mu_S) \qquad (2.1)$$

where $P(t)$ is algal growth rate at time t
P_{max} is the maximum growth rate
μ_L is the growth limitation factor due to light
μ_T is the growth limitation factor due to temperature, and
μ_N, μ_P, μ_S are the growth limitation factors due to availability of nitrogen, phosphorus and silica, respectively.

All growth limitation factors have a range of zero to one. Only those nutrient limitations applicable to a specific algal group are activated during these calculations. As mentioned earlier in this section, only the nitrate, soluble phosphorus and soluble silica forms of nutrients are assumed to be available for algal growth. In the case of nitrogen-fixing cyanobacteria, they are assumed to fix dissolved nitrogen only when insufficient nitrate is available, since fixation is less energetically efficient.

The maximum growth rates are parameters specified as data in an input file. The form of each limitation factor is given in Table 2.1.

Table 2.1 Formulae for Planktonic Algal Growth Limitation Factors

Variable	Definition	Formulae	
μ_L	Light limit Daily average from hourly values	$\mu_L^{(hour)} = \dfrac{e^{(1-\frac{L}{L_{max}} e^{-K_d H_{max}})} - e^{(1-\frac{L}{L_{max}})}}{K_d\, H_{max}}$ where L is hourly solar radiation between 5 a.m. and 8 p.m. L_{max} is optimum radiation for algal growth K_d is the light extinction coefficient H_{max} is the maximum depth of the element $\mu_{L\,(avg)} = \Sigma\,(\mu_{L\,(5)}, \mu_{L\,(6)} \ldots \mu_{L\,(20)})/15$	(2.2) (2.3)
μ_T	Temperature limit	$\mu_T = \theta^{-abs(T-Tc)}$ where T_c is the temperature for maximum growth θ is a parameter	(2.4)

μ_N	Nitrogen limit	$$\mu_N = \frac{NO_x}{NO_x + S_N}$$ where S_N is the half saturation coefficient for nitrogen and NO_x is the concentration of oxidised nitrogen	(2.5)
μ_P	Phosphorus limit	$$\mu_p = \frac{FP}{FP + S_p}$$ where FP is filtered phosphorus concentration and S_p is the half saturation coefficient for phosphorus	(2.6)
μ_S	Silica limit	$$\mu_s = \frac{Si}{Si + S_s}$$ where Si is silica concentration and S_s is the half saturation coefficient for silicon	(2.7)

The light limitation factor is the depth and time integrated form of the Steele (1965) equation. The light extinction coefficient is a linear function of total algal concentration and suspended solids. The temperature limitation is that recently recommended by Reynolds (pers. comm.) The individual nutrient limitations are Michaelis-Menten functions, long used in this context, where S_n and S_s are the half saturation coefficients for each nutrient.

Algal mortality is represented as a first order decay process, whereas loss rate of carbon due to respiration is given by a temperature-dependent Arrhenius function. Dead algae become detritus, which decays according to first order kinetics.

Although SALMON-Q contains algorithms for benthic algae and rooted macrophytes, their representation will not be described here, as they were not activated in the Berowra Creek estuary model.

2.3 Algal Bloom Simulation

2.3.1 Hydrodynamics and Dispersion

The hydrodynamic component of the SALMON-Q model of Berowra Creek estuary was calibrated for various flow conditions against water level data provided by MHL from four sites over the period May-September 1995. Seven tidal harmonics were found to be needed to adequately define the level variation at the downstream boundary (confluence with the Hawkesbury River). Salinity concentrations at four sites provided by MHL were also satisfactorily calibrated under low inflow conditions by selection of suitable values of the three components of the dispersion coefficient.

2.3.2 Nutrient Sources and Sinks

The existing catchment conditions, including runoff and nutrient generation rates from all catchment land uses, were refined in accordance with the most recent data obtained from Hornsby Shire Council and Sydney Water. The Berowra Creek catchment was subdivided into 27 sub-catchments, see Figure 2.1, that were individually modelled using the HSPF package. The hydrologic component of the HSPF models was calibrated against flow data from the three gauge sites available (Waitara, Tunks and Pyes Creeks). Input rainfall time series were generated by a modified Thiessen technique using hourly data from two rainfall stations in the catchment. Other required meteorological time series (cloud cover, dewpoint, wind speed and evaporation) were generated from data from Sydney Airport (Mascot), except for solar radiation which was available only from Blacktown during the period of interest.

The locations and nutrient loadings for EPA-licensed discharges and areas serviced by septic tanks were revised according to recent catchment maps and relevant studies (AWT EnSight 1997, NSW EPA et al. 1996, Martens and Warner 1994). Updated time series of treated effluent discharges were also made. Water quality components of the HSPF models were calibrated against data (from both grab samples and autosamplers) at five locations, two of which are located in Berowra Creek. The calibration period was from April 1994 to February 1996, with a preceding warm-up period of twelve weeks.

The flow and concentration from the 27 sub-catchments were aggregated into ten time-varying inputs to the SALMON-Q estuary model. Figure 2.2 shows the input locations and associated aggregated sub-catchment areas. Loadings from boats and picnic areas into the estuary were also estimated and included in the inputs to the SALMON-Q model. Concentrations of water quality variables at the downstream boundary were input as the average of the corresponding variables at the nearest station in the Hawkesbury River upstream and downstream of the confluence with Berowra Creek. Further details of the calibration procedure and results are provided in AWT EnSight (1997a) or Sydney Water Corporation (1997).

2.3.3 Limiting Factors for Algal Growth

As discussed in Section 2.1, diatoms and dinoflagellates were represented as two distinct groups of algae in the SALMON-Q model of Berowra Creek estuary. No other algal groups (greens, blue greens) were included, as their densities are generally low relative to the other two groups. The parameter values obtained from calibrating the model are shown in Table 2.2.

Table 2.2 Calibrated Parameter Values Used in the Limitation Factor Analysis

Algae	L_{max} J/cm^2	T_c °C	θ	S_N mg/L	S_P mg/L	S_s mg/L
Diatoms (Di)	30	21	1.01	0.05	0.003	0.3
Dinoflagellates	45	25	1.04	0.06	0.004	-

To compare various options for STP upgrading, the calibrated HSPF and SALMON-Q models were run using the meteorological conditions from the three-year period 1992-94 as inputs. The values of the algal growth limitation factors obtained for 'existing conditions', calculated using equations 2.2-2.7 over this period, are shown in Figures 2.3-2.5 for three sampling sites well spaced along the estuary – Square Bay (NB11), Berowra Ferry (NB02) and Crosslands (NB06), in upstream order. The site locations are shown in Figure 2.2 and the maximum depths used in the calculation of their light limitation function (see Equation 2.3) are shown in Table 2.3.

Table 2.3 Depths Used in the Limitation Factor Analysis

Site	**Square Bay (NB11)**	**Berowra Ferry (NB02)**	**Crosslands (NB06)**
H_{max} (m)	13.7	10.5	1.75

For each site, the time series for individual nutrient limitation factors are first shown for each algal group (e.g. Figures 2.3 a) and b) for diatoms and dinoflagellates at Square Bay). These plots are followed by plots, for each algal group, of the combined (minimum) nutrient limitation with the light and temperature limitations (e.g. Figures 2.3 c) and d) for Square Bay). The values of these three limitations are multiplied to obtain the final value of limitation to algal growth given by Equation 2.1.

At Square Bay, phosphorus is the most limiting nutrient in years 1992-93, controlling both diatom and dinoflagellate growth, whereas nitrogen limited algal growth in 1994 (Figures 2.3 a) and b). Comparison of light, temperature and nutrient limitations shown in Figures 2.3 c) and d) indicates that light limitation was the major factor controlling both diatom and dinoflagellate growth. There were a few short periods in 1994 in which nutrients (nitrogen) was of equal importance to light in limiting algal growth, as light and nutrient limitations are multiplicative.

At Berowra Ferry, phosphorus is the most limiting nutrient, again controlling both diatom and dinoflagellate growth most of the time (except for two short periods in 1994, in which nitrogen was limiting – Figures 2.4 a) and b). Light most significantly limited algal growth through the whole period, although nutrient (phosphorus) limitation was of equal importance in the summer seasons of 1993-94 and 1994-95 – Figures 2.4 c) and d).

At Crosslands (Figure 2.5), the concentrations of nitrogen and phosphorus were higher because this site is closer to the discharge point of West Hornsby STP, which discharges large nutrient loadings. Consequently, silica became the nutrient limiting diatom growth. However, nutrients were much less limiting at this site than at the sites further downstream. Light limitation was an even more significant factor controlling algal growth at this site than it is for the other two sites downstream, but this is due to reduction in nutrient limitation, rather than greater light limitation.

There is a marked seasonality in the temperature limitation on dinoflagellates at all sites, but almost no temperature limitation on diatoms. This reflects the general ability of diatoms to grow at lower (winter) temperatures than the dinoflagellates.

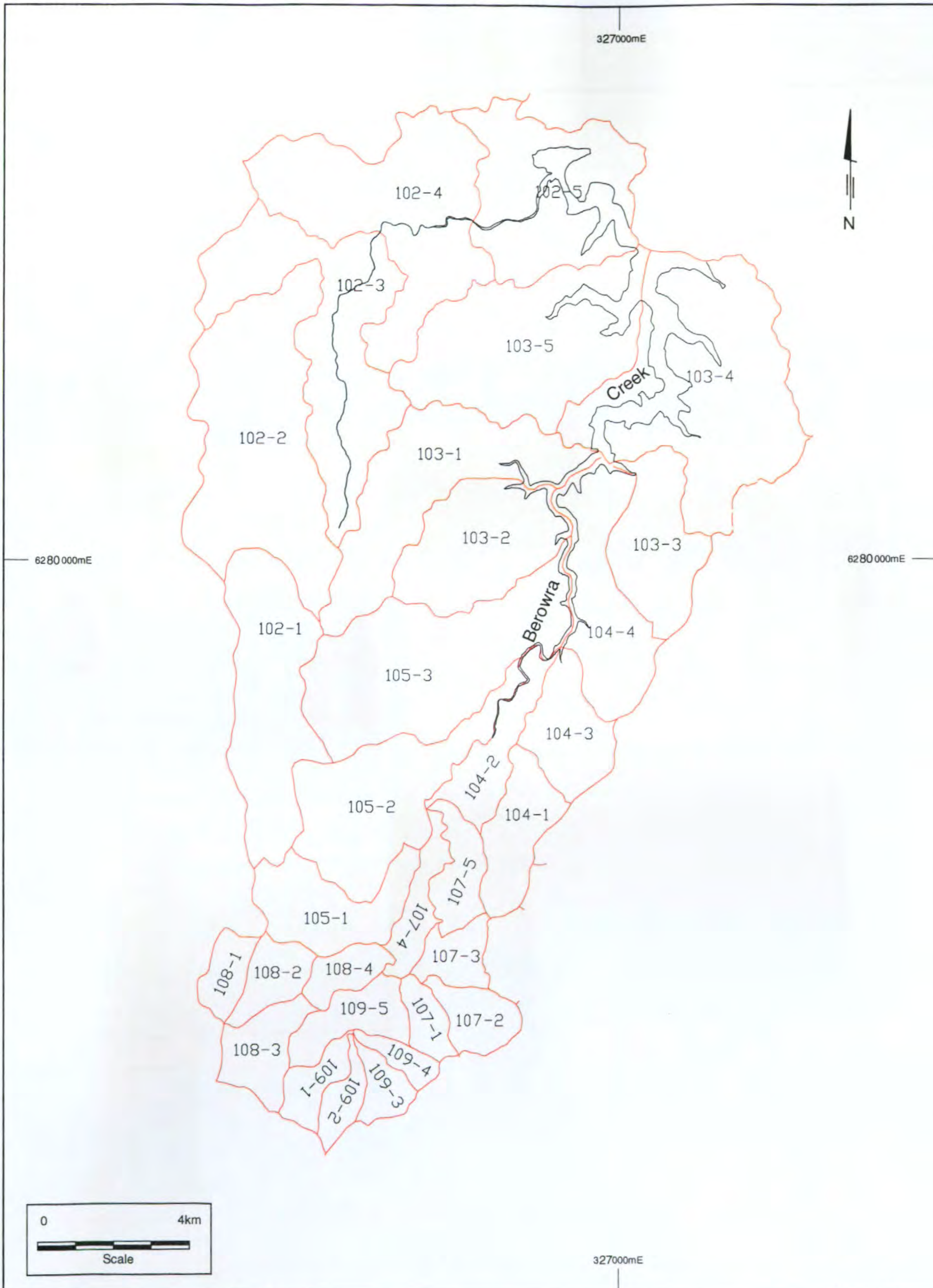
The values of the light limitation factor at Square Bay are smaller than at Berowra Ferry, which are lower than those at Crosslands, reflecting the differences in water depth.

2.3.4 Algal Group Succession

The smaller limitation of temperature on the growth of diatoms, compared with that on dinoflagellates, leads to a faster increase in diatom growth in the winter-spring period each year. This is reflected in the summer peak in diatom biomass occurring earlier than that in dinoflagellate biomass (AWT EnSight 1997). The earlier downturn in diatom biomass in the 1993-94 and 1994-95 summers is probably a result of diatoms requiring more phosphorus and nitrogen per unit increase in biomass than do dinoflagellates.

2.4 Summary of SALMON-Q

The SALMON-Q model simulates the algal blooms in Berowra Creek and was found to provide a reasonable match with the existing data. The algal blooms were found to be limited more by light availability, with algal succession occurring due to a temperature response. It appears that the nutrient concentrations only decrease to levels where growth limitation occurs on a sporadic basis. The effect of horizontal mixing and tidal flushing was not analysed although this does appear to be important for determining the spatial distribution.



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

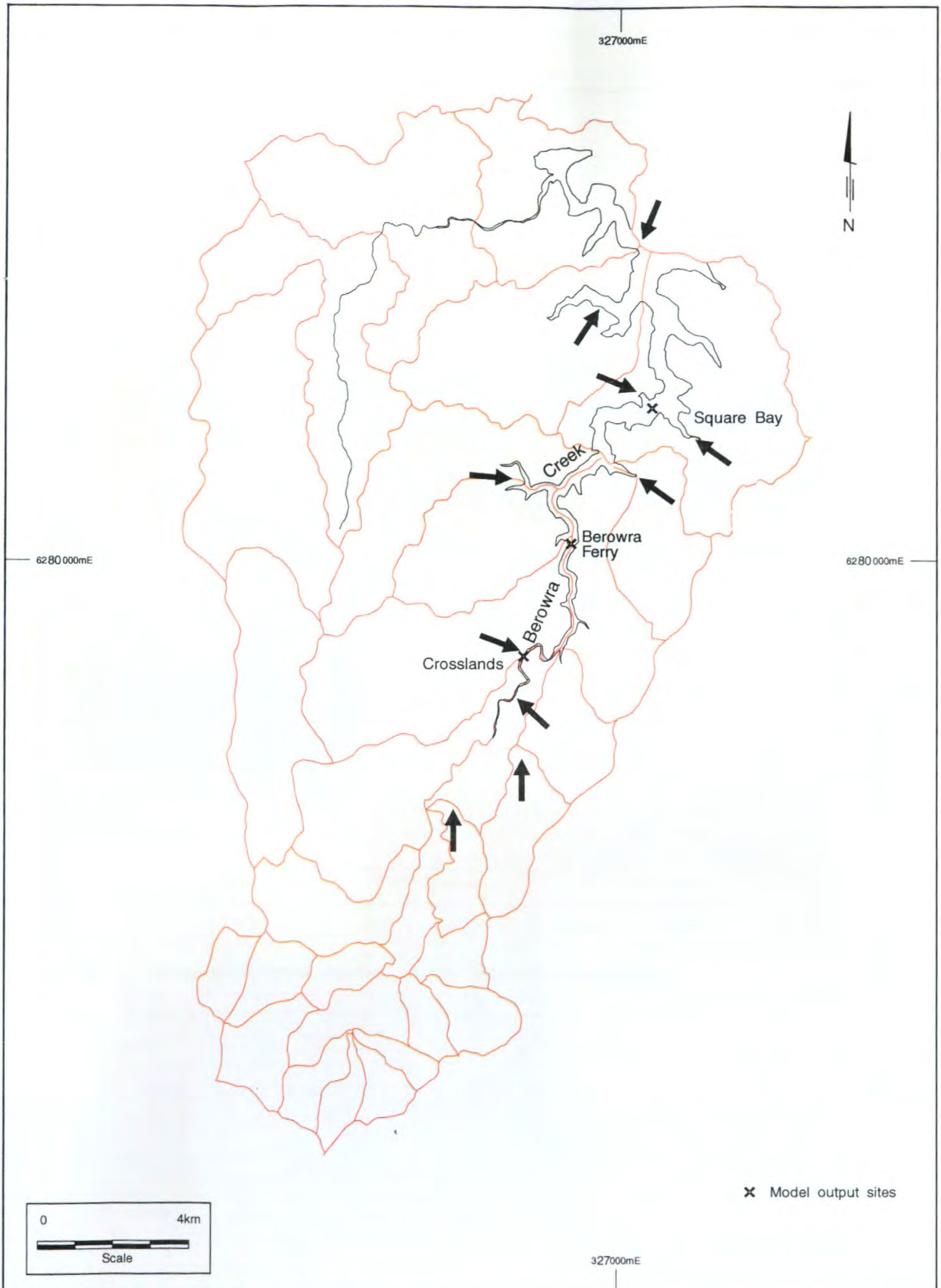
MANLY HYDRAULICS LABORATORY

BEROWRA CREEK SUB-CATCHMENTS

MHL Report 928

Figure 2.1

DRAWING 928BER21.CDR



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

MANLY HYDRAULICS LABORATORY

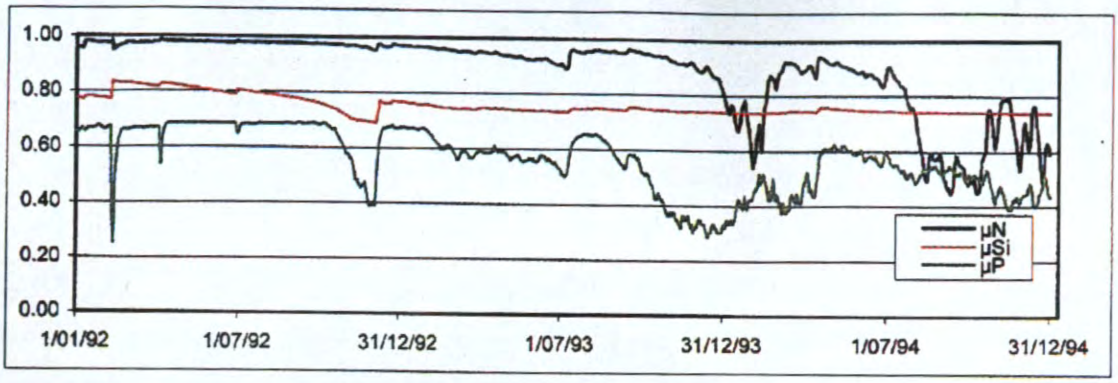
LOCATIONS OF MODELLED CATCHMENT INFLOWS TO THE ESTUARY

MHL Report 928

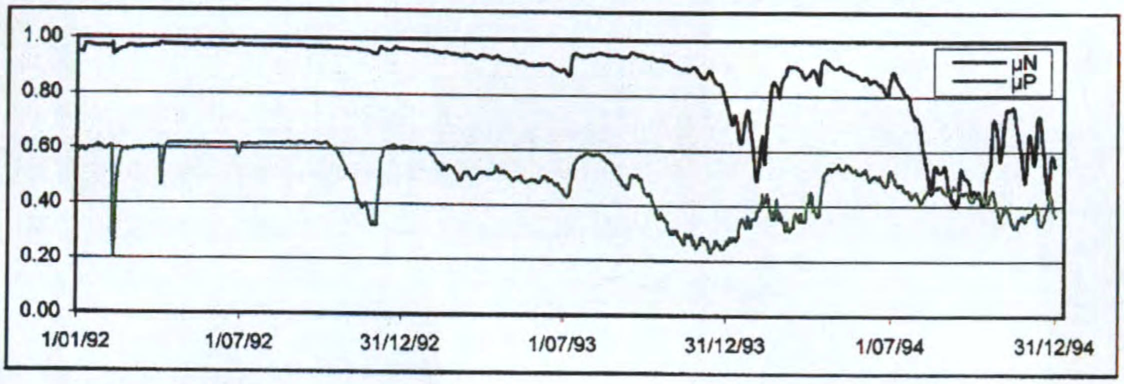
Figure 2.2

DRAWING 928BER22.CDR

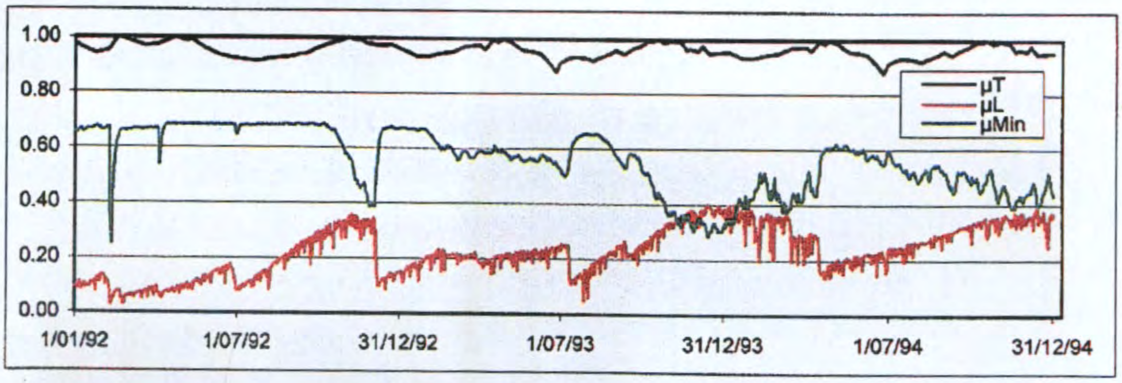
a) **Nutrient limitation factors for diatoms**



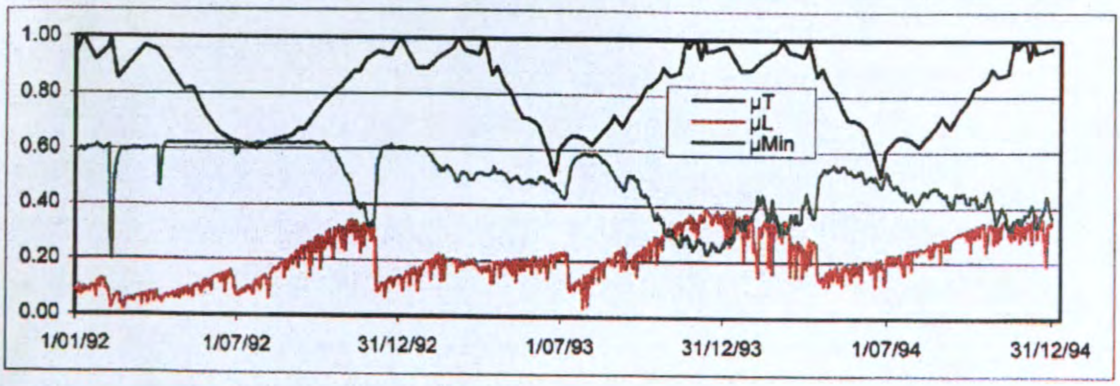
b) **Nutrient limitation factors for dinoflagellates**



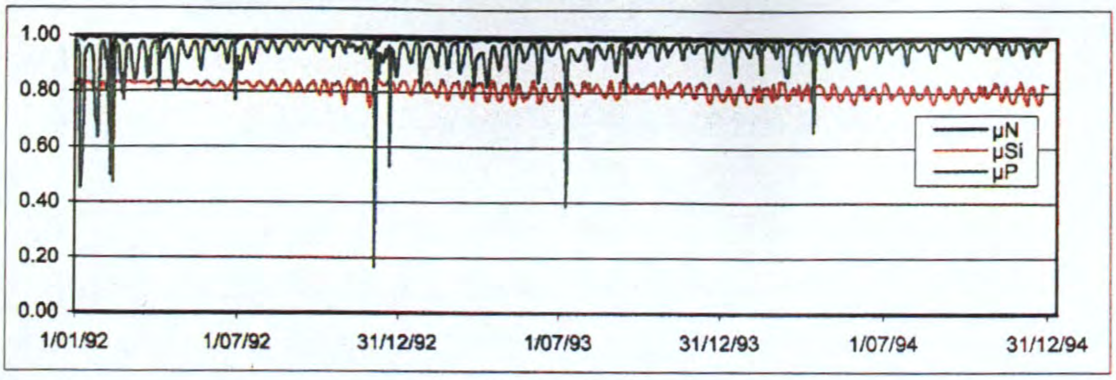
c) **Temperature, light & nutrient limitation factors for diatoms**



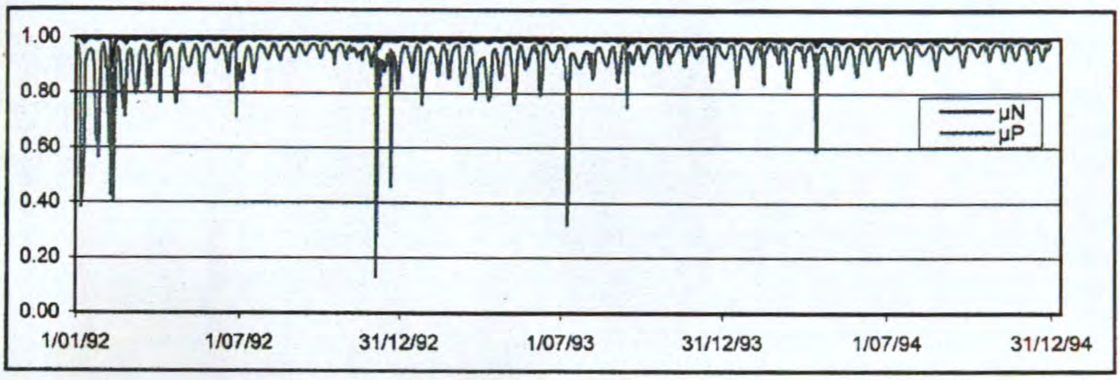
d) **Temperature, light & nutrient limitation factors for dinoflagellates**



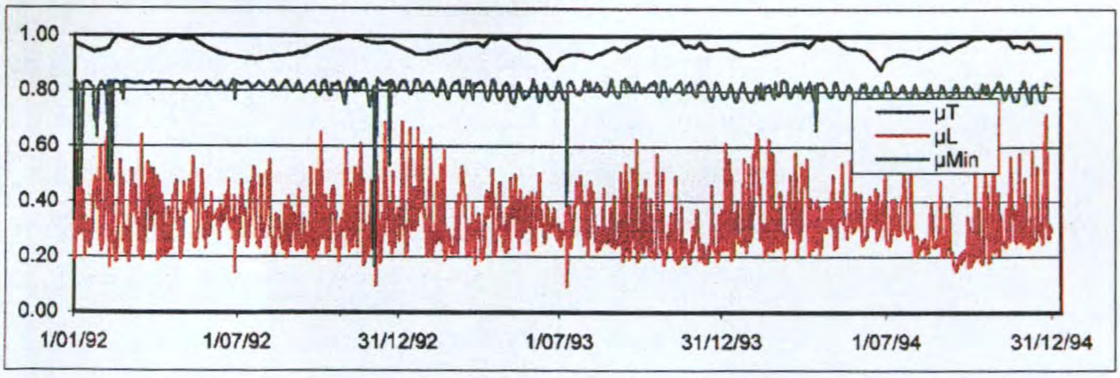
a) Nutrient limitation factors for diatoms



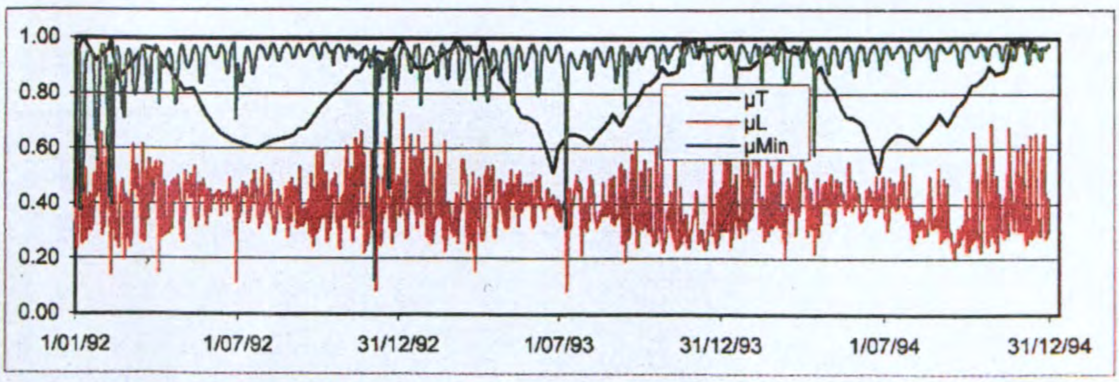
b) Nutrient limitation factors for dinoflagellates



c) Temperature, light & nutrient limitation factors for diatoms



d) Temperature, light & nutrient limitation factors for dinoflagellates



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	PHYTOPLANKTON GROWTH LIMITATION FACTORS AT CROSSLANDS	MHL Report 928 Figure 2.5 DRAWING 928BER25.CDR

3. Water Quality Surveys

3.1 Objectives

The data collection exercise carried out in Berowra Creek from November 1997 to February 1998 is reported in MHL Report No. 917 (MHL 1998a). The aim of these investigations was to determine spatial and temporal variations in water quality and hydrodynamics of the estuary. Data gathered consisted of:

- meteorological variables and water temperature over the three-month period;
- temperature, salinity, density, DO, pH, chlorophyll-a and PAR data collected in short-term intensive data collection exercises;
- water samples collected by Hornsby Shire Council which were analysed for chlorophyll-a, algal species and nutrients;
- water samples and plankton net tows collected by DLWC for phytoplankton and zooplankton analysis; and
- Sediment cores and water samples collected by the EPA for analyses of sediment nutrient release.

A summary of the extent of data collected is presented in Figure 3.1

3.2 Outline of Methodology

Two long-term data collection sites were established at strategic locations within the estuary, one being the site at which the thermistor chain was deployed and the other where the meteorological station was established at Berowra Marina. Figure 3.2 shows the location of these sites and the study area.

The thermistor chain was deployed in Berowra Creek near Cunio Point from 9 December 1997 to 16 February 1998. The chain consisted of 39 temperature sensors deployed in an MHL-designed floating mooring system which kept the sensors at a constant depth below the surface. The sensor depths ranged from 0.18 to 16.36 m below the surface.

A temporary meteorological station was established at Berowra Marina to ensure that meteorological data was relevant to the water data being collected. The station measured wind speed and direction, solar irradiance, humidity, air temperature and air pressure. Daily rainfall data recorded at the rainfall station in Goodwyn Road, Berowra, from 1 December to 28 February 1998 was obtained from the Bureau of Meteorology.

Simulated water levels at Berowra Marina were calculated from water levels measured at Patonga as no actual water level data was collected within the estuary during this data collection period.

A Sea-Bird Electronics SBE25-03 Sealogger CTD water quality profiler was used to measure profiles at selected locations throughout the estuary. Parameters measured included temperature, conductivity, depth, dissolved oxygen, chlorophyll-a and PAR (photosynthetically active radiation).

Personnel from HSC collected water samples for chlorophyll-a, algal counts and nutrient analysis. Water quality profiles were also measured with a YeoKal 611 intelligent water quality analyser. Samples were collected from two sites within the study area and were taken at 1 m below the surface for chlorophyll-a and algal species analysis, and 1 m below the surface and near the bed for nutrient analysis.

Personnel from DLWC collected water samples for analysis of zooplankton and phytoplankton and chlorophyll-a. Plankton sampling took place during eight cruises on the estuary during December 1997 and February 1998. Three sites within the estuary were sampled on each occasion, these were upstream of the phytoplankton bloom location, within the phytoplankton bloom and downstream of the bloom. Three replicate tows were undertaken at each site. Sampling also consisted of three 120ml algal samples collected at random at each station for counting and identification. Chlorophyll-a biomass data was provided from the Seabird CTD profiler.

Detailed descriptions of the methodologies used for each of the variables measured are provided in MHL Report No. 917 (MHL 1998a).

3.3 Weather, Thermistor Chain and Water Level Results

3.3.1 Rainfall

Rainfall was generally low, although on six days significant rainfall did occur: 20 December 1997, 6, 21, 25, 26 January and 8 February 1998. As a consequence of the lack of significant rainfall during the study period fresh water inflows occurred only intermittently and for short duration's. Rainfall and meteorological data for the week 8-14 February are shown in Figure 3.3.

3.3.2 Wind

Throughout most of the study period, a fairly predictable diurnal pattern of land and sea breeze winds occurred, with early morning winds from the west (offshore) changing to the north-easterly sea breeze (onshore) as the day progressed. The sea breeze wind was sustained into the evening, turning towards the west again in the late evening.

Given this pattern of wind directions, and the north-south orientation of Berowra Creek, wind mixing events in the estuary are expected to be short-lived and to penetrate only the near-surface layer. This is due to the limited fetch lengths available and sheltering by the high-ridged topography of the surrounding land forms.

One area where wind mixing may be more important is around Flat Rock (Figure 3.2) where the estuary has an east-west orientation for approximately two kilometres and winds are funnelled along this reach.

3.3.3 Solar Radiation

Solar radiation showed diurnal fluctuations varying between 0 and 1,800 W/m^2 as expected at the latitude of the study area for a summer period. Cloudy days result in significant reduction to the light reaching the water surface (Figure 3.3). The amount of light penetrating the water surface largely determines the growth of photosynthesising plants. The heating of the near-surface waters also creates stratification.

3.3.4 Air Temperature

Air temperature varied between 11°C and 40°C over the study period. Significant cool periods occurred on 16, 17, 18 and 29 December, 26 and 27 January, and 18 and 17 February. Air temperature is compared to the near-surface and near-bottom water temperature in Figure 3.3. Solar heating and warm air temperatures generate stratification in the water as can be seen on 12 February 1998 (Figure 3.3) when the surface water became warmer than the bottom water. Overnight in the absence of solar heating the water surface cools due to heat transfer across the air-water interface.

3.3.5 Tides

The tides in Berowra Creek have been discussed in detail in MHL Report No. 745 (1997) and reviewed in MHL Report No. 855 (MHL 1998b). In summary, tides are predominantly semi-diurnal with a mean spring range at Berowra Ferry of 1.4 m. The tidal prism for the estuary is around 13 x 10^{6}m^3 and upstream of Berowra Ferry the tidal prism reduces to approximately 1.6 x 10^{6}m^3.

Water levels for the period were obtained from the Patonga data as no water level gauge was installed in Berowra Creek. The Patonga data were transformed to a location representing Berowra Ferry by subtracting one hour from the time at Patonga and amplifying by a factor of 1.097 (Figure 3.3).

3.3.6 Thermistor String Temperature Data

The thermistor chain data are presented in the water quality data report (MHL 1998a). Temperature isotherms over the deployment period show the effects of heating and cooling of the near-surface waters, a gradual warming of the bottom waters from 24°C in early December 1997 to 27°C by mid-February 1998 (Figure 3.4). A warm sunny period between 19 and 25 December resulted in significant surface heating and development of thermal stratification down to about 5 m depth over five days. A cold snap on 27 and 28 December destroyed the stratification and the whole water column became isothermal on 29 December (Figure 3.5). Similar events occurred between 5 and 10 January 1998, on 17 January and 10 February 1998.

The isothermal conditions do not necessarily reflect complete mixing of the water column because mixing is controlled by water density that is affected by both temperature and salinity. The salinity data are discussed below.

The thermistor string data indicate a strong response of the estuary to both meteorological and tidal forcing. Stratification develops in the near surface due to solar heating (Figure 3.5, 30/12/97) and fresh water inflows (Figure 3.7, 26/01/98). Mixing due to cooling overnight and tidal flows effectively breaks down the stratification. Note that the oscillations in the deeper waters are tidal while in surface waters diurnal heating and cooling cycles are more prevalent (Figure 3.5). This cycling in the stratification has important implications for the water quality and algal blooms in the system.

3.4 Freshwater Inflows

Freshwater inflows contribute to the water quality by addition of nutrients to the system and by contributing to the stratification. Discharge data were collected on Pyes, Tunks and Waitara creeks by AWT Ensight in 1995 (AWT 1995). The data indicate a long tail of the hydrograph following major rainfall events (Figure 3.8). The fresh water inflow to the system was estimated from the rational method (*Australian Rainfall and Runoff* 1987) for the past summer and nutrient loads (Section 4) estimated using the derived flows.

3.5 Salinity

The salinity data provide a useful means for assessing the flushing and mixing characteristics of the system. Depth-averaged salinity at high tide along the estuary on five different dates is shown in Figure 3.9.

The typical estuarine gradient from fresh low salinity upstream to saltier oceanic water near the confluence with the Hawkesbury River fluctuates according to the fresh water inflow. The salt wedge character is also typical, as demonstrated by the salinity contours of 15 December 1998 (Figure 3.10).

Vertical salinity profiles collected in the deep hole near Calabash Point are presented in Figure 3.11. Between 2 and 15 December the salinity increased by approximately 1 psu over the whole water column but the stratification remained. The salinity increases at 14 m and 15.5 m on the profile from 2 December indicate recent inflows of different water masses to the deeper regions. Between these two dates the water in the deep holes has been completely replaced by the water from downstream.

By contrast, between 10 and 16 February 1998, the profiles indicate very little change in the stratification suggesting the water in the deep holes is not well flushed. This observation is supported by the dissolved oxygen data in the deep waters that decreased over the six days.

3.6 Nutrients

Hornsby Shire Council personnel collected water samples and measured water quality profiles at two sites in Berowra Creek on 18 occasions between 3 November 1997 and 13 February 1998. The first site was in 7 m water depth 50 m downstream from Berowra Ferry and the second site was at the 17 m deep hole off Calabash Point. These two sites are referred to as Berowra Ferry and Calabash Point. The samples analysed for nutrients were collected 1 m below the surface and 1 m above the bed at each site.

Nutrient analysis was undertaken by Analytical Services AWT EnSight. Water samples were analysed for suspended solids, filtered total phosphorus, filtered oxidised nitrogen, ammonia nitrogen, total nitrogen, total phosphorus and chlorophyll-a. The bioavailable nutrients filtered total phosphorus and dissolved inorganic nitrogen (addition of ammonia and oxidised nitrogen concentrations) are shown in Figure 3.12 along with chlorophyll-a.

Half saturation constants used in the SALMON-Q model - nitrogen 0.06 mg/L and phosphorus 0.004 mg/L - are generally exceeded except in the surface waters at Calabash Point which is generally just downstream of the bloom peak chlorophyll concentration. The higher values of the surface concentrations at Berowra Ferry are consistent with the uptake of nutrients by phytoplankton as they move downstream.

In the deeper waters fluctuations at Calabash Point are consistent with nutrient release during stratified periods leading to a gradual build up of nutrients in the deeper layers. Mixing events then cause a decrease in deep water concentrations due to dilution with lower concentration near surface waters. Chlorophyll-a concentrations decrease rapidly below the photic zone at around 3 m depth.

3.7 Light Attenuation

Light attenuation is calculated from the underwater light profiles collected by the Seabird CTD. The light attenuation coefficient is estimated from the vertical profiles by applying linear regression to the log transformed light data.

Attenuation coefficients and photic depth derived from the profiles collected on 2 December are shown in Figure 3.13.

The photic depth is defined as the depth at which light intensity falls to 1% of the surface light intensity and is a rough measure of zone within which photosynthesis may occur. In the case of phytoplankton that can only utilise light above a critical level, the zone within which they can actively utilise light is generally much less than the photic depth. As an example, assuming a critical light intensity of 200 $\mu E/m^2/s$ then the depth at which this occurs is also shown in Figure 3.13. In the upstream reaches the clearer waters will support phytoplankton production down to about 2 m while downstream of the deep holes the depth decreases to around 1 m. The SALMON-Q model light limitation factors vary between 180 $\mu E/m^2/s$ for diatoms and 270 $\mu E/m^2/s$ for dinoflaggelates. The dinoflaggelates require more light and hence their depth range will be more limited than diatoms.

3.8 Chlorophyll-a and Dissolved Oxygen

A Seabird CTD Recorder, fitted with a Wet Star fluorometer, was used to collect water quality data and chlorophyll-a biomass data. Random samples were also collected throughout each cruise to assess chlorophyll-a concentrations.

Biomass

Surface phytoplankton biomass was higher between stations 8-10 on each of the eight cruises (Figure 3.14). There was however, substantial temporal variability, with biomass either declining or increasing. Very high levels (45 µg/L) of chlorophyll-a were measured in the bloom area on 2 December. Concentrations were lower in the bloom area on 10 December (18 µg/L) and increased to 35 µg/L by 15 December.

Phytoplankton biomass in the bloom area was high between 6 and 12 February (45 to 55 µg/L respectively, Figure 3.14). There was a significant decline in chlorophyll a concentration (to 30 µg/L) in the bloom area. A decline in biomass was again apparent on 16 February (µg/L).

There was a significant relationship between algal biomass and total algal counts ($r=0.8$, $n=24$, $p<0.05$). Counts ranged from between 286 cells/mL at the downstream locations to 16000 cells/mL at the bloom stations.

Vertical profiles of chlorophyll-a at the deep hole near Cunio Point are shown in Figure 3.15.

3.9 Phytoplankton

Sampling of phytoplankton by DLWC personnel took place during eight cruises on the estuary during December 1997 and February 1998. There were three sampling sites within the estuary, at Arcadia (upstream of the phytoplankton bloom), Calabash (phytoplankton bloom) and Kimmerikong (downstream of the bloom). Sampling took place during daylight at low tide slack to prevent contamination of samples with organisms which may have been transported upstream during high tides from the Hawkesbury River.

Plankton were captured using a 100 µm mesh net with a 20 cm diameter opening towed at the surface at approximately 1.5 m/s. The average volume filtered during each tow was 2.5 m^3.

The methodology is discussed in more detail in Appendix A.

DLWC data

Multi-dimensional scaling (MDS) analysis of the square root transformed abundance of phytoplankton taxa distinguish clearly between the December and February sampling periods (Figure 3.16). In addition phytoplankton communities were shown to be different between upstream, bloom and downstream study sites.

The major groups contributing to the differences between the December and February sampling regimes were *Chaetoceros* which dominated samples during December and a species of each of *Pseudonitzschia* and *Thalassiosira* which were the two most dominant taxa present during February.

Figure 3.17 shows the changes in abundance of the dominant taxa that occurred in the bloom over the sampling period. This emphasises the dominance of the phytoplankton community by a single taxon during December and by two taxa during February. The change in dominance of the phytoplankton community from a single taxon to two taxa can be seen when assessing species diversity over the study period. Shannon diversity gives an indication of the contribution of taxa to the composition of the community. Diversity increases in relation to the number of individuals of each taxon in the community. During December, *Chaetoceros* dominated the bloom (65%-90%). Between 14 and 19 taxa were present at this time, but a low diversity was calculated (Figure 3.18) because of the dominance of a single taxon. The greater diversity during February reflects the dominance of the community by more than a single taxon.

Hornsby Shire Council Data
Analysis of the community structure of phytoplankton over the sampling period showed a similar trend to that from the bloom site data collected by DLWC. Data from the ferry site and the Calabash site had a similar community structure (Figure 3.19), indicating that despite spatial variability of phytoplankton biomass in the general area of the bloom shown elsewhere in this document, there was not much spatial variability in the general bloom area. The dominant groups of phytoplankton identified at each of the two sites were diatoms.

Data from the Ferry and Calabash sites were analysed separately, and provided evidence of a shift in the dominant taxa over time (Figures 3.20 and 3.21). Major taxonomic shifts were found between the November-December period and the January-February period. Similarity percentages analysis indicated that the greatest similarity between the November-December group was due to the presence of *Chaetoceros* sp. (40%) and between the January-February group was due to the presence of *Thalassiosira* sp. (33%). The dissimilarity between the two groups was the result of *Chaetoceros* (10%), *Nitzschia pungens* (10%), *Skeletonema costatum* (9%) and *Thalassiosira* sp. (8%). Shifts in the major taxa are indicated more clearly in Figure 3.22.

3.10 Zooplankton

Sampling of zooplankton by DLWC personnel took place during eight cruises on the estuary during December 1997 and February 1998. There were three sampling sites within the estuary, at Arcadia (upstream of the phytoplankton bloom), Calabash (phytoplankton bloom) and Kimmerikong (downstream of the bloom). Sampling took place during daylight at low tide slack to prevent contamination of samples with organisms which may have been transported upstream during high tides from the Hawkesbury River.

Plankton were captured using a 100 µm mesh net with a 20 cm diameter opening towed at the surface at approximately 1.5 m/s. The average volume filtered during each tow was 2.5 m^3.

The methodology is discussed in more detail in Appendix A.

Zooplankton samples were analysed in a laboratory. Such analysis established zooplankton biomass, diversity of community, size of plankton/particles and diversity and percentage composition of each community.

Results

Zooplankton / phytoplankton biomass dynamics December 1997.

Twenty-two taxa of zooplankton were identified during the study period. Dominant taxa are shown in Figure 3.23 and Table 3.1. *Oithona*, a cyclopoid copepod and copepod nauplii were the most abundant of all taxa throughout the study period.

Table 3.1 Numerically Dominant Groups Used to Characterise Size Spectra Indicated by Image Analysis

Taxa			Species	Size range (esd, µm)
Chaetognatha			*Sagitta*	381-818
Crustacea				
	Copepoda			
		Calanoida	*Acartia bispinosa*	346-557
			Paracalanus Sp.	196-303
			Gippslandia estuarina	172-287
		Cyclopoida	*Oithona Sp.*	129-275
		Harpacticoida	*Euterpina acutifrons*	164-412
nauplii				91-166
	Decapoda			
		Brachyuridae	zoeae larvae	401-742
Tintinnidae				101-187
Medusae				381-818

Inspection of the zooplankton size spectra classifications within the bloom during December and February indicated that the zooplankton community was generally similar over the entire study period (Figure 3.23). Some changes were evident however, with *Paracalanus* sp. not being present during February and *Acartia bispinosa* not being present during December.

Integrated data over the study period indicated that there was a significantly greater phytoplankton biomass (30 µg/L ± 5) in the bloom area, than at the statistically similar, upstream (10 µg/L) and downstream (11 µg/L) sites. Similar trends were found with zooplankton biomass integrated over the period, with an average of 200 mg/m³ in the bloom being significantly more than 140 mg/m³ at the upstream station and 120 mg/m³ at the downstream station (Figure 3.24a).

A significant correlation ($r=0.51$, $n=24$, $p<0.05$) between phytoplankton and zooplankton biomass at each sampling occasion and each site was found.

Comparison of data from the upstream station over the study period indicated that phytoplankton and zooplankton biomass was highly variable. At the upstream station, phytoplankton ranged between 4 and 15 µg/L. The greatest range between sampling dates was between 2 and 10 December and between 10 and 15 December. During this stage zooplankton biomass declined from 150 to 120 mg/m³. These data suggested that there had not been sufficient time for zooplankton biomass to increase between 10 and 15 December. During February, data indicated that phytoplankton biomass was lower when zooplankton biomass was higher and phytoplankton biomass was higher when zooplankton biomass was lower (Figure 3.24b).

Similar results were obtained at the downstream station, see Figure 3.24c, where phytoplankton biomass was similar to the upstream station ranging between 3 and 15 µg/L.

There appears to have been a lag in the response of zooplankton to increased phytoplankton biomass and a lag in the response of phytoplankton to increasing zooplankton biomass.

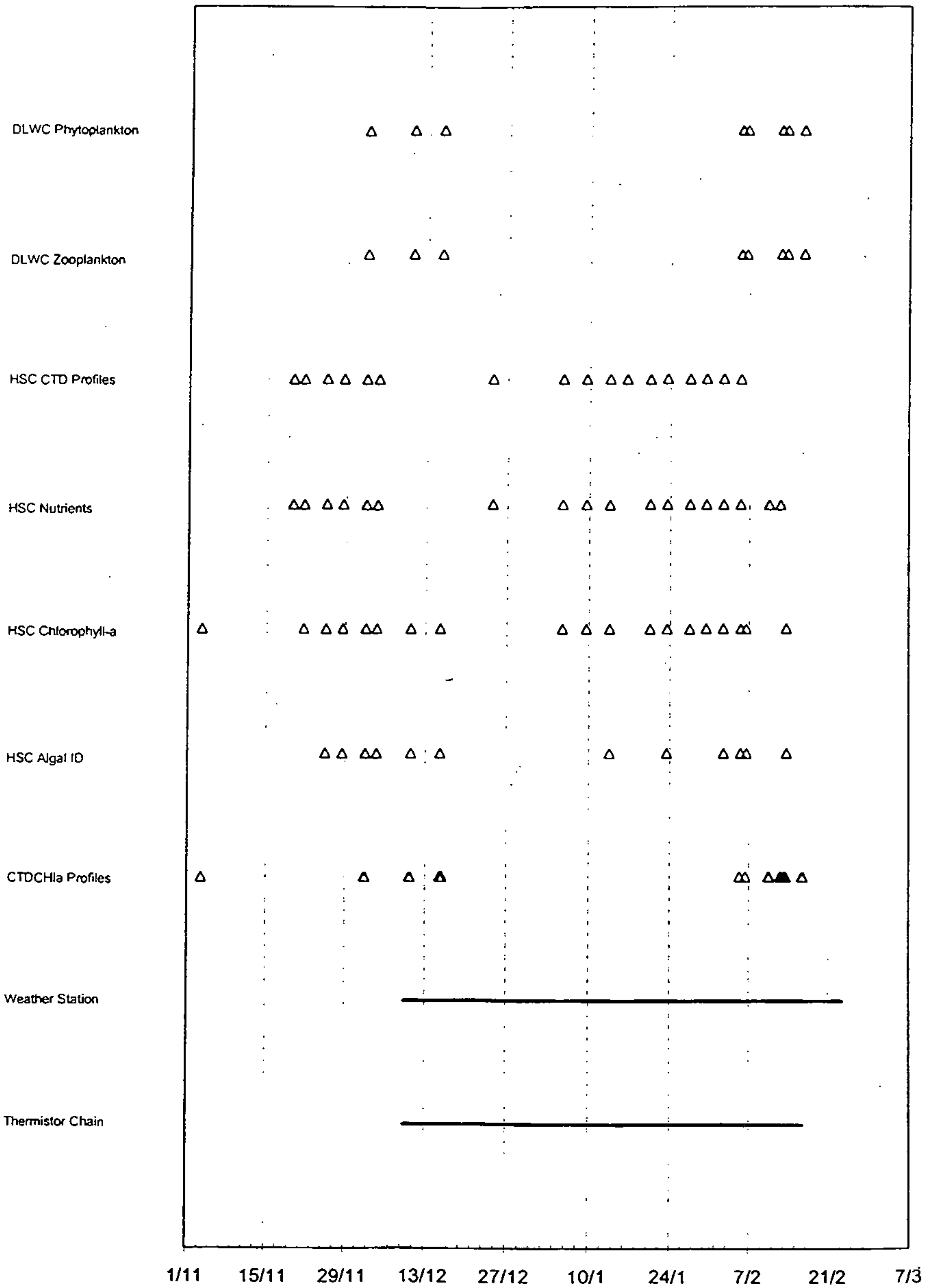
Similar trends were evident at the bloom station, however the range of phytoplankton biomass and zooplankton biomass over time was greater (Figure 3.25a). There appeared to be a three to five day (72-120 hour) lag between any increase in the zooplankton biomass following an increase in phytoplankton biomass. A similar lag time was apparent in the decline in phytoplankton biomass following the increase in zooplankton biomass.

Small and medium sized zooplankton

The change in zooplankton biomass in response to variation in phytoplankton biomass was investigated further using counts of particle size classes. The smallest four particle size classes and the four size classes above this were each pooled and were termed small and medium zooplankton respectively. Only the eight smallest particle size classes were used because small zooplankton taxa and small zooplankton reproductive stages show the most rapid response to environmental variability (Bays and Crisman 1983, Uye 1994, Rissik et al. 1997).

From 2 to 10 December small zooplankton declined rapidly in response to decreased phytoplankton biomass (Figure 3.25b). Over the same period there was an increase in the number of medium particles. Between 10 and 15 December, the number of small particles increased and there was a corresponding decrease in medium particles. At the end of January and over the February sampling period, phytoplankton biomass increased rapidly from 38 µg/L to 60 µg/L before declining steadily to 12 µg/L. Over this time, zooplankton biomass increased after an initial lag period and then declined. Small particles increased from 5 to 6 February and then increased and declined over subsequent sampling occasions, with average number remaining constant over the period. The medium zooplankton increased over the study period.

The zooplankton development appears to log the phytoplankton development and attains significant populations capable of exerting pressure on the phytoplankton bloom. This bloom-crash cycle is typical of these communities. There are not sufficient data, however, to quantify the detailed grazing rates and hence the implications for the phytoplankton.

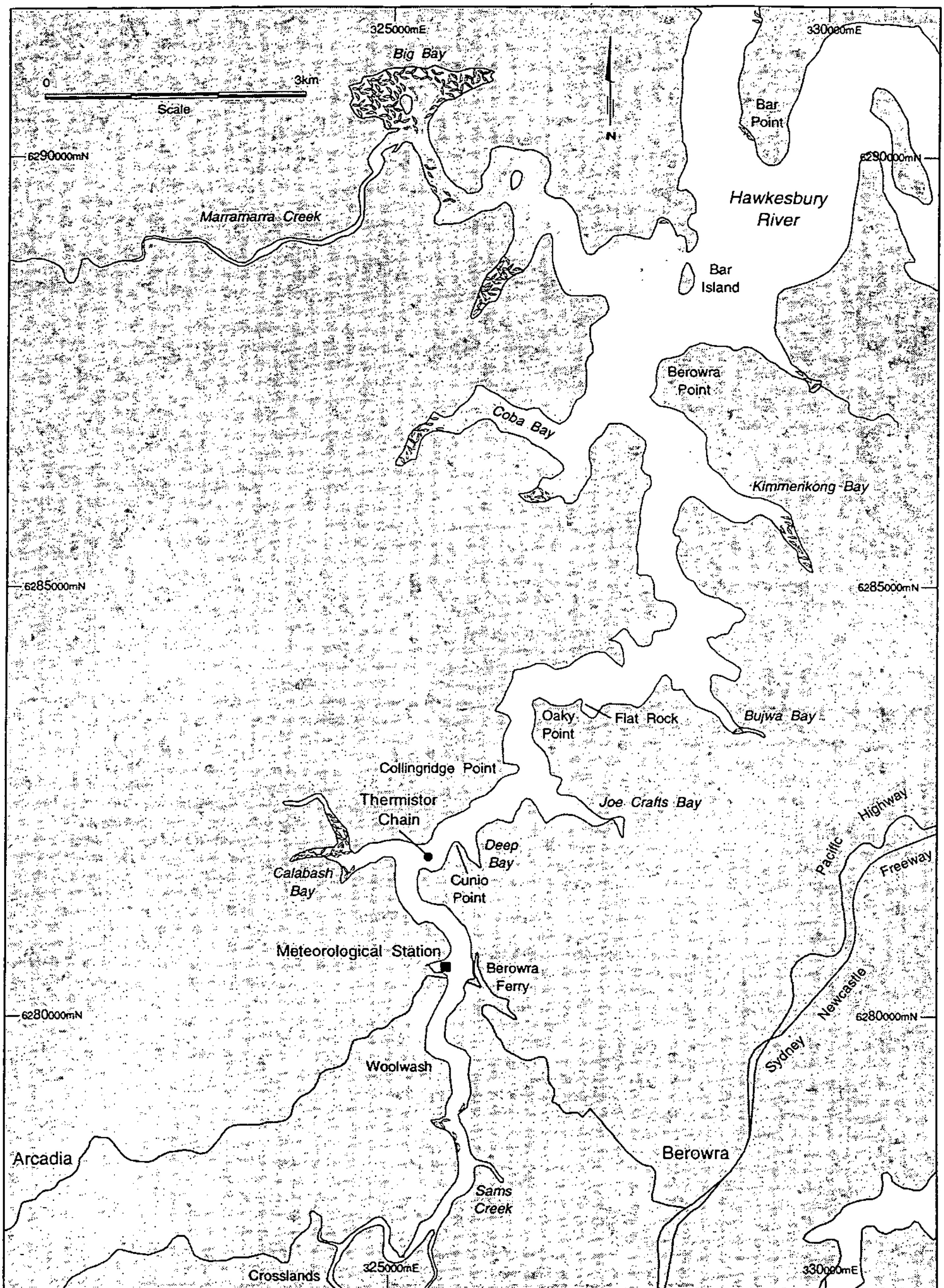


NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

SUMMARY OF DATA COLLECTED - BEROWRA CREEK
NOVEMBER 1997 TO FEBRUARY 1998

MHL Report 928
Figure 3.1

DRAWING 928_3_01.DRW



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

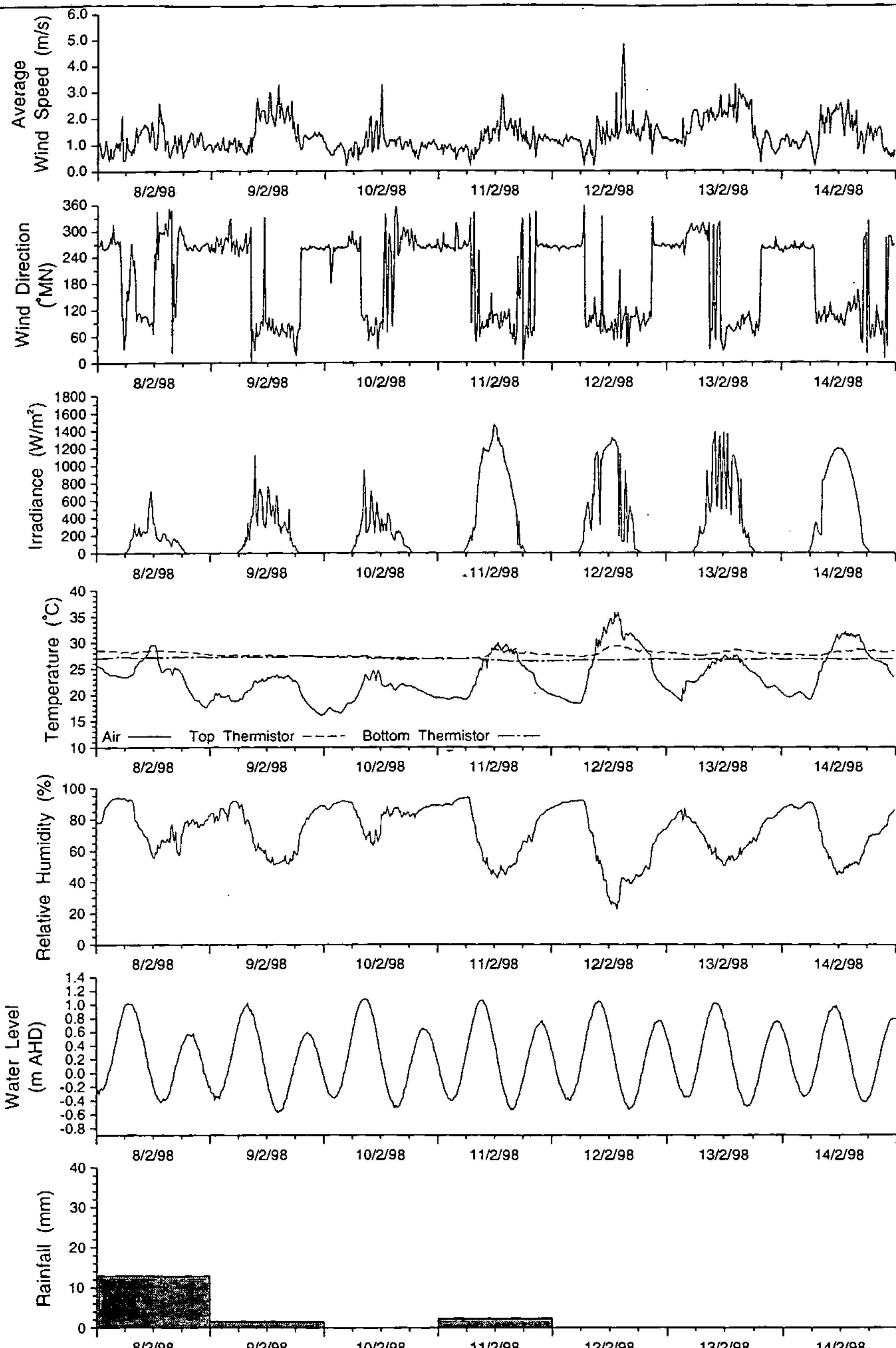
MANLY HYDRAULICS LABORATORY

DETAILED LOCATION MAP OF THE STUDY AREA

MHL Report 928

Figure 3.2

DRAWING 932_3_02.DRW



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

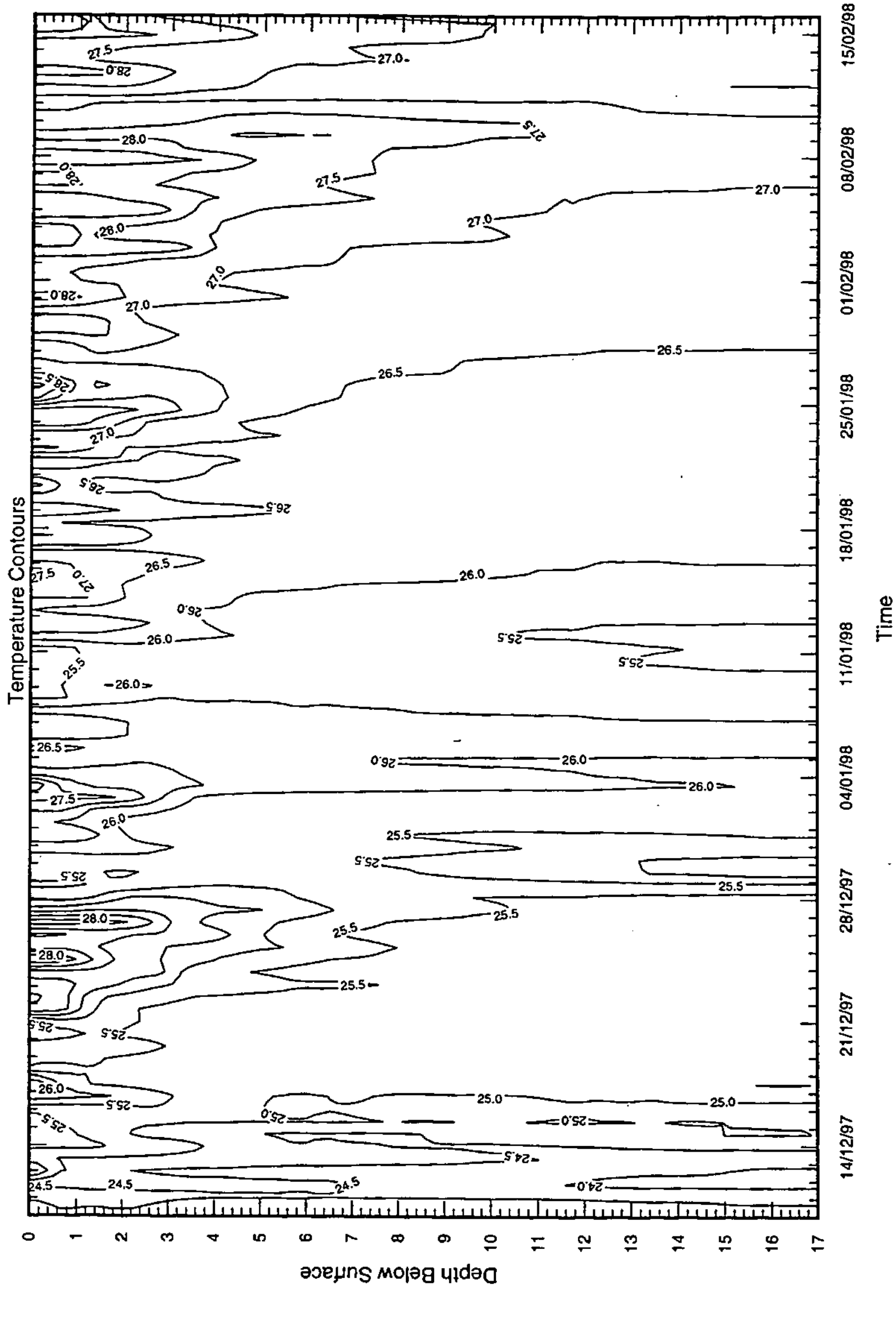
MANLY HYDRAULICS LABORATORY

SUMMARY OF AVAILABLE ENVIRONMENTAL DATA
8 TO 14 FEBRUARY 1998

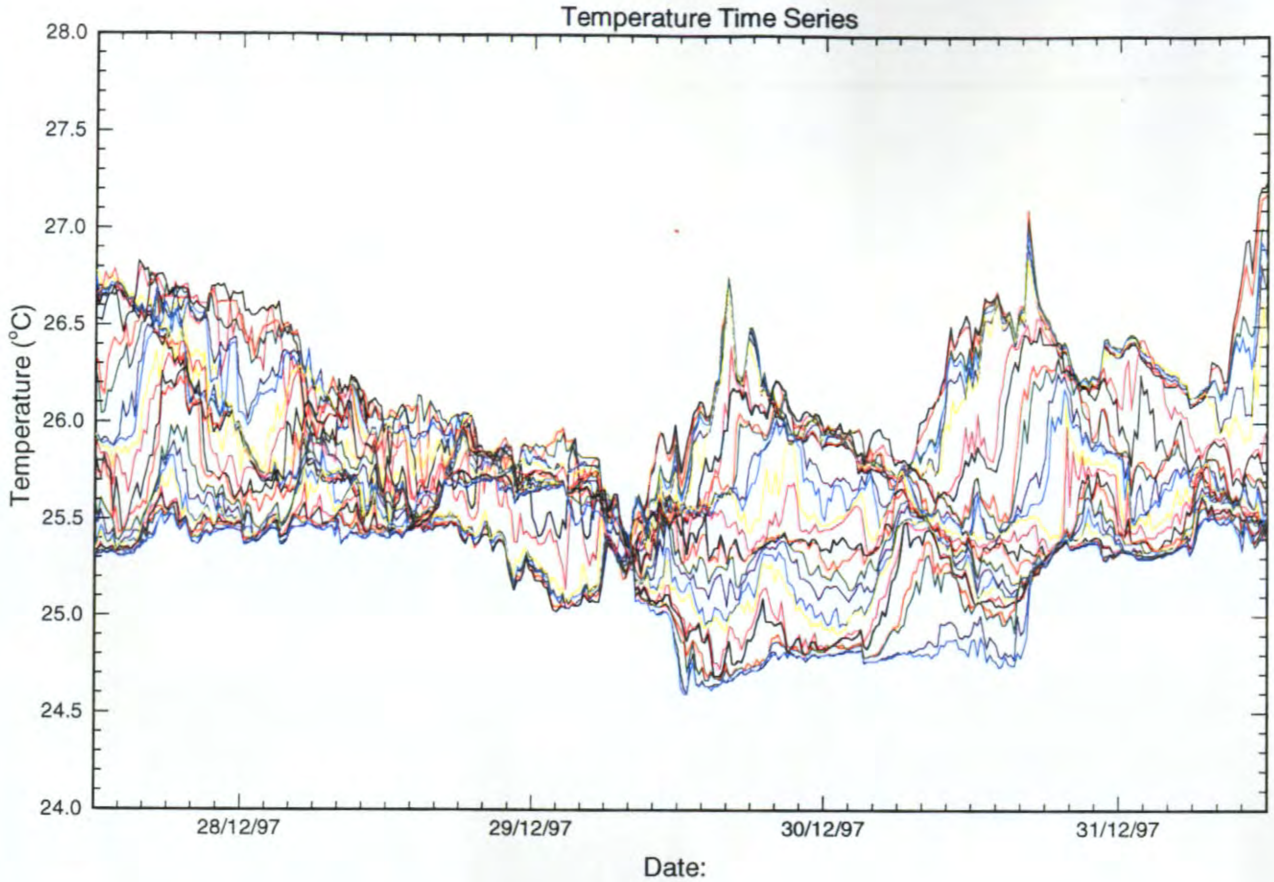
MHL Report 928

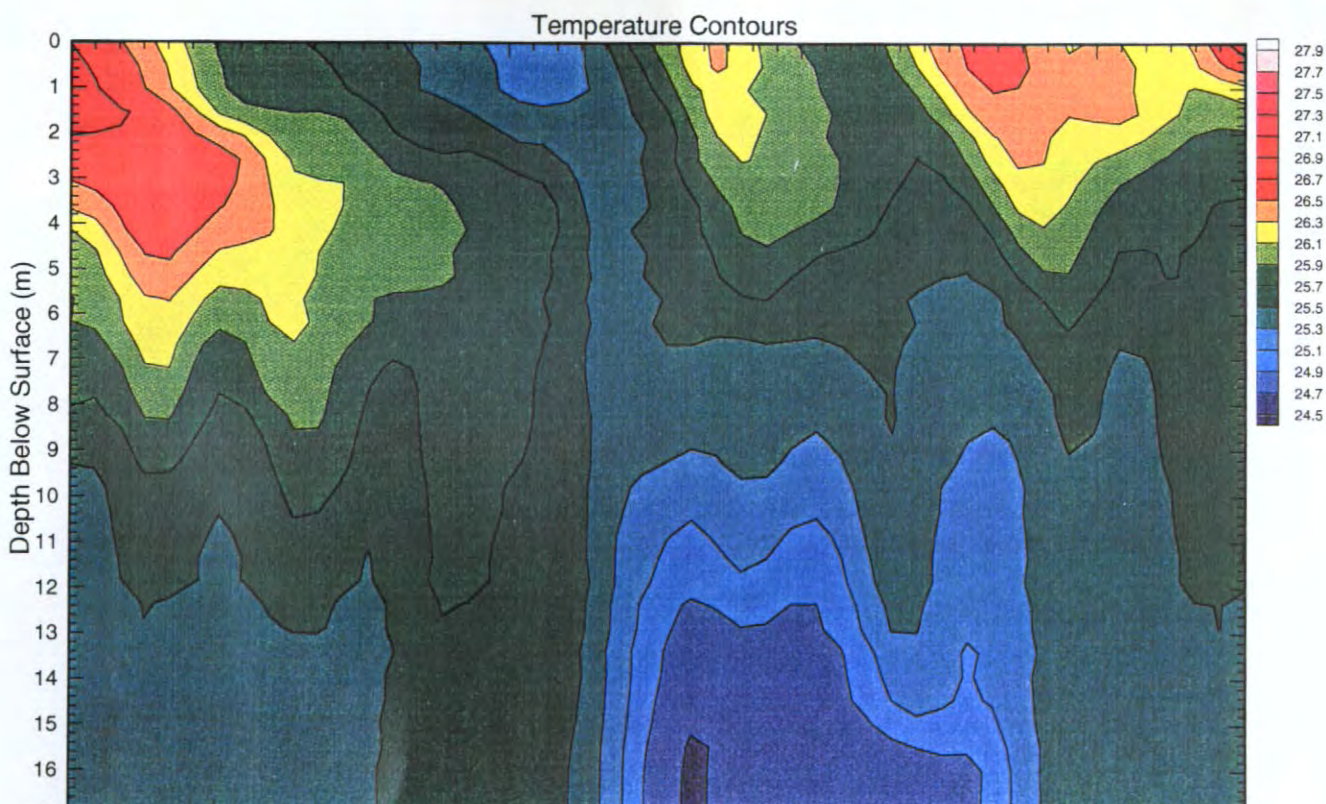
Figure 3.3

DRAWING 928_3_03.DRW

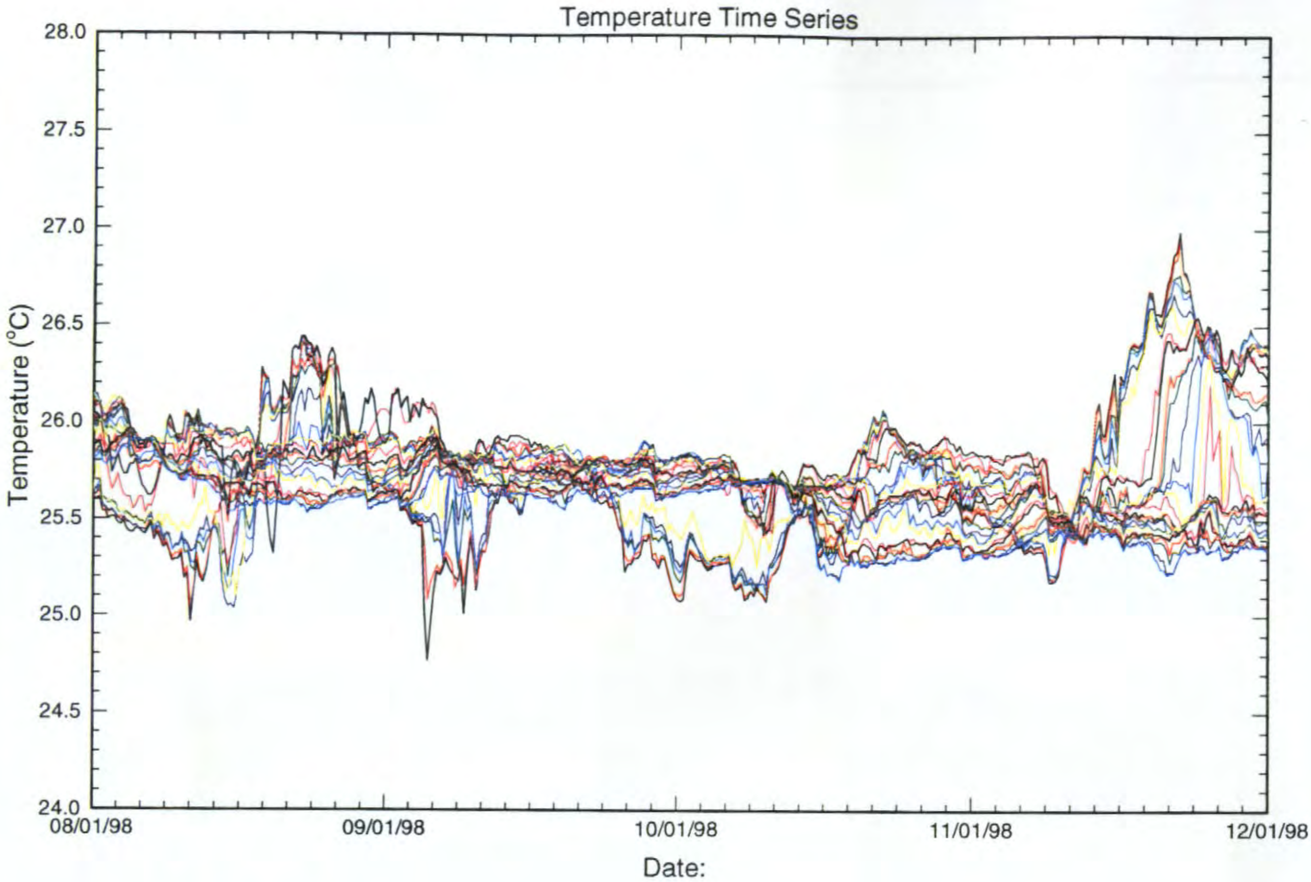


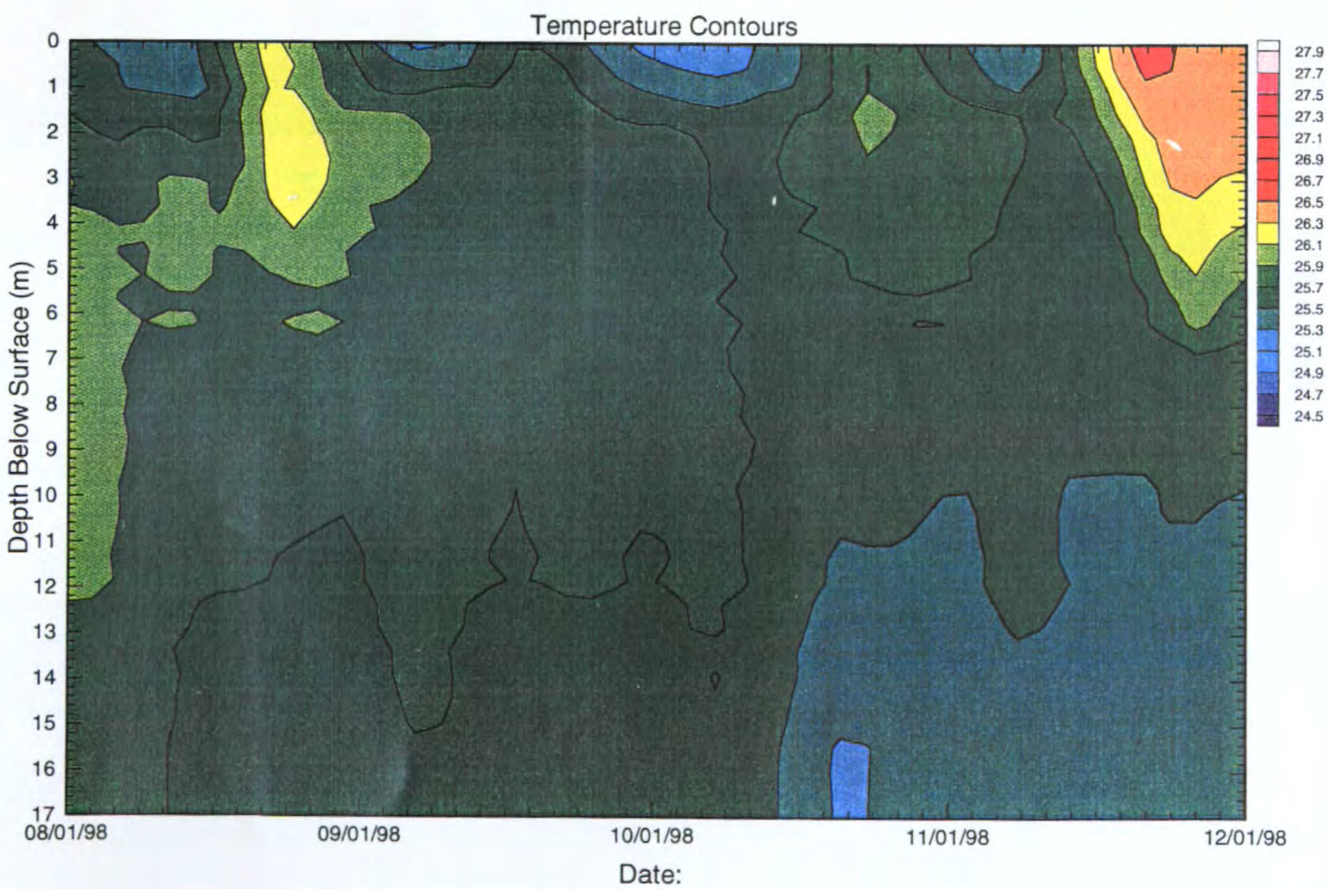
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	BEROWRA CREEK - CUNIO POINT TEMPERATURE CONTOURS 10 DECEMBER 1997 TO 16 FEBRUARY 1998	MHL Report 928 Figure 3.4 DRAWING TC4.BPS



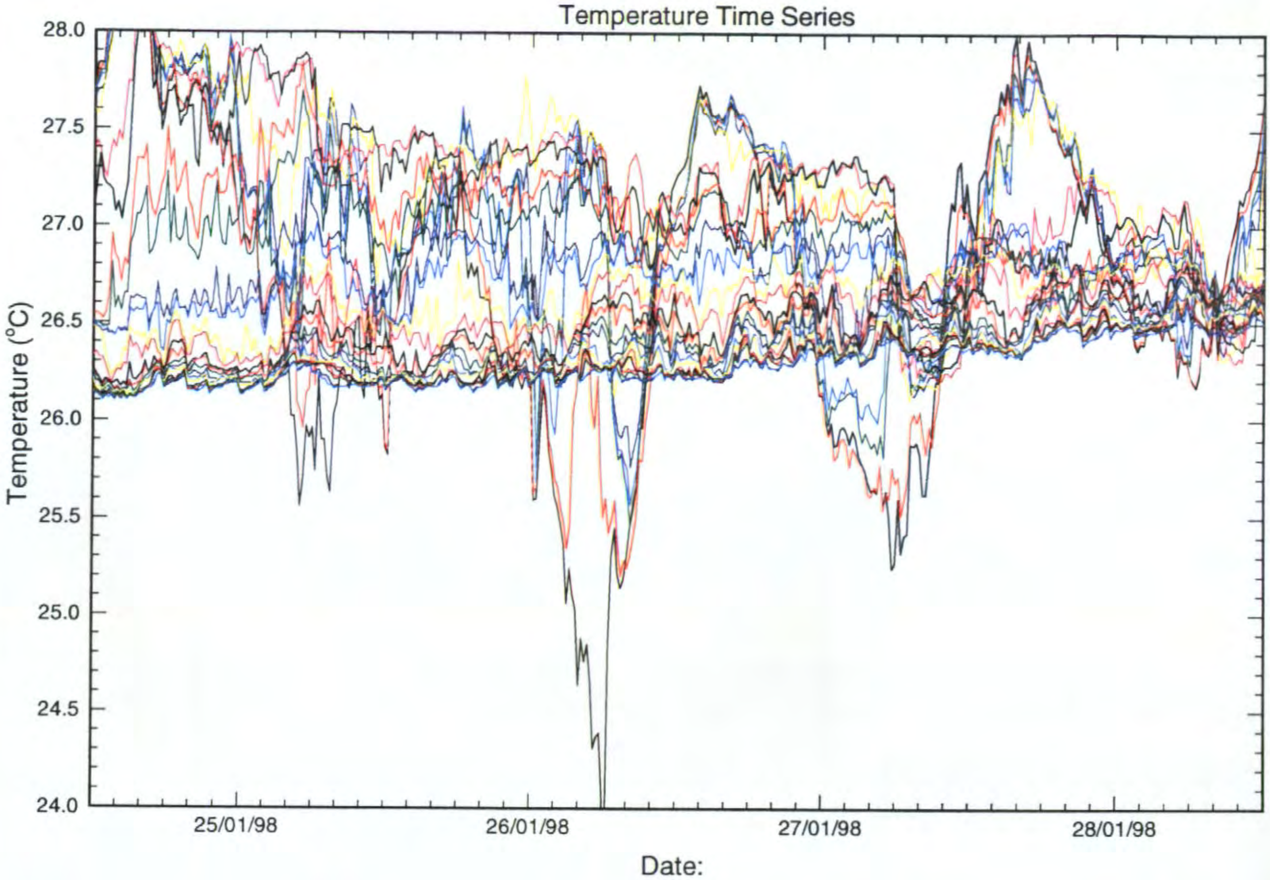


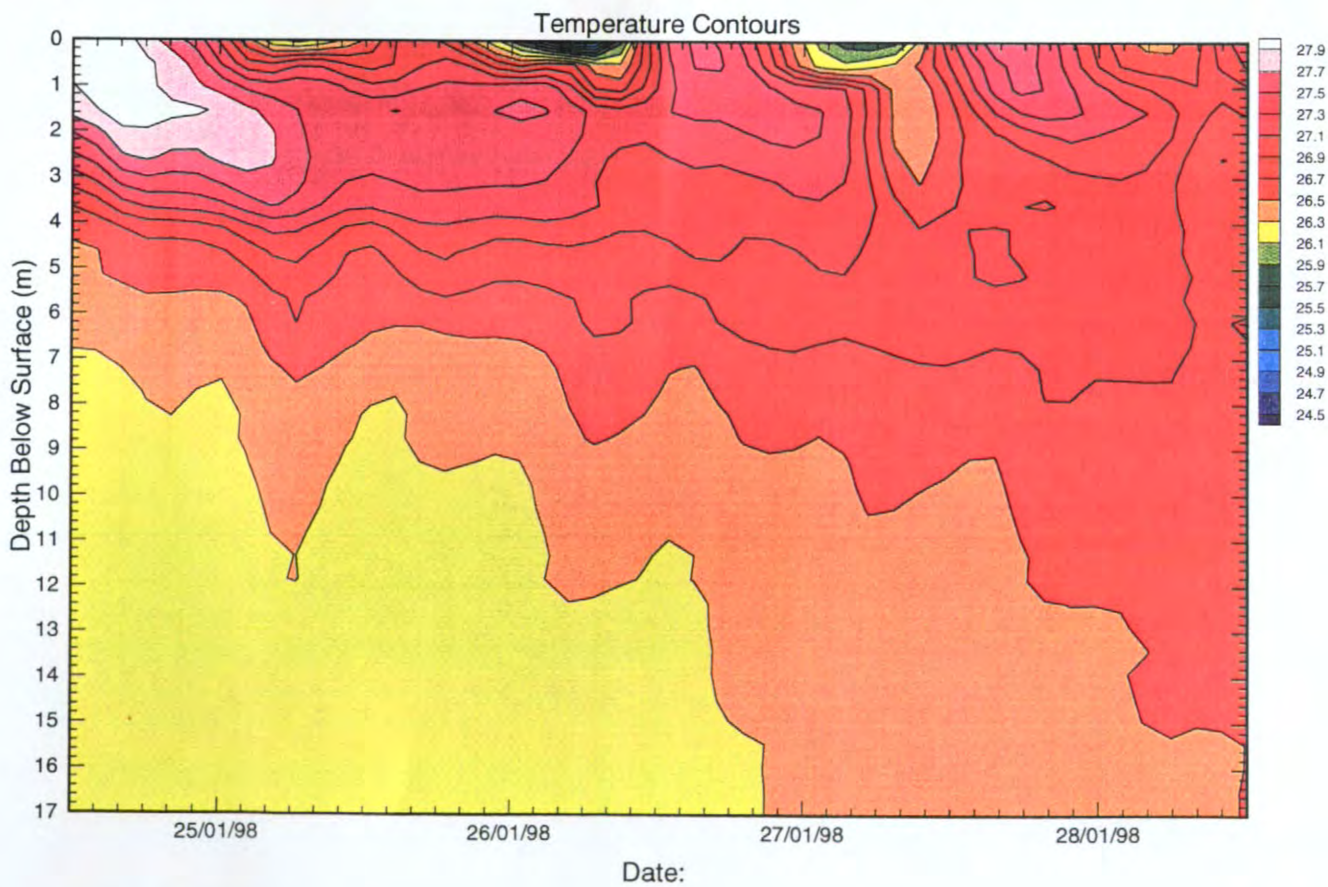
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	BEROWRA CREEK - CUNIO POINT TEMPERATURE TIME SERIES AND CONTOURS 27 TO 31 DECEMBER 1997	MHL Report 928 Figure 3.5 DRAWING TC1_COL.BPS





NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	BEROWRA CREEK - CUNIO POINT TEMPERATURE TIME SERIES AND CONTOURS 8 TO 11 JANUARY 1998	MHL Report 928 Figure 3.6 DRAWING TC2_COL.BPS





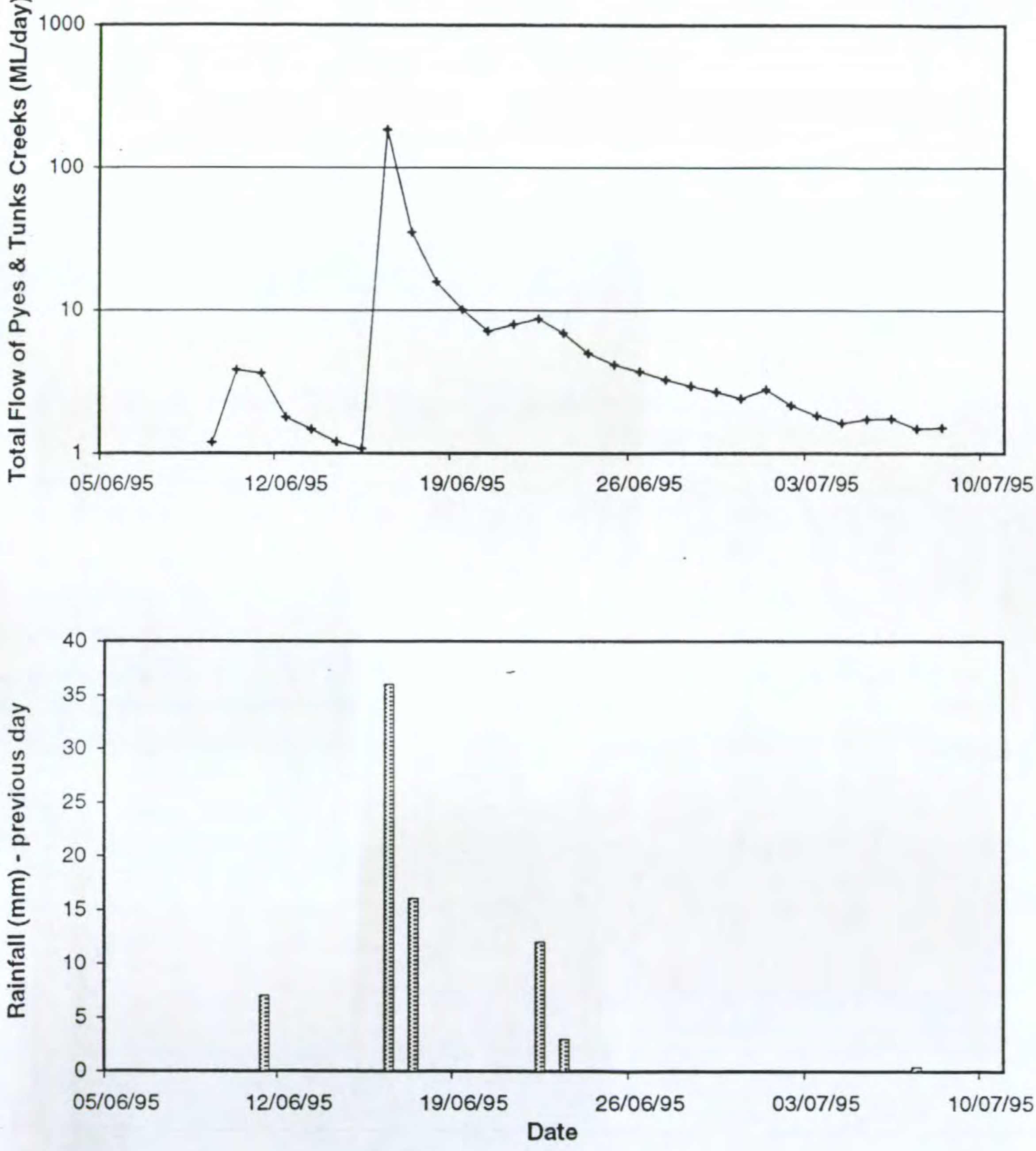
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

BEROWRA CREEK - CUNIO POINT
TEMPERATURE TIME SERIES AND CONTOURS
24 TO 28 JANUARY 1998

MHL Report 928

Figure 3.7

DRAWING TC3_COL.BPS



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

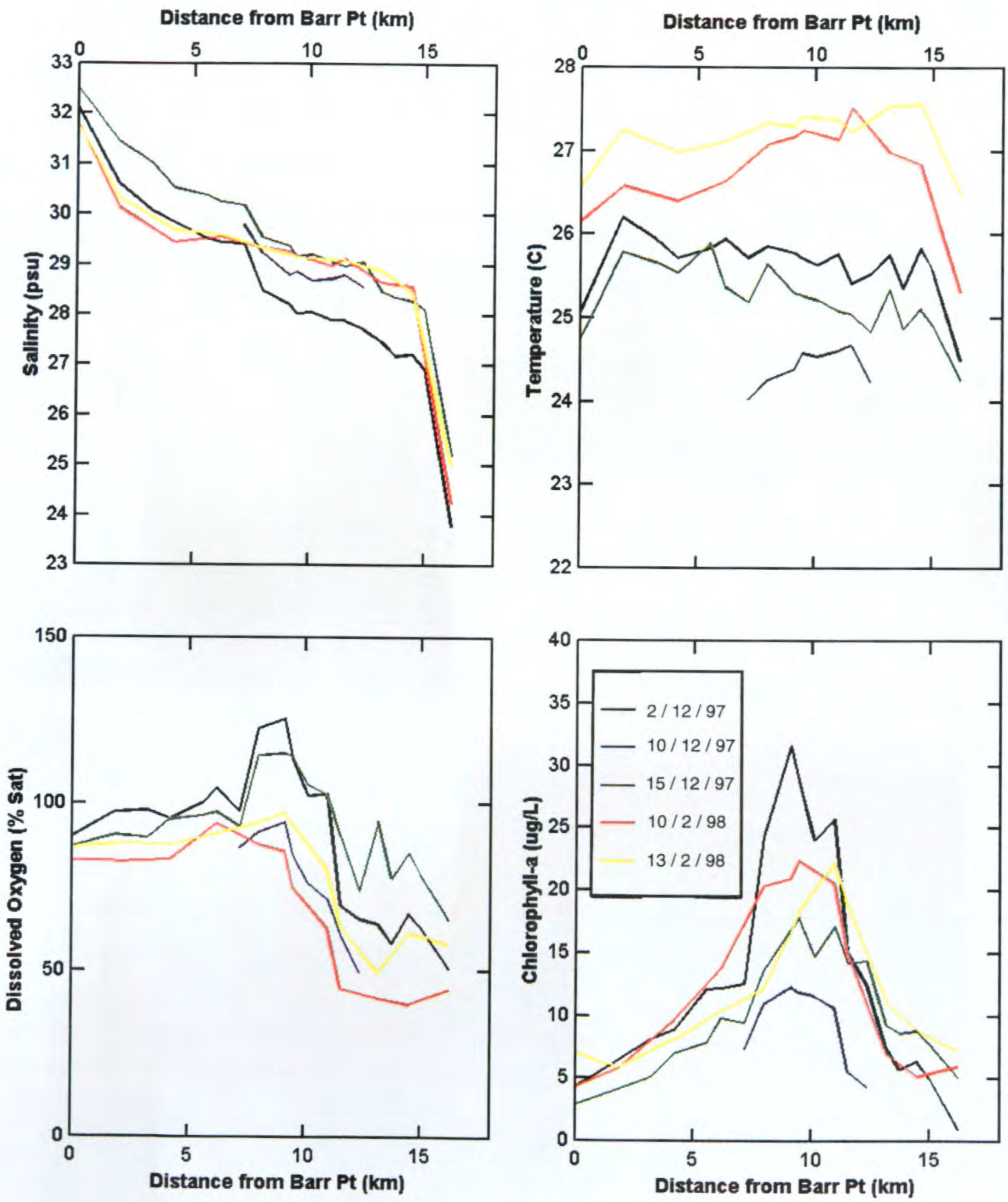
RAINFALL AND DISCHARGE HYDROGRAPH FOR BEROWRA CREEK

MHL Report 928

Figure 3.8

DRAWING 928_3_08.DRW

Depth Averaged Data



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

MANLY HYDRAULICS LABORATORY

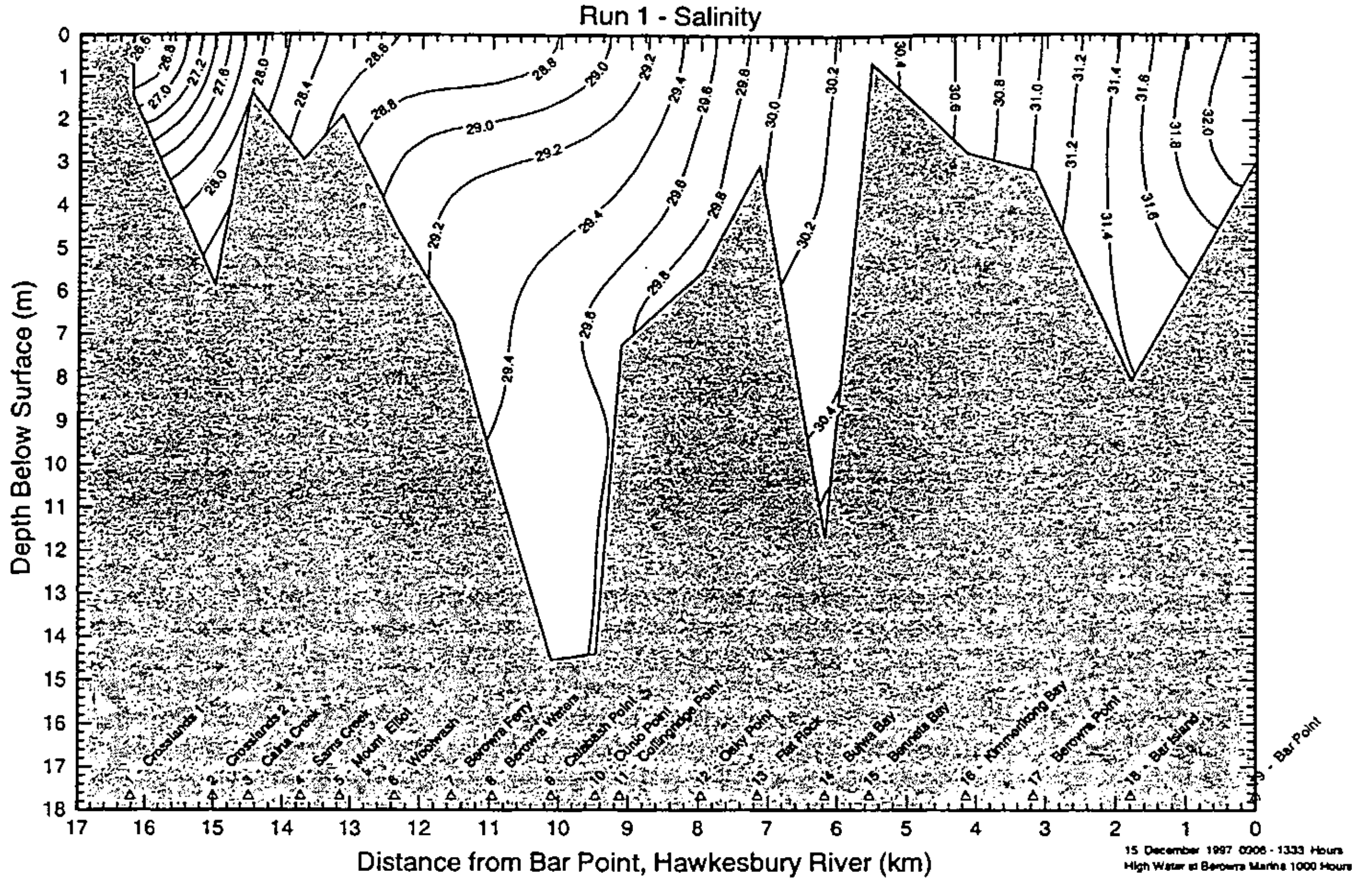
CTD DATA IN BEROWRA CREEK ESTUARY

MHL Report 928

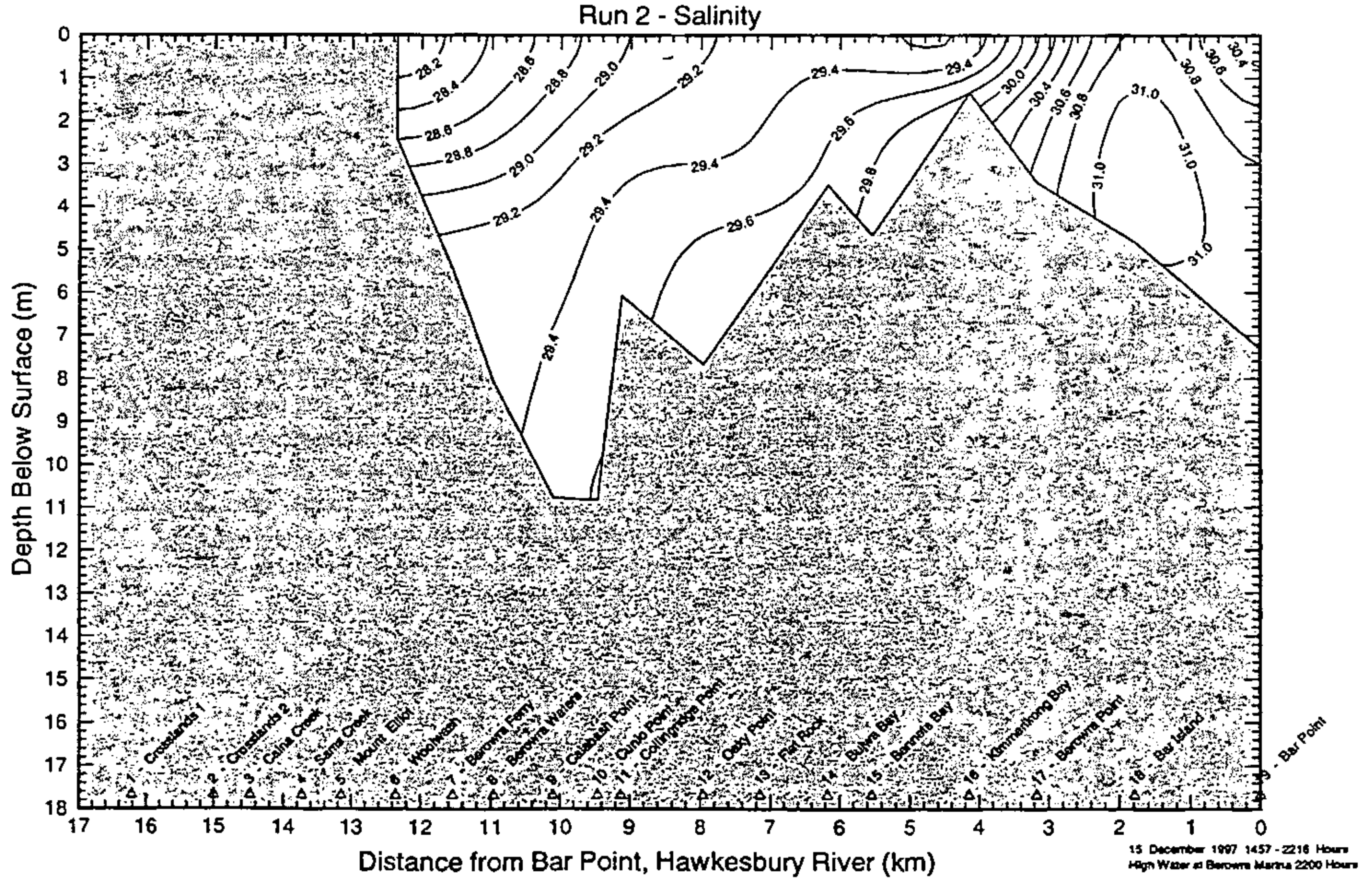
Figure 3.9

DRAWING 928ber39.CDR

Run 1 - Salinity



Run 2 - Salinity



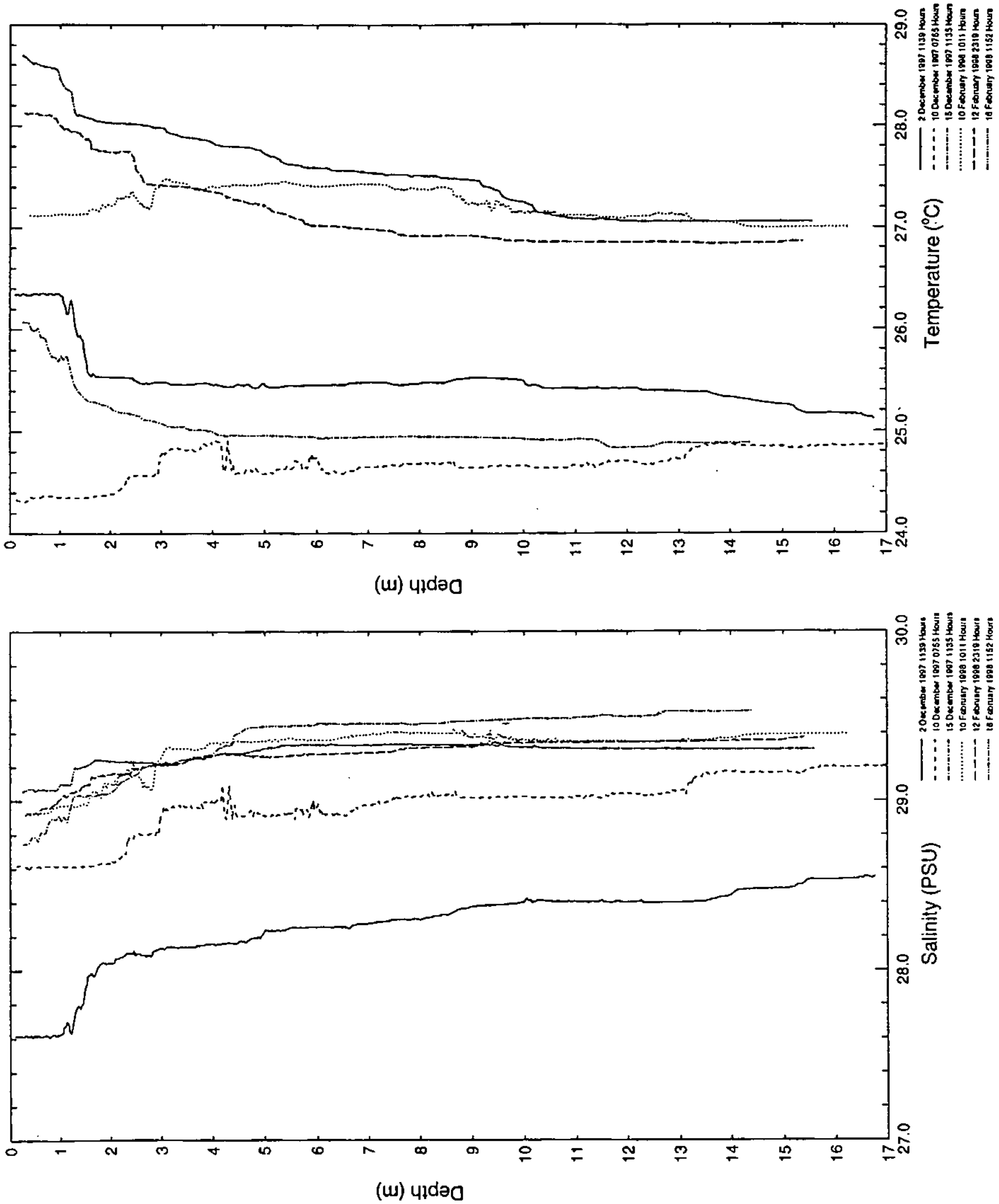
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

BEROWRA CREEK
LONG CHANNEL SALINITY CONTOURS
15 DECEMBER 1997

MHL Report 928

Figure 3.10

DRAWING SAL BPS



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

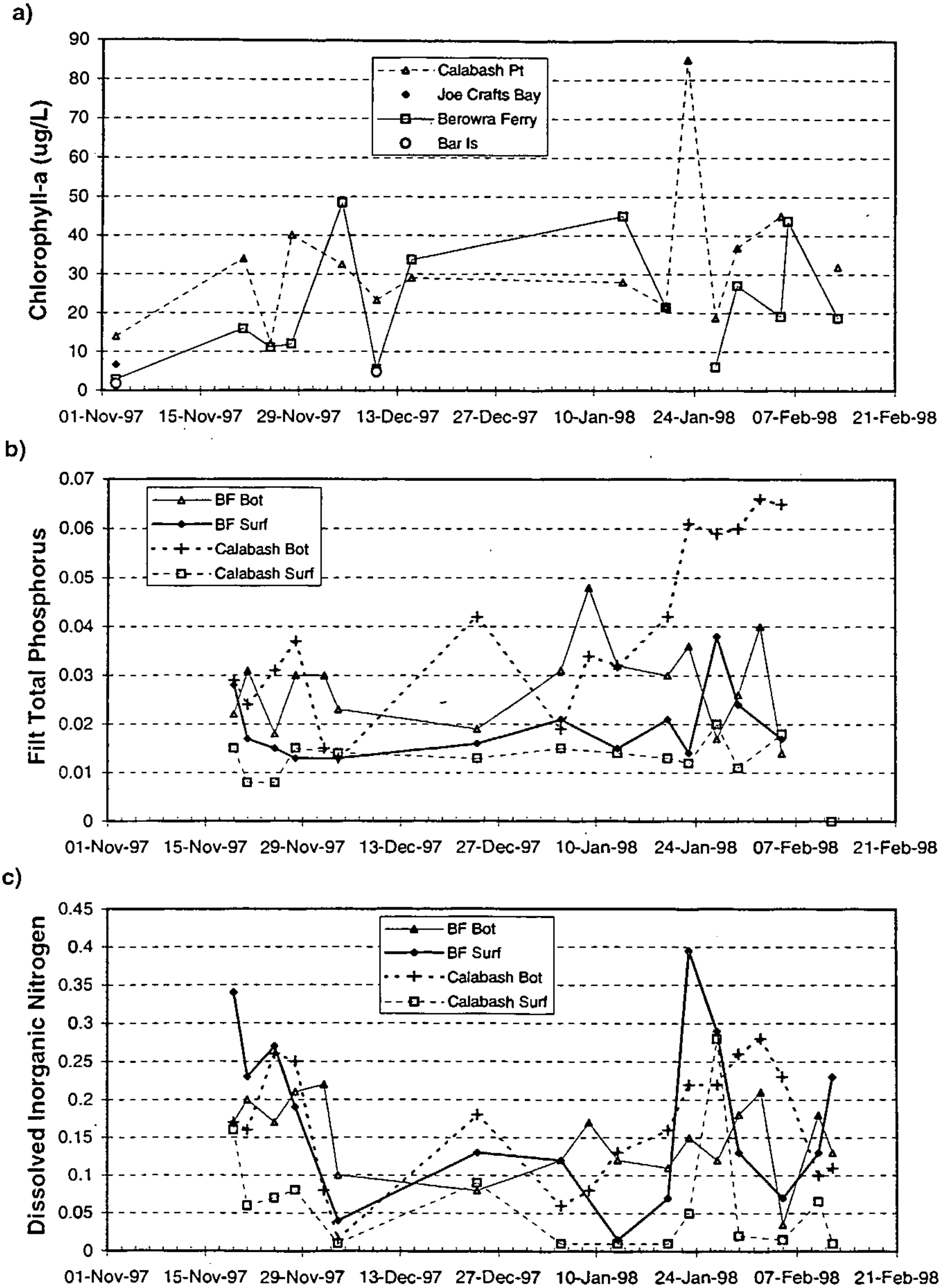
MANLY HYDRAULICS LABORATORY

SALINITY AND TEMPERATURE
CUNIO POINT
2 DECEMBER 1997 TO 16 FEBRUARY 1998

MHL Report 928

Figure 3.11

DRAWING 928_0311.BPS



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

MANLY HYDRAULICS LABORATORY

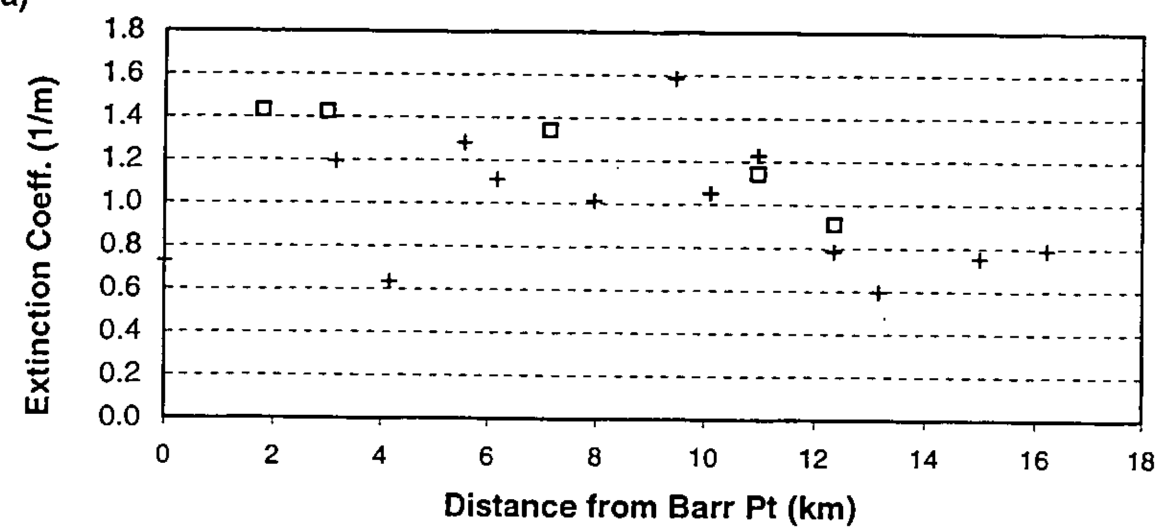
CHLOROPHYLL - a, FILTERED TOTAL PHOSPHORUS AND DISSOLVED INORGANIC NITROGEN IN BEROWRA CREEK OVER SUMMER 1997-98

MHL Report 928

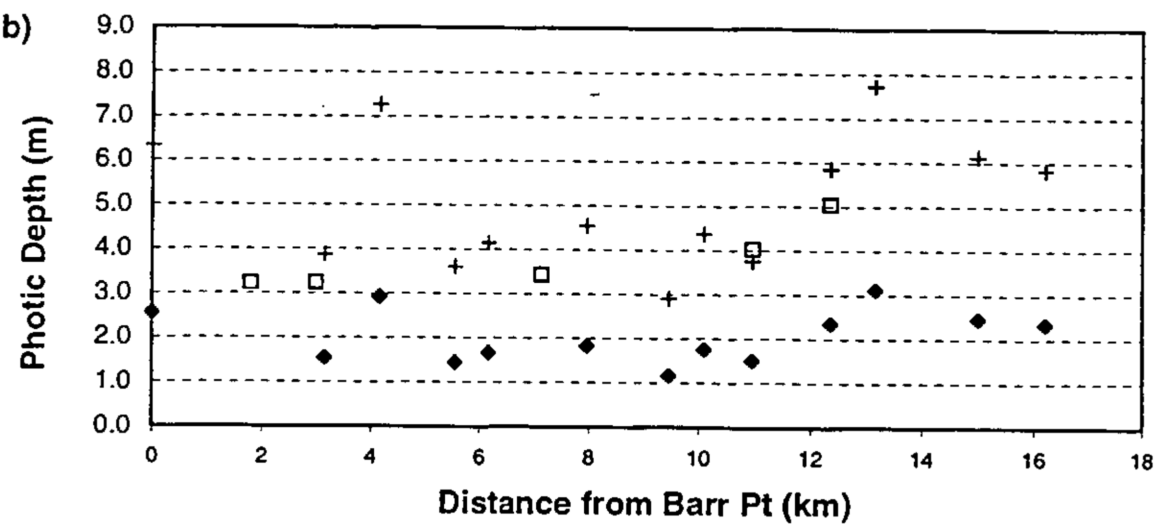
Figure 3.12

DRAWING 928_3_12.DRW

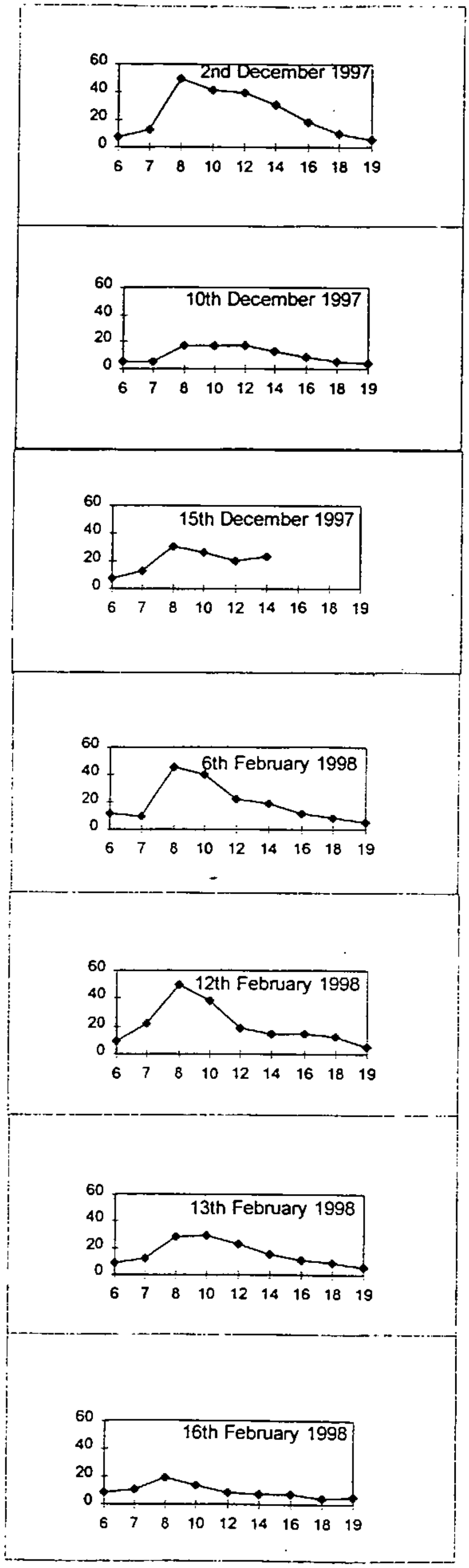
a)



b)



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	ATTENUATION COEFFICIENT AND PHOTIC DEPTH 2 DECEMBER 1997	MHL Report 928 Figure 3.13 DRAWING 928_3_13.DRW



Legend

y axis: µg Chla/L

x axis:

site no.	Distance from Hawkesbury R. (km)
6	12.35
7	11.54
8	10.96
10	9.46
12	7.96
14	6.17
16	4.15
18	1.79
19	0

NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

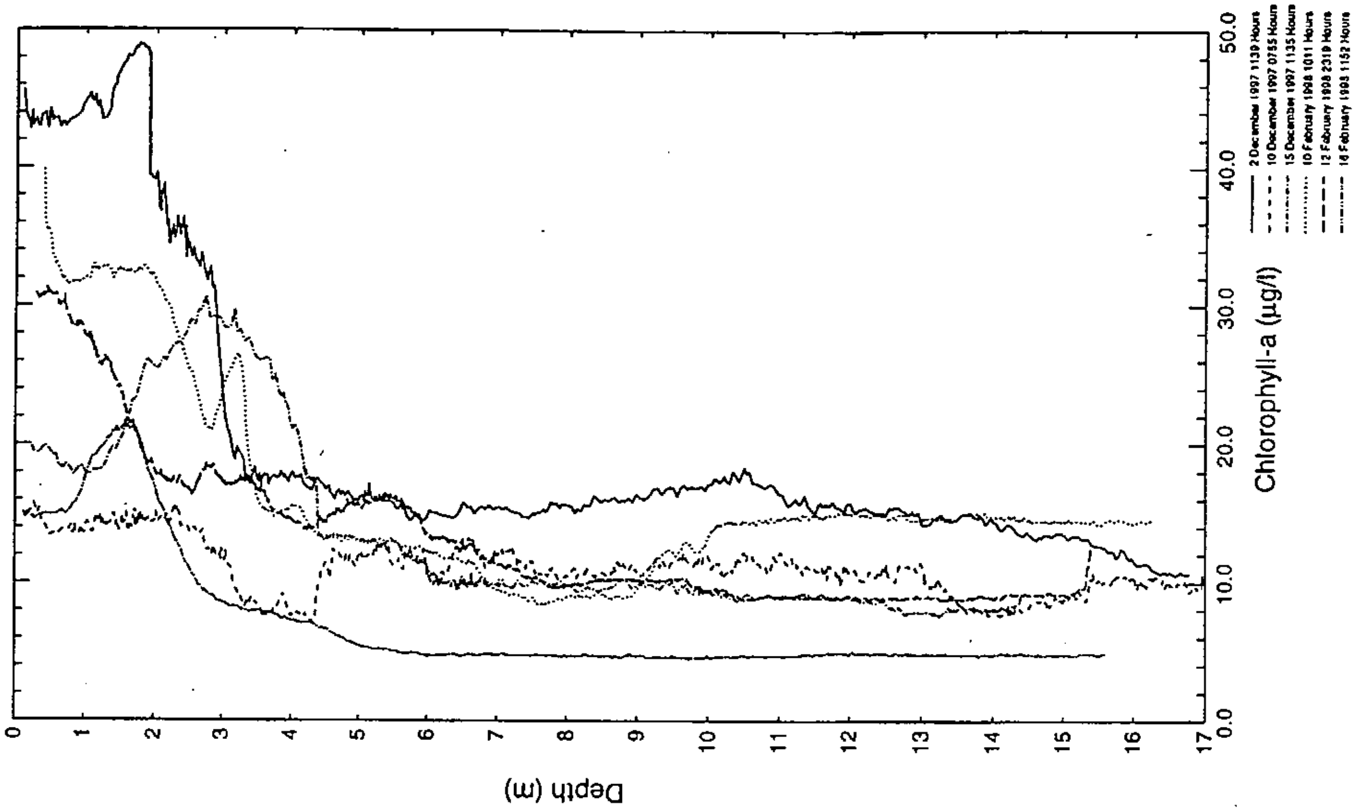
MANLY HYDRAULICS LABORATORY

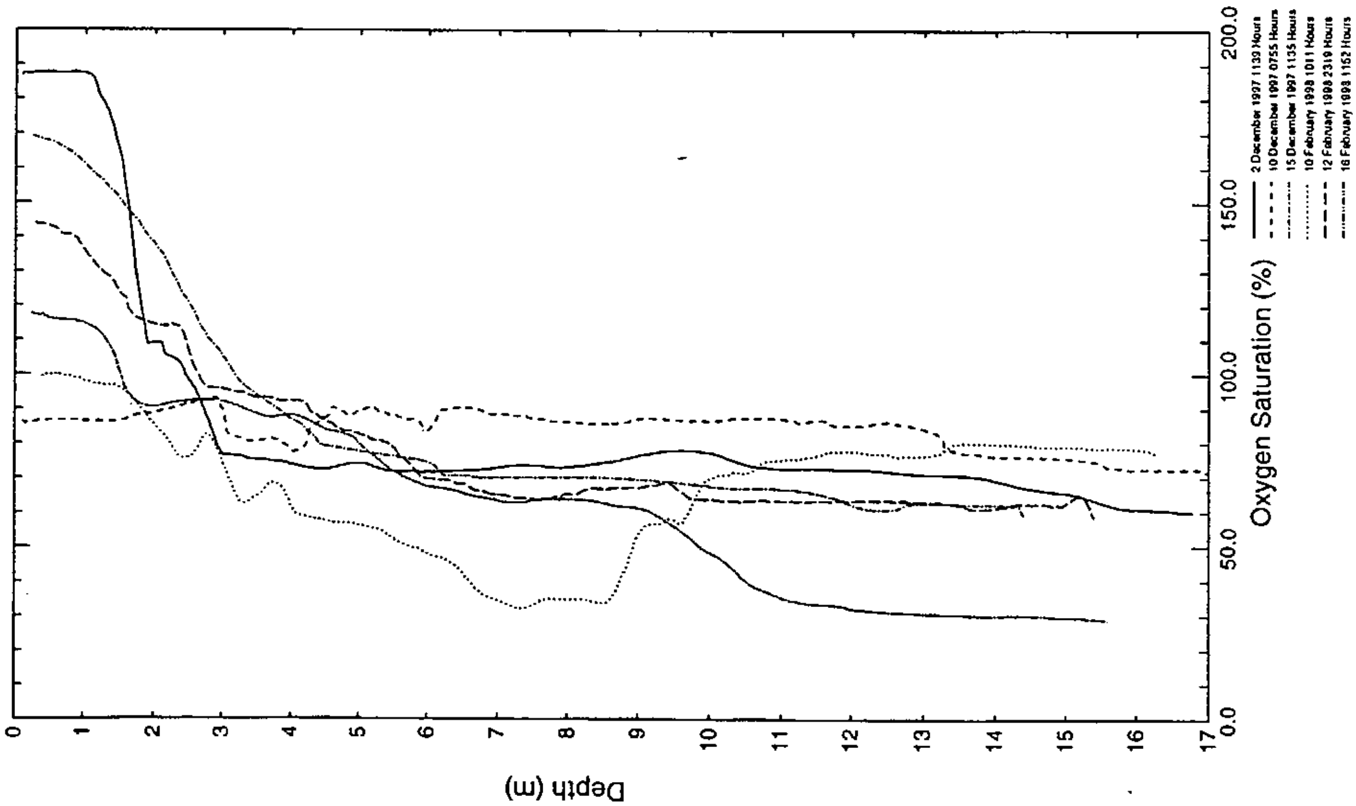
CHLOROPHYLL - a AT LOW TIDE IN BEROWRA CREEK ESTUARY

MHL Report 928

Figure 3.14

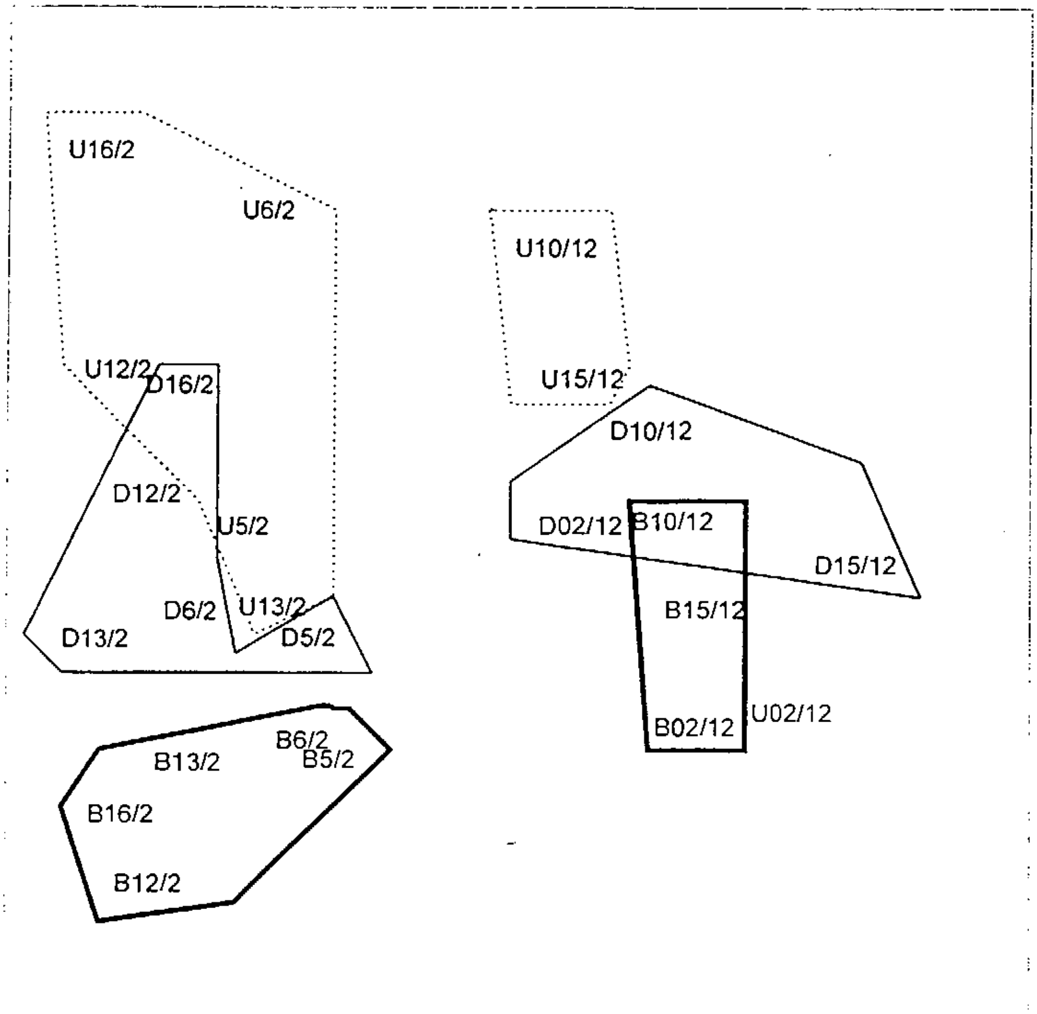
DRAWING 928_3_14.DRW





NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	CHLOROPHYLL-A AND DISSOLVED OXYGEN CUNIO POINT 2 DECEMBER 1997 TO 16 FEBRUARY 1998	MHL Report 928 Figure 3.15 DRAWING 928_0315.BPS

The six groups indicate separation between upstream, bloom and downstream sites over the December and February sampling periods.



Legend

Symbols indicate site and date

U	Upstream of bloom
B	In bloom
D	Downstream of bloom

02/12	2 December 1997
10/12	10 December 1997
15/12	15 December 1997
5/2	5 February 1998
6/2	6 February 1998
12/2	12 February 1998
13/2	13 February 1998
16/2	16 February 1998

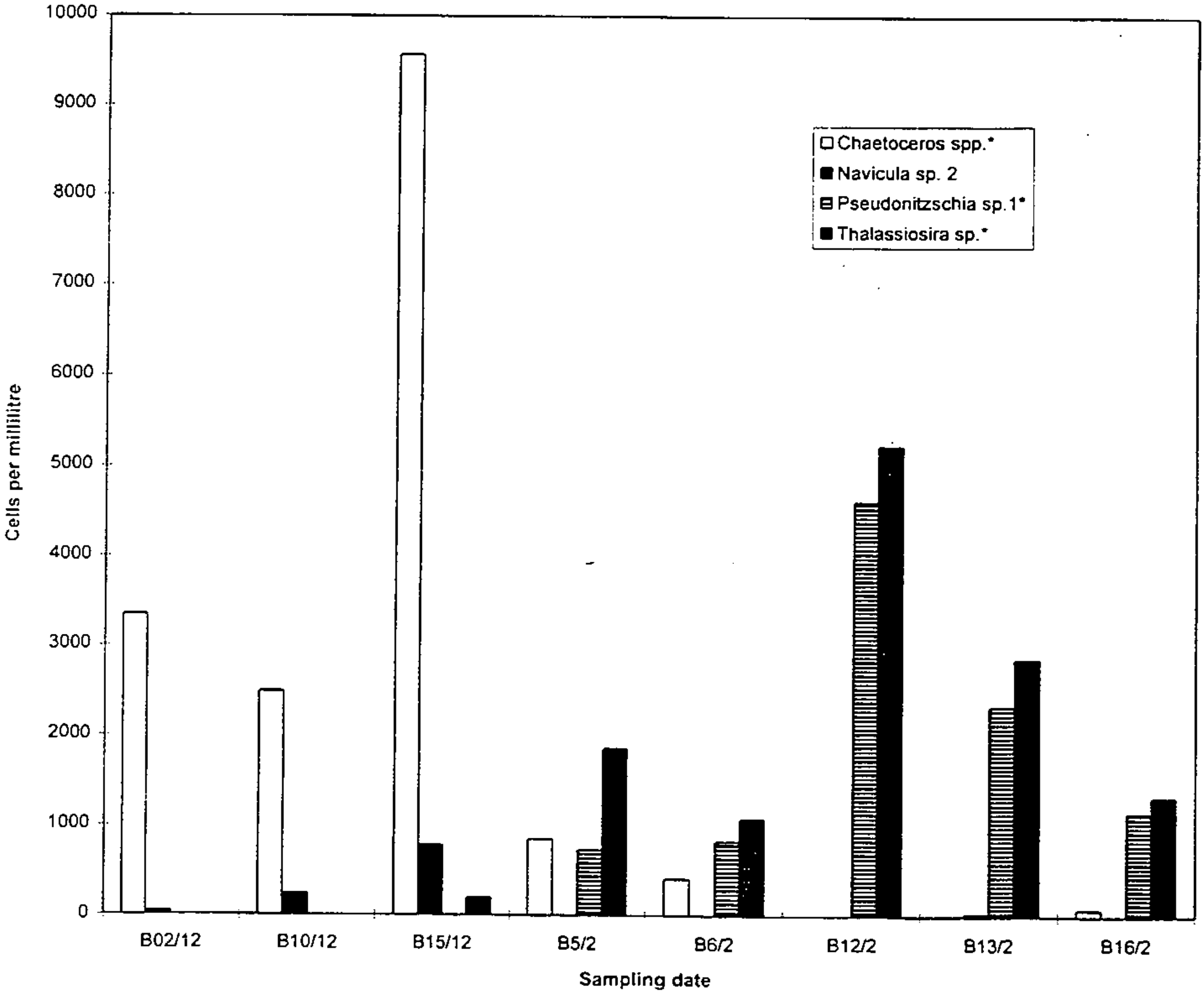
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

MULTIDIMENSIONAL SCALING ANALYSIS OF PHYTOPLANKTON TAXA FOR SUMMER 1997-98
STRESS = 0.13

MHL Report 928

Figure 3.16

DRAWING 926_3_16.DRW



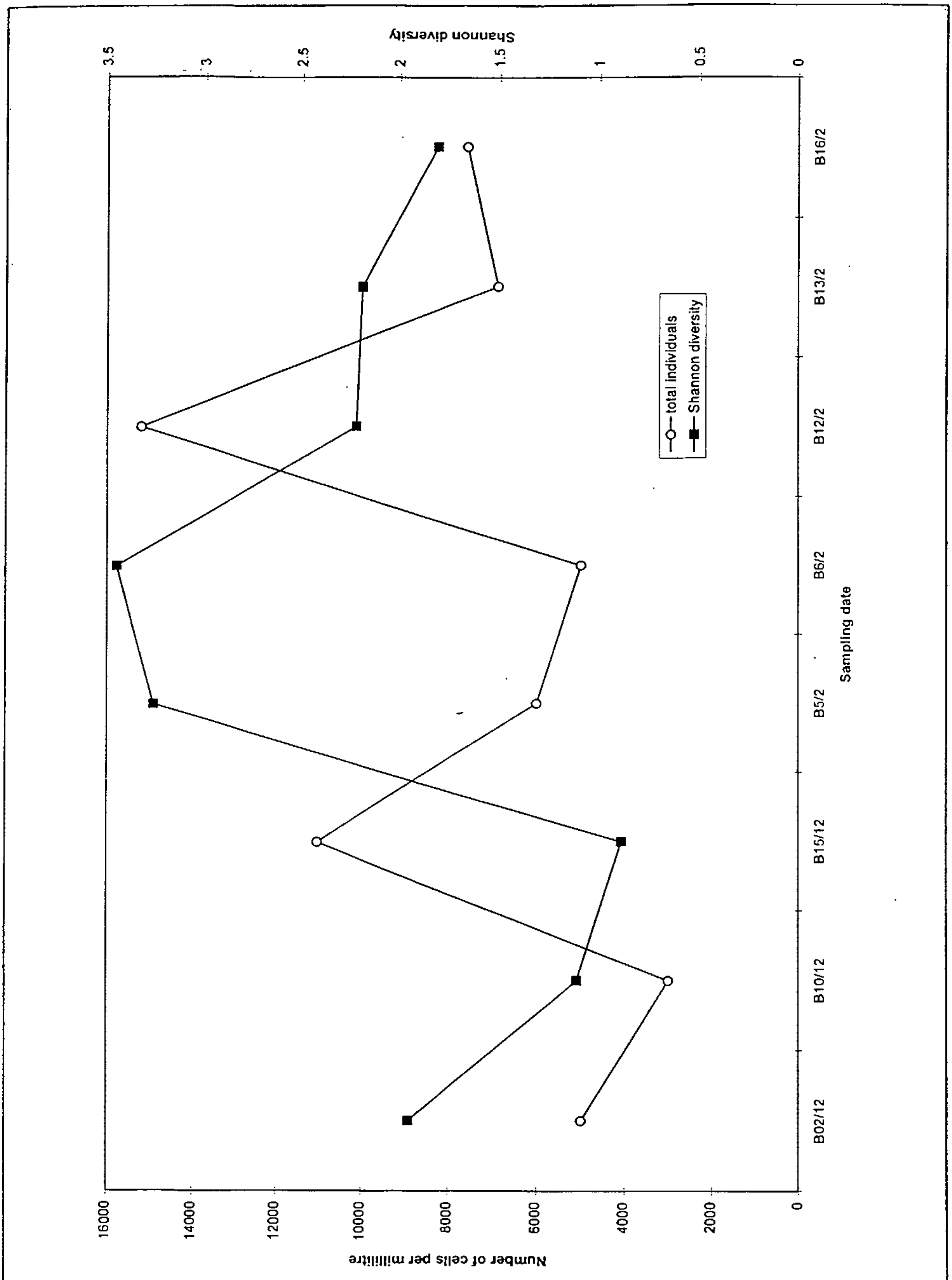
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

SUCCESSION OF DOMINANT TAXA IN THE BLOOM SAMPLES (DLWC) DURING DECEMBER 1997 AND FEBRUARY 1998

MHL Report 928

Figure 3.17

DRAWING 928_3_17.DRW



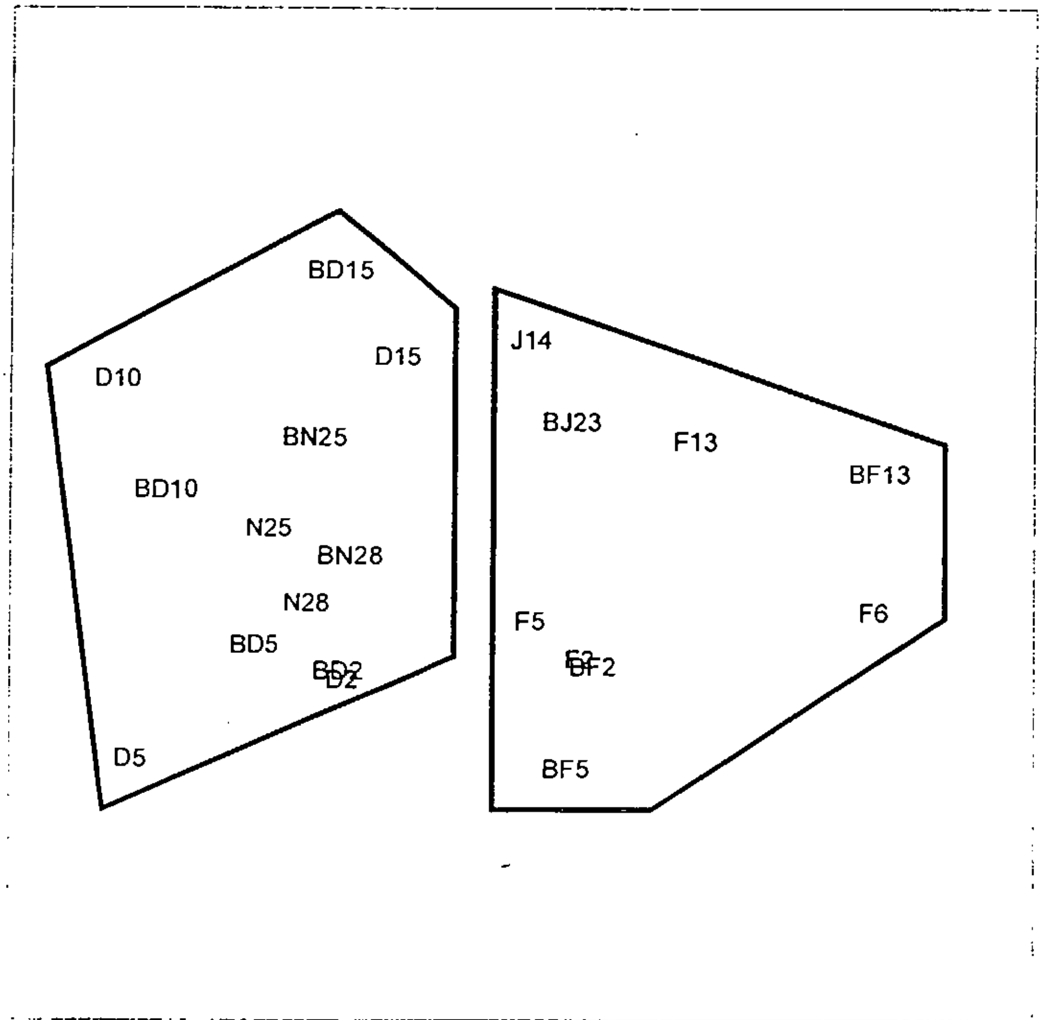
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

ALGAL COUNTS AND SHANNON DIVERSITY WITHIN THE BLOOM DURING DECEMBER 1997 AND FEBRUARY 1998

MHL Report 928
Figure 3.18

DRAWING 928_3_18.DRW

The two groups indicate separation between the two sampling periods November/December 1997 and January/February 1998.



Legend

Symbols indicate site and date

B Calabash Bay

N25 25 November 1997

N28 28 November 1997

D2 2 December 1997

D5 5 December 1997

D10 10 December 1997

D15 15 December 1997

J14 14 January 1998

J23 23 January 1998

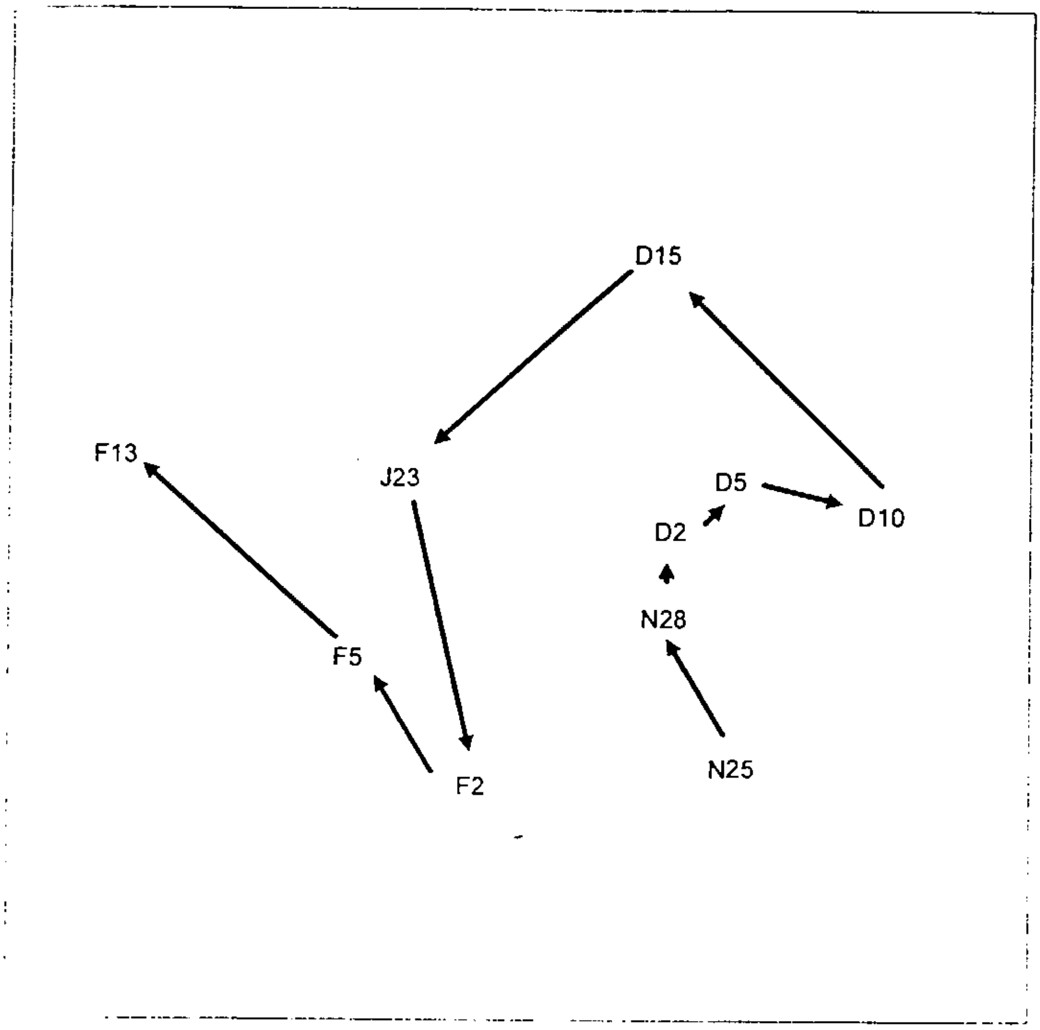
F2 2 February 1998

F5 5 February 1998

F6 6 February 1998

F13 13 February 1998

Arrows indicate direction of community changes over time.



Legend

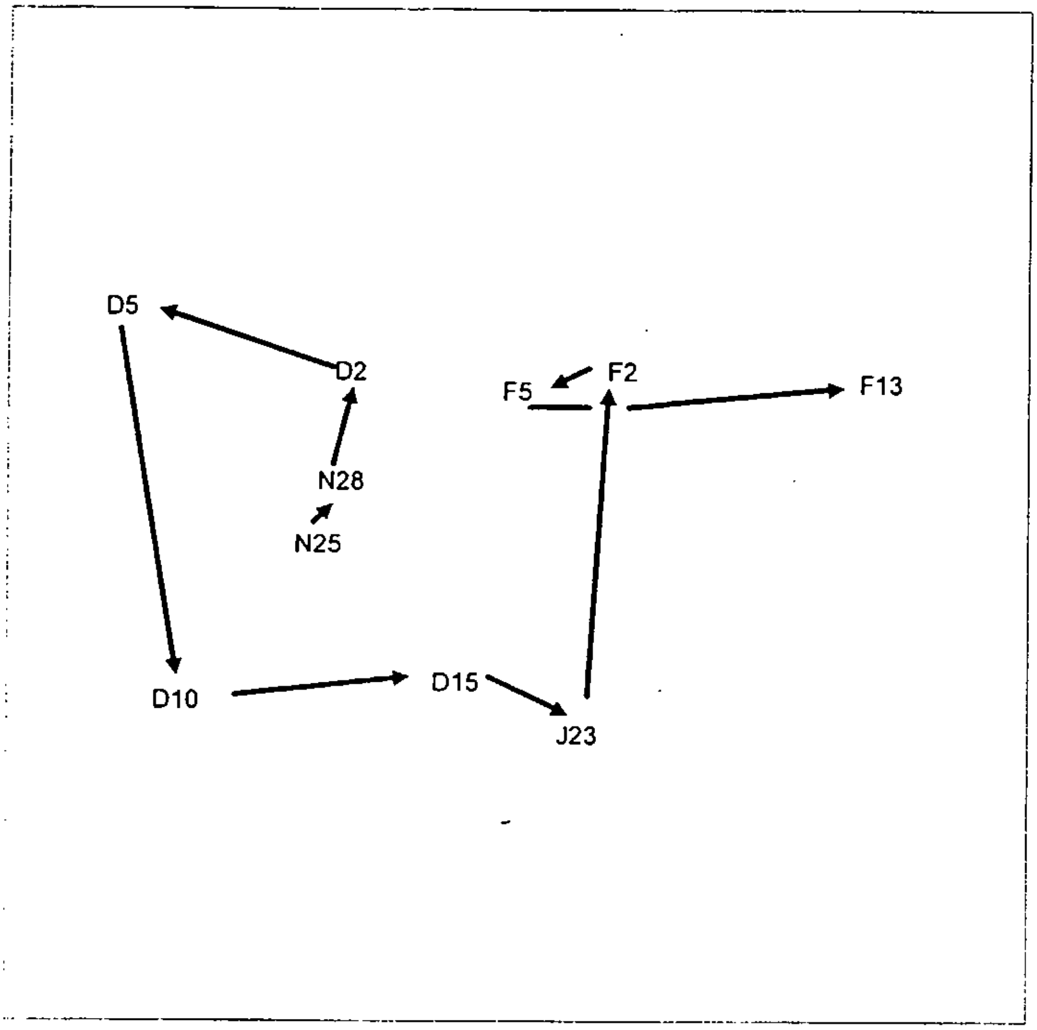
N25 25 November 1997
N28 28 November 1997
D2 2 December 1997
D5 5 December 1997
D10 10 December 1997
D15 15 December 1997
J23 23 January 1998
F2 2 February 1998
F5 5 February 1998
F13 13 February 1998

NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

MULTIDIMENSIONAL SCALING ANALYSIS OF PHYTOPLANKTON SAMPLES FROM CALABASH BAY SUMMER 1997-98

MHL Report 928
Figure 3.20
DRAWING 928_3_20.DRW

Arrows indicate direction of community changes over time.



Legend

N25	25	November 1997
N28	28	November 1997
D2	2	December 1997
D5	5	December 1997
D10	10	December 1997
D15	15	December 1997
J23	23	January 1998
F2	2	February 1998
F5	5	February 1998
F13	13	February 1998

NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

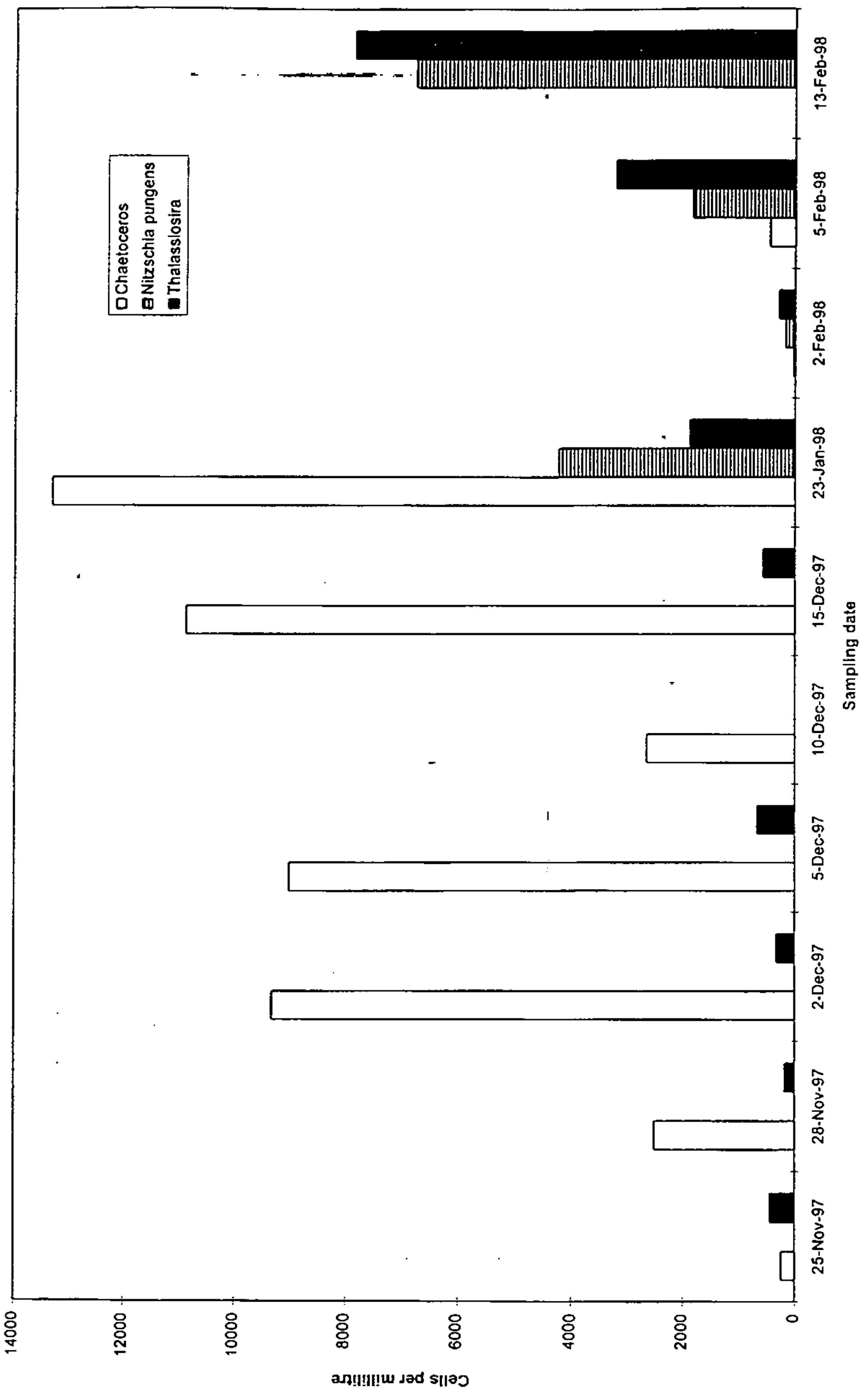
MANLY HYDRAULICS LABORATORY

MULTIDIMENSIONAL SCALING ANALYSIS OF PHYTOPLANKTON SAMPLES FROM BEROWRA FERRY SUMMER 1997-98

MHL Report 928

Figure 3.21

DRAWING 928_3_21.DRW



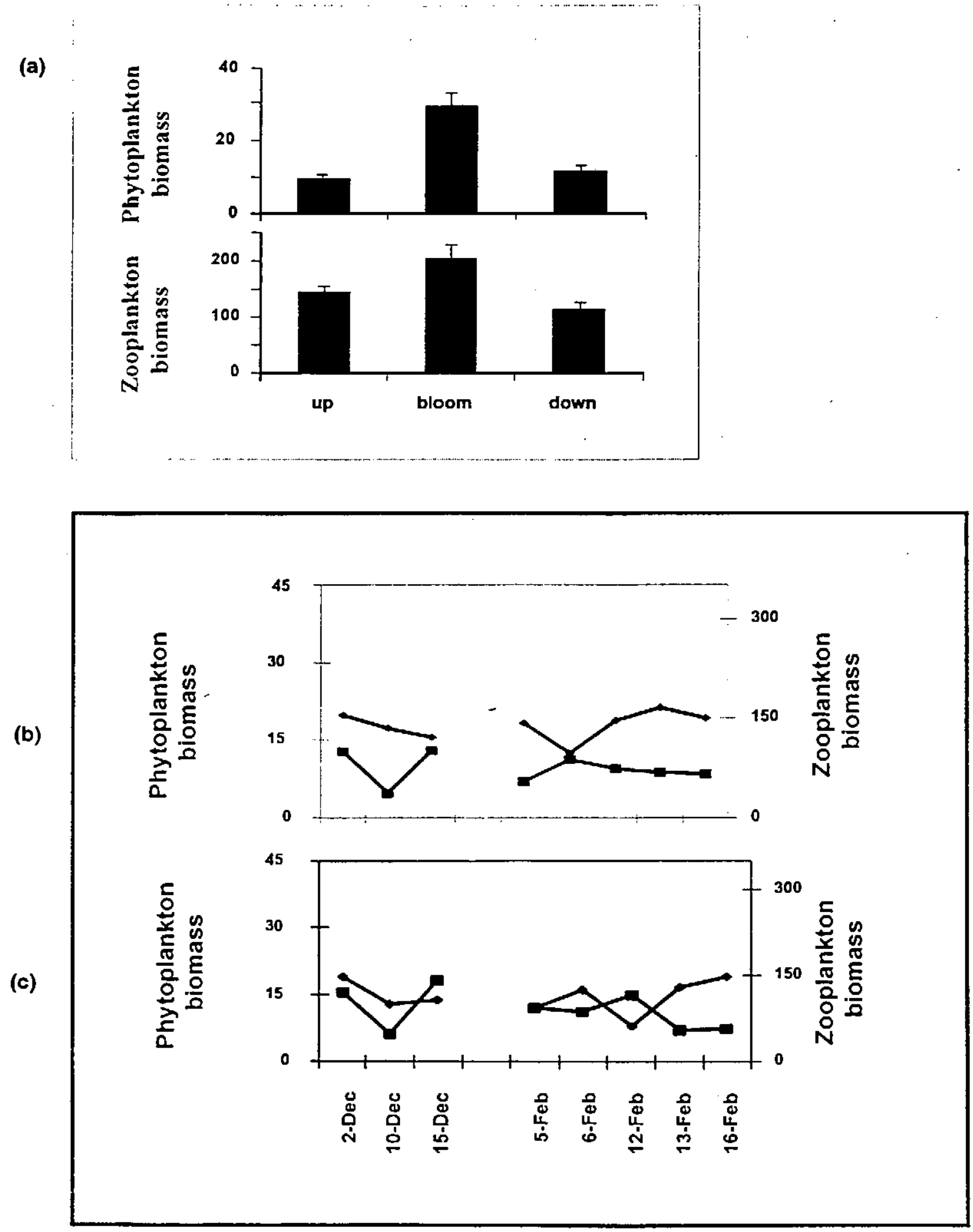
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

SUCCESSION OF DOMINANT TAXA AT CALABASH BAY BLOOM SAMPLES (HSC) DURING SUMMER 1997-98

MHL Report 928

Figure 3.22

DRAWING 928_3_22.DRW



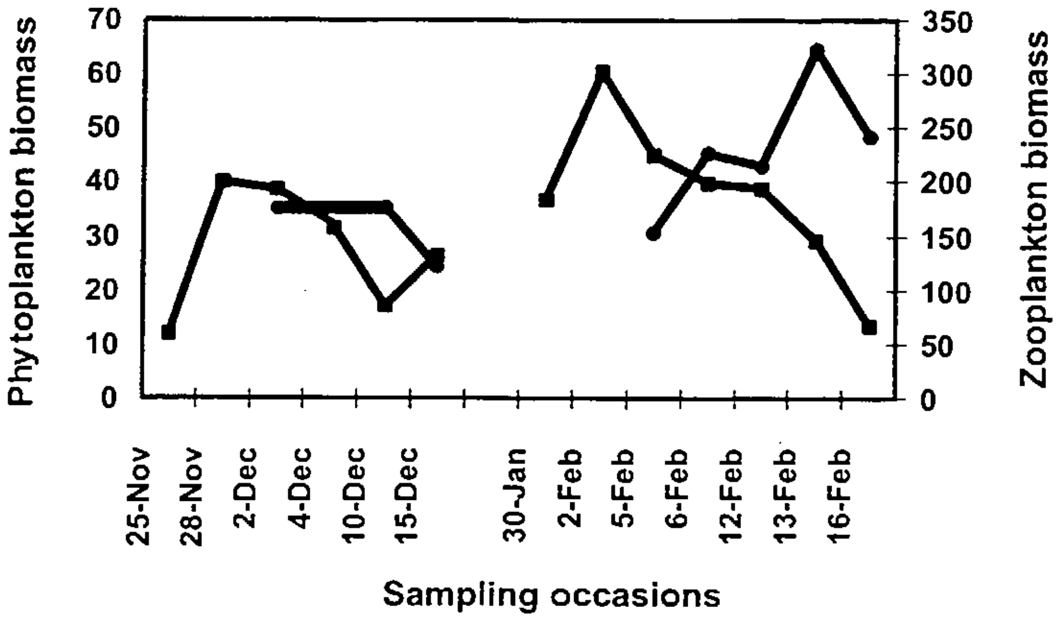
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

AVERAGE BIOMASS AND VARIABILITY OF PHYTOPLANKTON AND ZOOPLANKTON OVER THE STUDY PERIOD

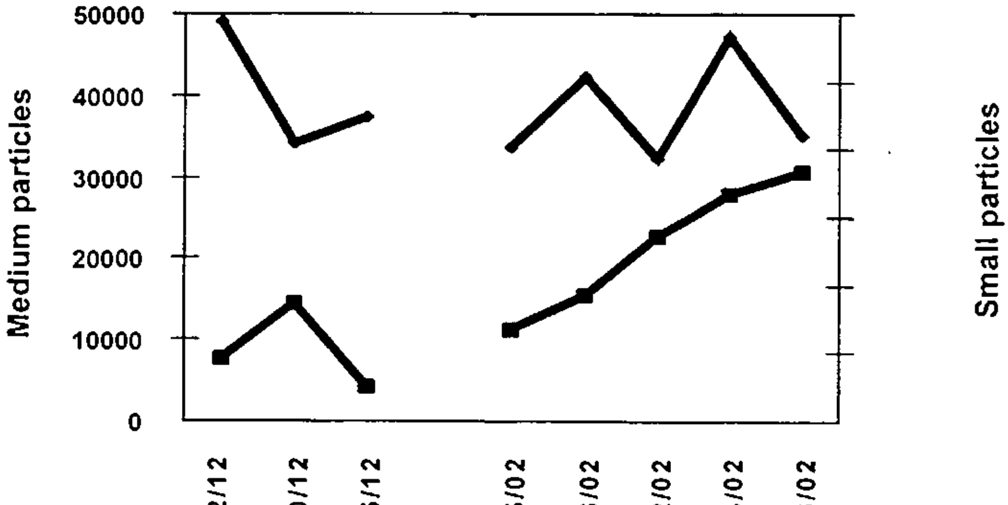
MHL Report 928
Figure 3.24

DRAWING 928_3_24.DRW

(a)



(b)



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	(a) PHYTOPLANKTON AND ZOOPLANKTON BIOMASS AND (b) SMALL AND MEDIUM PARTICLES AT THE BLOOM STATION	MHL Report 928 Figure 3.25 DRAWING 928_3_25.DRW

4. Nutrient Cycling

4.1 Sediment Nutrient Fluxes

No significant relationship was found between nutrient flux measurements and initial nutrient concentration overlying the sediment core. Therefore, all flux measurements were compared with regard to the origin of the sample and the oxic status of the water column. Sediment-water exchanges of nutrients is mainly driven by microbial and physico-chemical processes, the variation of exchanges reflects the great spatial heterogeneity of the distribution of benthic bacterial populations and the structure of the sediment environments.

In terms of nutrient cycling, the overall lack of difference between deep sites and shallow sites, suggests that if the sediment in the area do have a significant influence on the algal response, it is not the sediment in the holes which is important. Sediments from the whole area investigated tend to flux nutrients at similar rates.

Therefore, no significant difference in benthic nutrient fluxes quantified in sediments from deep and shallow sites in the study area were detected, regardless of the oxygenation of the water column in the experimental cores.

Yet, sediments from both shallow and deep sites still constitute a substantial source of nitrogen, with important release of ammonium readily available for pelagic primary production. These sediments also constitute a source of phosphorus but its release from sediments was not significantly enhanced with hypoxic/anoxic conditions.

In situ microbial decomposition of organic material within the water column may also contribute to the inorganic nutrient pool. This source was not monitored during the investigations.

4.2 Riverine Loads

4.2.1 Concentrations

Nutrient loads to the system enter the estuary via a number of creeks. Estimates of nutrient loads are difficult to obtain because coincident sampling of concentration and discharge are required. Hornsby council have been monitoring water quality in a number of creeks on a bi-weekly and monthly basis over the past 3 years. Data from a selection of sites (Figure 4.1) have been analysed and box plots for total nitrogen and total phosphorus are in Figure 4.2. All data were included in the analysis with no attempt to separate wet and dry weather periods.

The influence of the STPs is most prevalent in total nitrogen, with sites at and just downstream of the STP discharges recording the highest concentrations. The other site recording high total nitrogen concentrations was Arcadia Tip Leachate Pond. Total phosphorus concentrations were highest in Sam's Creek while sites downstream of the STP discharge TP concentrations were generally similar or slightly higher than the average creek concentration. The very high TP concentrations in Sam's Creek require further investigation to determine the source. Even though the TP concentration of the STP discharges is similar to the other creeks the STP loads are considerably greater because of the persistent discharge from the STPs. While the natural dry weather flows in most creeks are less than 0.001 ML/day the STP flows are around 5 and 10 ML/day at Hornsby Heights and West Hornsby, respectively. During dry weather the STP loads exceed the catchment loads (Table B2, Appendix B).

4.2.2 Load Estimates

Pollutant loads from the upstream catchment heavily influence the water quality of the Berowra Creek estuary. Phosphorus and nitrogen loads into the estuary are of particular concern as they may force the estuary to enter a eutrophic state, hence increasing the likelihood of harmful algal blooms occurring.

Pollutant loads from catchments are subject to a very large number of influences which may be difficult to quantify. Probably the two most important of these are rainfall and land use.

Water quality modelling for the whole Berowra Creek catchment has been carried out by AWT EnSight (1997a, 1997b). The modelling provides a comprehensive assessment of water quality in the catchment, drawing on previous data gathered and estimating loads where no data was available.

Figures 4.3 and 4.4 show a comparison of the model results against pollutant load data for total phosphorus and total nitrogen. The estimated loads at Galston Gorge from the AWT EnSight model and from the data collected are in good agreement, given the number of assumptions that had to be made regarding the flows and concentrations from the STP. The predictions that the model makes for the other sub-catchments can therefore be expected to be reliable. However, it is desirable to have a more easily tractable way to estimate catchment loads for Berowra Creek rather than using the model. Two regression techniques were therefore used to try to relate the daily loads to daily rainfall for each sub-catchment. The techniques used were simple linear correlation and linear correlation with a step function at low rainfall. These are discussed in Appendix B along with the comparison of the model results against pollutant load data.

4.3 Comparison with Previous Studies

The nutrient concentrations downstream of the STPs and in the estuary at Crosslands and Berowra Waters are shown in Tables 4.1, 4.2 and 4.3. These data were compiled previously (Berowra Technical Working Party November 1995 Berowra Creek Draft Water Quality Management Strategy) and the last column tabulates the most recent data.

The Hornsby Heights and West Hornsby STPs were completed in 1981 and 1975 respectively. The limited data available for years prior to the completion of the plants make it difficult to assess any trends in the total phosphorus concentrations (Table 4.1). The decrease between 1977 and 1992-93 is due to the introduction of phosphorus removal at the STPs. Over the past six years there appears to have been little change in the TP concentration, although the data suggest there may have been a slight increase in TP concentration.

Oxidised nitrogen concentrations do not show any significant trends over the past six years, although the concentration at Berowra Waters appears to have decreased (Table 4.2). Total nitrogen concentrations also show similar trends, with the suggestion of a decrease in concentrations during the past four years (Table 4.3).

Table 4.1 Comparison with Results of Previous Studies Total Phosphorus (mg/L)

Monitoring Site	Guideline	1977 (SPCC)	1992-93 (AWT)	1994-95			1994-97 (HSC)
				(AWT)	(HC)	(EPA)	
Berowra Creek at Berowra Waters	-	0.08 (<0.02-0.29) 9	0.02 (0.01–0.05) 11	0.03 (0.01-0.06) 9	-	0.047 (0.018–0.130) 45	* 0.051 (0.028-0.072) 33
Berowra Creek at Crosslands	-	0.52 (0.06–2.6) 9	0.06 (0.02–0.17) 11	0.04 (0.02–0.06) 9	-	0.063 (0.018–0.130) 30	
Berowra Creek at Fishponds Waterhole	0.01–0.1	6.8 (6.5–7.1) 2	0.19 (0.08–0.79) 11	0.05 (0.04–0.05) 3	0.044 (0.021–0.073) 10	-	0.121 (0.011-2.359) 40
Waitara Creek downstream of West Hornsby STP	0.01–0.1	6.8 (6.3–9.2) 4	0.35 (0.18–1.19) 11	0.07 (0.01–0.13) 5	0.074 (0.22–0.205) 20	-	0.086 (0.012-0.679) 82
Calna Creek downstream of Hornsby Heights STP	0.01–0.1	2.0 (0.84–3.4) 5	0.17 (0.05–0.36) 11	0.13 (0.07–0.29) 9	0.125 (0.26–0.312) 21	-	0.175 (0.014-1.023) 82

* 19/11/97 – 12/2/98

Table 4.2 Comparison with Results of Previous Studies Oxidised Nitrogen (mg/L)

Monitoring Site	Guideline	1977 (SPCC)	1992-93 (AWT)	1994-95			1994-97 (HSC)
				(AWT)	(HC)	(EPA)	
Berowra Creek at Berowra Waters	0.01–0.1	0.14 (0.005–0.6) 9	0.45 (0.05–0.71) 11	-	-	0.57 (<0.01–2.7) 45	* 0.090 (0.005-0.39) 33
Berowra Creek at Crosslands	0.01–0.1	0.8 (0.05–4.7) 8	3.34 (1.2–7.2) 11	-	-	3.61 (1.3–19.0) 30	
Berowra Creek at Fishponds Waterhole	-	3.5 (2.5–4.9) 2	12.4 (0.87–29) 11	-	11.3 (1.14–22.5) 10	-	11.267 (1.16-23.2) 40
Waitara Creek downstream of West Hornsby STP	-	2.5 (2.0–3.2) 4	20.0 (0.9–36) 11	-	14.0 (1.11–22.7) 20	-	15.375 (1.11–23.94) 82
Calna Creek downstream of Hornsby Heights STP	-	10.0 (5.9 – 16.0) 5	10.4 (0.62 – 29.8) 11	-	29.6 (0.78 – 47.2) 21	-	24.15 (0.31 - 47.2) 82

* 19/11/97 – 12/2/98

Table 4.3 Comparison with Results of Previous Studies
Total Nitrogen (mg/L)

Monitoring Site	Guideline	1977 (SPCC)	1992-93 (AWT)	1994-95 (AWT)	1994-95 (HC)	1994-95 (EPA)	1994-97 (HSC)
Berowra Creek at Berowra Waters	-	0.39 (0.1–0.65) 5	1.05 (0.68–4.7) 11	0.89 (0.6–1.4) 9	-	0.84 (<0.3–3.24) 45	* 0.526 (0.37-0.94) 33
Berowra Creek at Crosslands	-	3.1 (1.15–7.2) 6	3.7 (2.0–6.2) 11	3.0 (0.86-4.7) 9	-	3.56 (<0.3–19.36) 30	
Berowra Creek at Fishponds Waterhole	0.1–0.75	26.2 (25.5–26.9) 2	12.8 (3.2–30.1) 11	9.1 (2.1–23.1) 3	19.1 (2.16–22.9) 10	-	12.475 (2–24) 39
Waitara Creek downstream of West Hornsby STP	0.1–0.75	30.7 (24.2–38.0) 4	20.9 (2.3–37.2) 11	13.8 (0.67–24.8) 5	19.6 (4.48–54.8) 20	-	18.545 (2.24-54.8) 82
Calna Creek downstream of Hornsby Heights STP	0.1–0.75	7.3 (8.5–17.3) 5	13.7 (1.5–30.8) 11	33.7 (22.9-49.5) 9	38.3 (1.37–119.0) 21	-	28.612 (1.04–119) 82

* 19/11/97 – 12/2/98

4.4 Nutrient Budget

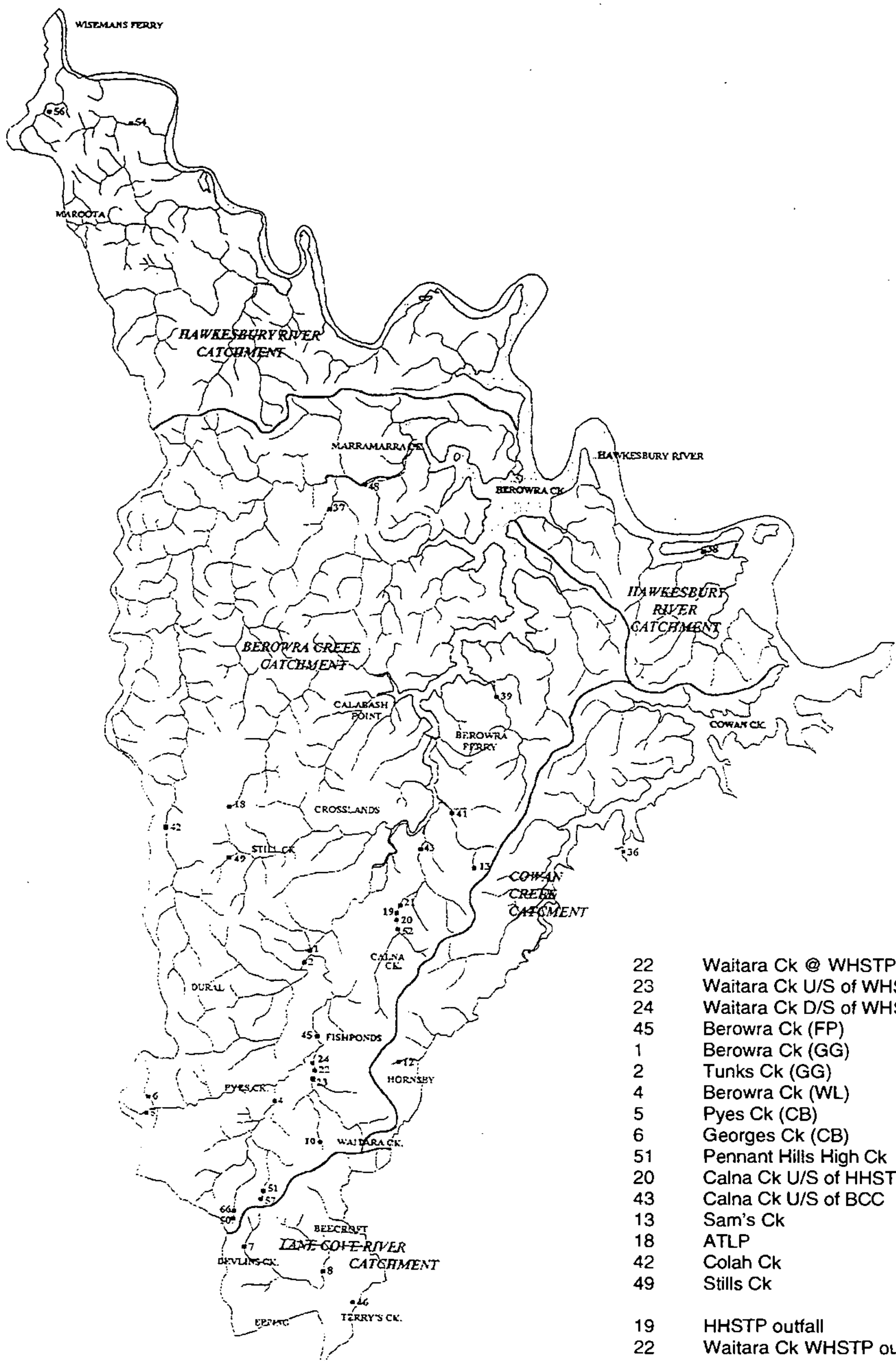
The monthly nutrient budgets for Berowra Creek have been estimated previously from measurements within selected creeks in the catchment and from models (AWT 1997). Results of catchment simulations have been analysed to derive a simple stepped linear regression between load and rainfall (Appendix B).

The regression assumes the load is comprised of a dry weather base flow component and wet weather component dependent upon rainfall. The results of the analysis were then applied to estimate monthly loads over the period December 1997 to February 1998 and are listed in Table B3.

The dry weather loads (Lo in Table B2) indicate the total phosphorus load varies between 0.05 and 0.6 kgP/day for creeks without STP inputs and is 0.91 kgP/day downstream of the Hornsby Heights STP and 1.38 kgP/day downstream of West Hornsby STP. For total nitrogen the loads in non-STP creeks vary between 0.1 and 1.51 kgN/day and downstream of the STPs the dry weather TN loads were 294 kgN/day and 216 kgN/day for West Hornsby and Hornsby Heights STPs, respectively. The dry weather flow is also considerably larger downstream of the STPs (5-10 ML/day) compared with the other creeks were base flows drop to less than 0.2 ML/day and during extended dry periods may cease to flow.

The STP inputs contribute roughly 25% of the total phosphorus and 97% of the total nitrogen during the 1997-98 summer, a relatively dry summer period. The remainder is generated within each of the sub-catchments.

The creeks that drain the urban and semi-rural areas contribute roughly 5-10 times the loads from the natural catchments.



22	Waitara Ck @ WHSTP
23	Waitara Ck U/S of WHSTP
24	Waitara Ck D/S of WHSTP
45	Berowra Ck (FP)
1	Berowra Ck (GG)
2	Tunks Ck (GG)
4	Berowra Ck (WL)
5	Pyes Ck (CB)
6	Georges Ck (CB)
51	Pennant Hills High Ck
20	Calna Ck U/S of HHSTP
43	Calna Ck U/S of BCC
13	Sam's Ck
18	ATLP
42	Colah Ck
49	Stills Ck
19	HHSTP outfall
22	Waitara Ck WHSTP outfall

NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

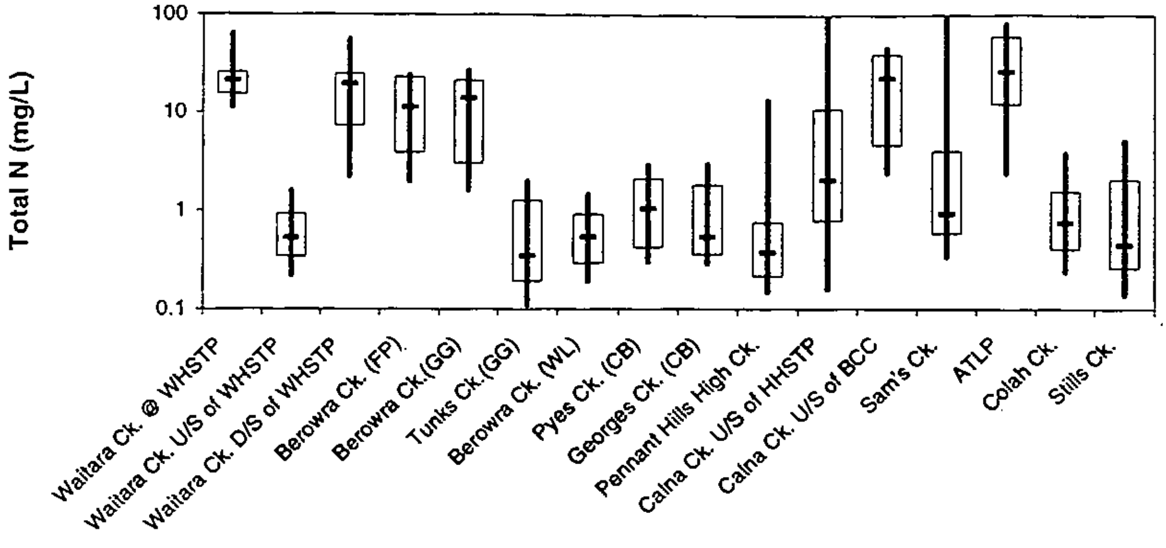
HORNSBY COUNCIL CATCHMENT WATER QUALITY SAMPLING LOCATIONS

MHL Report 928

Figure 4.1

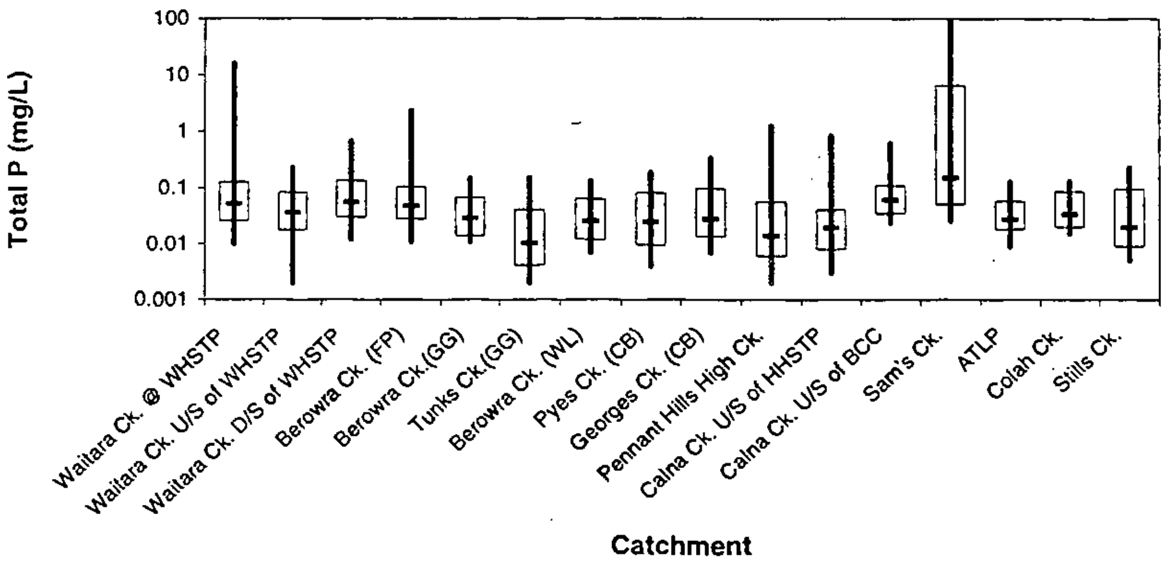
DRAWING 928_4_01.DRW

a)



WHSTP	West Hornsby Sewage Treatment Plant
U/S	Upstream
D/S	Downstream
FP	Fishponds
GG	Galston Gorge
WL	Westleigh
HHSTP	Hornsby Heights Sewage Treatment Plant
CB	Cherrybrook
BCC	Berowra Creek Confluence
ATLP	Arcadia Tip Leachate Pond

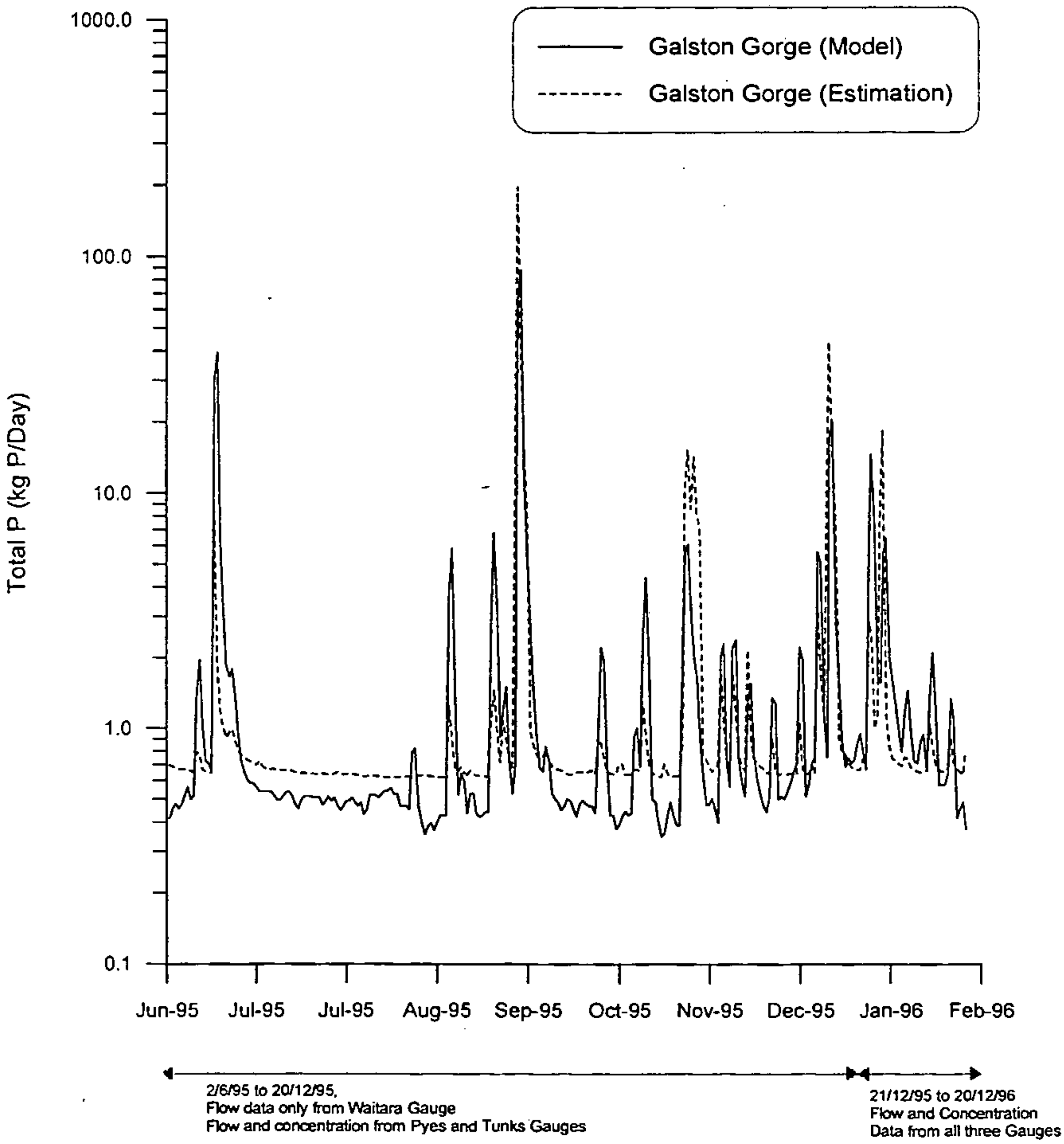
b)



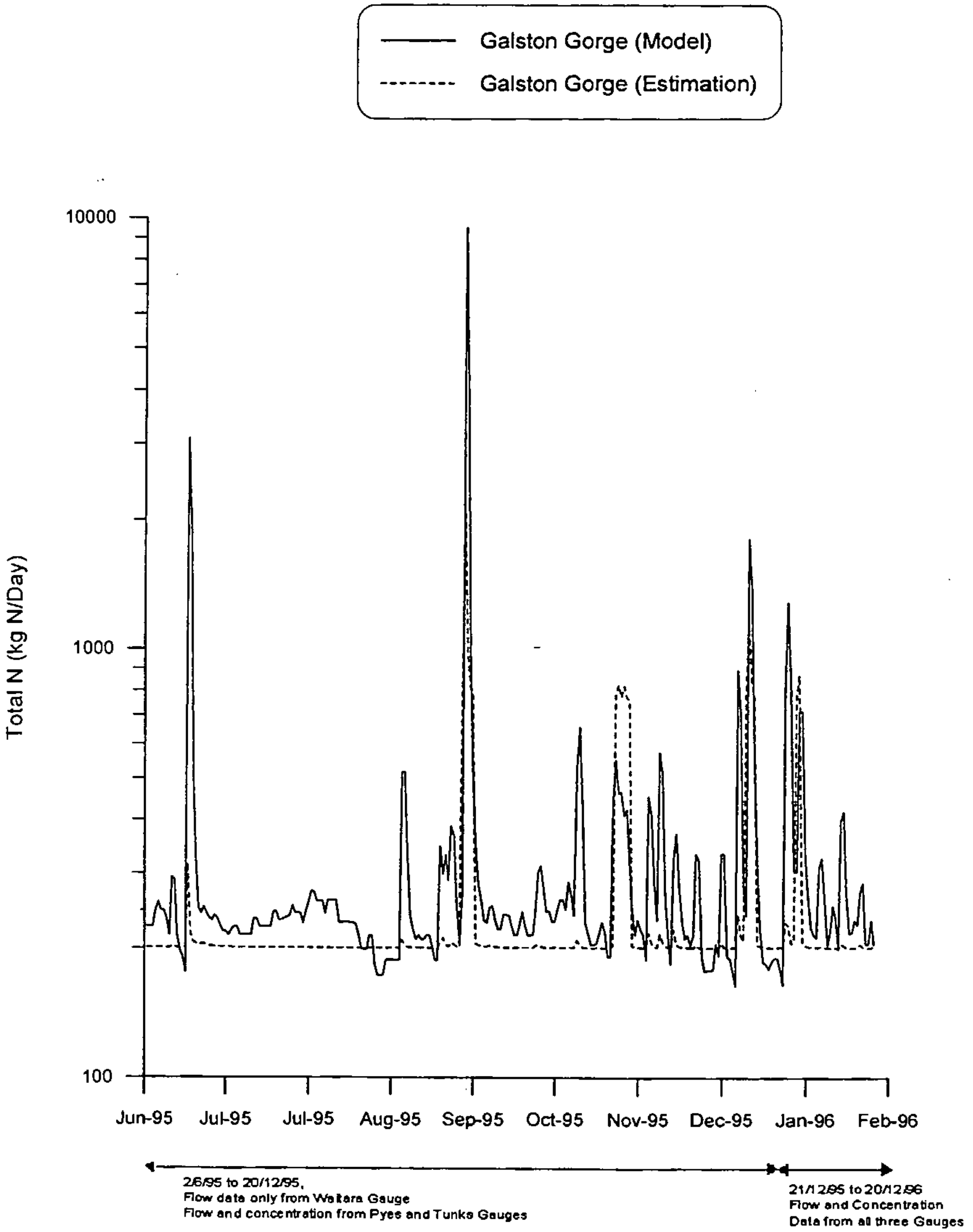
WHSTP	West Hornsby Sewage Treatment Plant
U/S	Upstream
D/S	Downstream
FP	Fishponds
GG	Galston Gorge
WL	Westleigh
HHSTP	Hornsby Heights Sewage Treatment Plant
CB	Cherrybrook
BCC	Berowra Creek Confluence
ATLP	Arcadia Tip Leachate Pond

NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

TN AND TP LOADS AT SITES WITHIN BEROWRA CREEK CATCHMENT



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	TOTAL PHOSPHORUS LOAD AT GALSTON GORGE	MHL Report 928 Figure 4.3



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

TOTAL NITROGEN LOAD AT GALSTON GORGE

MHL Report 928

Figure 4.4

5. Mixing and Flushing Characteristics

5.1 Mechanisms

A number of mechanisms causing currents in the Berowra Creek estuary have been identified. The major energy inputs are derived from tidal flows and fresh water runoff. The presence of stratification and the interaction with topographic effects also plays an important role in determining the currents in the system. A number of data sets have been utilised in the study.

Salt water intrusion along the bottom of the estuary, fresh water overflows and the vertical mixing between these flows is the dominant flushing mechanism. The presence of the STP discharges into the system maintains a relatively high base flow of fresh water. During dry periods this discharge maintains the longitudinal salinity gradient. Higher discharges associated with rainfall/runoff events cause a rapid flushing of the surface waters but only affect the deeper waters during extreme inflow events (> 1 in 5 year rainfall).

5.2 Flushing Estimates

The mixing characteristics in the estuary are influenced by the topography, tidal flows and freshwater inflows. The freshwater inflows determine the level of density stratification and hence the importance of gravitational circulation. A simple analysis of the topographic variations and tidal flow provides useful insight into the potential mixing and flushing characteristics. The depth along the estuary (Figure 5.1a) highlights the deep section near Calabash Point and a number of deep holes separated by sills.

The cross-section area, below the water surface, indicates the larger areas near the deep holes. Assuming a 1 m tide and applying conservation of volume admits an estimate of the depth- and width- averaged velocity at each cross section along the estuary. Multiplying the tidal velocity by the tidal duration (6 hours) provides an estimate of the tidal excursion (Figure 5.1d and e).

Note the weaker velocities near the deep holes imply reduced flushing in this area. Tidal exchange times for different reaches may be estimated by assuming that about $^1/_5$ of the tidal discharge is exchanged over the tidal period. The tidal exchange time may then be estimated as

$$T \sim \frac{V}{.2Q}$$

where V is the volume of the reach and Q the tidal discharge. The volumes were estimated over 4 km reaches to capture the tidal excursion. These estimates are listed in Figure 5.1 e) and highlight the relatively long flushing time near the deep holes.

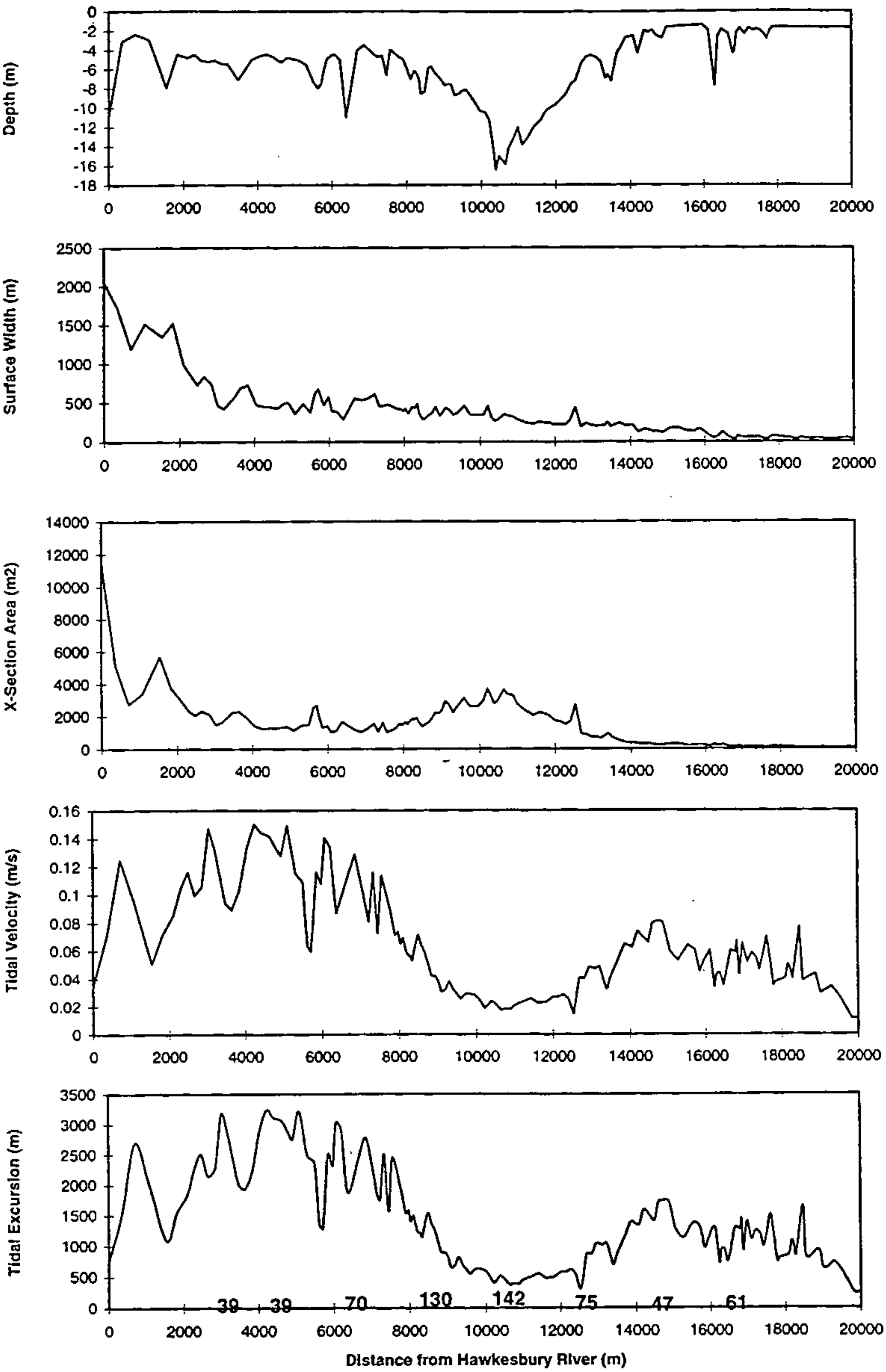
The spring neap cycle (Figure 5.2) also plays a role in the flushing characteristics. The tidal velocity estimates shown in Figure 5.1 assume a tidal range of 1 m which is typical of the estuary. Over the spring neap cycle (Figure 5.2) the tidal range varies from around 0.6 m to 1.8 m. During the lower neaps that occur once per month, the flushing is minimal and hence provided the other factors are favourable the algal blooms may be expected to attain larger biomass than during spring tides when maximal dispersion and flushing will effectively reduce the concentration.

5.3 Summary

Water movements and the exchange of water between the Berowra Creek estuary and the Hawkesbury influence the aquatic ecosystem within the estuary. The water circulation and mixing processes have been investigated through field experiments and interpretation of the collected data. Currents and water level variations were monitored for five months in 1995. During the 1997-98 summer salinity, temperature, dissolved oxygen, underwater light and chlorophyll-a were monitored in a series of intense profiling exercises on 10 occasions. This data was analysed and interpreted by dividing the estuary into a series of boxes along its length and salinity data used to assess the volume of water exchanged between boxes on average over about one week. Results of the analysis provide an estimate of the flushing time (or the time that most of the water in a given section of the estuary resides in that section). This time scale is of fundamental importance for the biotic characteristics of the system as it determines how long water-borne particles, nutrients and micro-algae may reside at one location.

Water circulation is dominated by tidal flows and superimposed over these daily variations is a long-term salt wedge-like exchange flow driven by the fresh water inputs to the creek, as shown schematically in Figure 5.3. The intrusion of saline water (from the Hawkesbury River) into the deeper middle section of the estuary is almost a continuous process. Salty (and hence dense) water from the Hawkesbury intrudes into the estuary and sinks to the deeper holes near Calabash Bay. Tidal action essentially mixes the fresh low salinity inflow water from the catchment and STPs with the saline intrusion to form a brackish mixture. This water appears to be then flushed from the system in the surface layer.

Large fresh water inflows (eg. >1:5 year rainfall) can completely flush the estuary but regular rainfall events generally cause only limited mixing and exchange. These flushing characteristics result in a water residence time within the area near the deep holes of about seven days while the reaches both upstream and downstream of the deep holes are characterised by shorter residence times, typically less than 2 days. These flushing characteristics hence play an important role in determining the distribution of phytoplankton which require reasonably stable (long residence times) conditions to reach bloom proportions.



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES

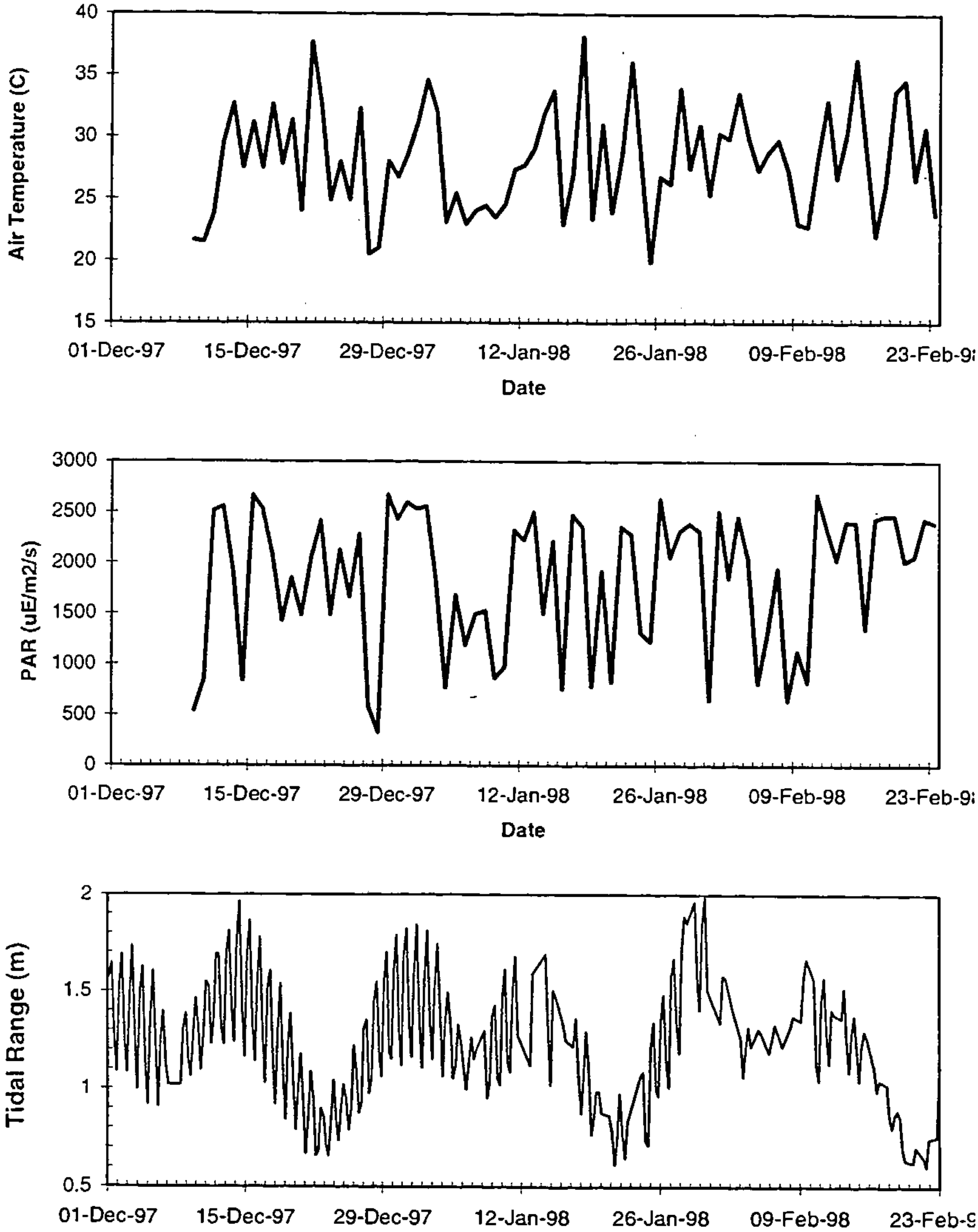
MANLY HYDRAULICS LABORATORY

THALWEG CHARACTERISTICS
BEROWRA CREEK ESTUARY

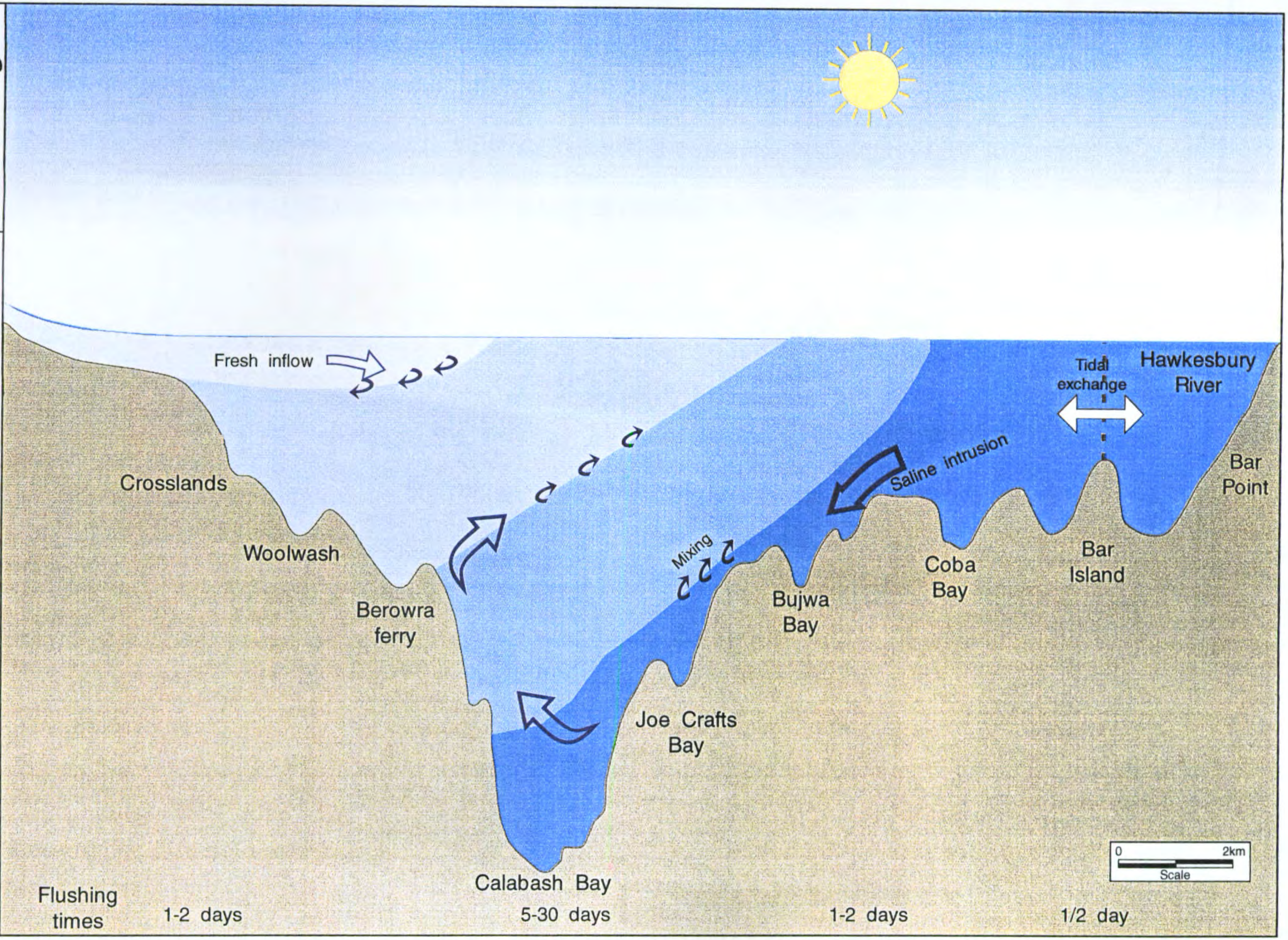
MHL Report 928

Figure 5.1

DRAWING 928_5_1.DRW



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES MANLY HYDRAULICS LABORATORY	AIR TEMPERATURE, SURFACE PAR (AVERAGE 0930 TO 1430) AND TIDAL RANGE AT BEROWRA CREEK	MHL Report 928 Figure 5.2 DRAWING 928_5_02.DRW



NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

BEROWRA CREEK ESTUARY
CIRCULATION AND FLUSHING

MHL Report 928

Figure 5.3

DRAWING 928BER53.CDR

6. Discussion

6.1 Nutrients and Light

The concentrations of dissolved inorganic nutrients in the Berowra Creek estuary are rarely low enough to limit algal growth. In the upper reaches nutrient concentrations in the water are significantly higher than the ANZECC (1992) guidelines and likewise Chlorophyll-a concentrations frequently exceed the guideline levels. The SALMON-Q model and data from the present study indicate that nutrient concentrations are generally greater than the threshold value (or half saturation constants) for uptake by most phytoplankton.

The major source of nutrients is the STP discharges that also provide the major freshwater input. In addition, during wet weather diffuse sources within the catchment form a significant contribution to the nutrient load.

A number of sources in the catchment including animal faeces, excess fertiliser from lawns and ovals detergents and on site sewage treatment, contribute to the overall diffuse load. Sediments are also generally more mobile in urban and cleared rural areas and hence the sediment loads to the estuary are generally higher than pre activity rates. These increased sedimentation rates are generally small and have resulted in only a relatively small increase in the rate of infilling at the estuary fluvial delta.

The SALMON-Q model results suggest that light and temperature are more important for limiting algal growth than nutrients. The present data indicate that the light reaching the water surface is highly variable and on some days of very low light levels the phytoplankton production will be very low.

Typical light extinction coefficients (Figure 6.1) indicate poor light penetration (high extinction coefficients) downstream of Calabash Point and relatively clear waters upstream of the Berowra Ferry. Interestingly, the highest extinction coefficients occur upstream of the bloom, suggesting the high suspended particulates in the waters downstream cause the reduced light penetration. The relatively high particulate (suspended solids) concentrations are derived from resuspension of fine muds that occupy the shallow areas and are mobilised by the tidal currents.

Light penetration is therefore limited to shallow surface layer downstream of about Cunio Point while upstream the penetration is to greater depths and hence may be utilised by the microalgae.

6.2 Flushing and Chlorophyll-a

Flushing of the Berowra Creek estuary is dominated by freshwater inflow events and tidal flows. Flushing times range from and 160 hours in the deep holes at Calabash Point and decrease either side of this reach to less than 50 hours. The vertical stratification following freshwater inflows causes longer retention of water in the deep holes where deoxygenation occurs. Tidal pumping of more dense water from downstream effectively replaces the deeper waters within the holes, thereby improving the oxygen content of deep water.

In the algal bloom during the day photosynthesis results in supersaturation of the surface water dissolved oxygen (Figure 3.9).

Freshwater inflows to the Creek are dominated by the STP discharge (15-20 ML/day) during dry weather. This inflow is significantly larger than the pre-urbanisation base flow which is estimated to be about 0.3 ML/day. Hence the increase in freshwater input has effectively pushed the salinity gradient downstream such that the upper reaches of the Creek (upstream of Crosslands) are exposed to much lower salinities than prior to the effluent input. This may explain the disappearance of oysters from these reaches of the creek.

The continual input of freshwater and nutrients during dry weather is probably more important for maintaining the algal blooms than a single large inflow (much of which will flush through the system). It is suggested that breaking the continual flow during dry weather may be more effective at controlling the blooms through nutrient limitation, than trying to reduce the nutrient loads. This could be achieved through effluent re-use or diversion.

6.3 Phytoplankton

Phytoplankton communities at Berowra were dominated by diatoms, shifting from *Chaetoceros* during November-December and *Thalassiosira* and *Pseudonitzschia* during January-February. The shift in these communities is probably the result of natural succession.

The differences between upstream, downstream and bloom communities were driven by concentrations of the dominant taxa in the community and by the presence or absence of certain taxa in low abundance.

Importantly, certain phytoplankton taxa have been reported to be harmful to humans in some circumstances. Unfortunately however no clear-cut correlations between algal concentrations and their potential harmful effects have been reported (Hallegraeff 1993). Three of the dominant taxa identified during the study, *Chaetoceros*, *Nitzschia pungens* and *Pseudonitzschia* and the additional taxa identified in Figure 3.17 are potentially harmful taxa. *A.Chaetoceros* species has been identified as being potentially harmful and there is no evidence that this specific taxon is dangerous. The possibility that it may be harmful cannot be discounted however.

Two taxa that are problematic are *Nitzschia pungens* and *Gymnodinium catenatum*. These taxa are known toxic algae. *Nitzschia pungens* is known to produce toxins causing amnesic shellfish poisoning and has been found in Berowra previously (Hallegraeff 1995).

Many algal taxa are able to produce cysts which can remain dormant in the sediment for many years and then bloom when conditions are favourable. This is essential to take into account in any management of the estuary and regular monitoring should be undertaken.

The algal blooms in Berowra Creek are similar to the blooms observed in the Hawkesbury River and in the Brunswick River near Mullumbimby. Both these rivers have significant effluent discharges that have been linked to the increase in primary production and algal blooms. It is interesting that the blooms have been largely confined to the mid reaches of the

Creek and to date have not been observed in Marramarra Creek. This is most likely due to lower nutrient loads entering Marramarra Creek and also the more turbid water resulting from resuspension on the tidal mud floats in Marramarra Creek.

It must be remembered that phytoplankton are an important component of the ecosystem. Management of the system requires a careful balance between reducing the nuisance blooms but maintaining the food source for secondary producers such as zooplankton, oysters and fish.

6.4 Zooplankton

Zooplankton biomass was consistently greatest in the bloom area (Figure 3.14). Zooplankton abundance and zooplankton biomass were significantly related to phytoplankton biomass. However, the relationship was not as close as expected because of the lag in response of zooplankton to increased or decreased phytoplankton biomass.

There was no evidence of any physical dynamics such as estuarine fronts, tidal fronts and surfacing internal waves, that may have concentrated phytoplankton or zooplankton at the bloom area during this study. In addition, primary productivity was assessed in the estuary, data indicated that it was significantly greater within the bloom area than upstream or downstream. It is therefore most likely that the increased zooplankton within the bloom was a direct response to increased food availability.

Phytoplankton within the three sampling sites were dominated by one of two taxa during the course of the study: *Chaetocerus* during the December period and *Thallasiosira, Pseudonitzschia* and *Skeletonema* during February.

Integration of data over the study period indicate that biomass of small zooplankton and phytoplankton were greatest at the Berowra Waters-Calabash Bay region of the estuary. The relationship between phytoplankton biomass and zooplankton biomass, while significant, was fairly weak. When considered over the time of study however, it is clear that there was a lag between response times of zooplankton to variation in phytoplankton biomass. Similar results (even negative relationships) have been reported in other studies and have been used to infer top down control of phytoplankton by zooplankton, or increase of primary production when zooplankton were not present in sufficient concentrations to exert any form of control on the biomass (Harvey et al. 1935, Martin 1965, Nival et al. 1975).

There was distinct temporal variability in phytoplankton biomass at each of the three sampling sites. Zooplankton appear to respond to increases or decreases in phytoplankton biomass. There was a lag effect in the timing of the response, probably the result of slower turnover times/stage times associated with zooplankton (days) compared to phytoplankton (hours).

Effect of Grazing
Zooplankton grazing can affect phytoplankton communities in a number of ways. Rapid responses of small zooplankton taxa and small sized stages to increased food availability can affect the size structure of the zooplankton community. Zooplankton grazing can exert top down control on phytoplankton numbers, to the extent of being able to control phytoplankton blooms in some systems. Selective zooplankton grazing can alter the community composition of phytoplankton and can enable less nutritive taxa to become more dominant

Small zooplankton taxa such as *Oithona* are important indicators of changed food availability in estuaries. They have rapid generation times, quick turnover of different stages and have been reported to have fast responses to changes in food availability. During this study, the bloom area had consistently greater concentrations of small particles than the upstream or downstream reaches. This suggests that increased secondary production and the rapid response of small zooplankton to increased food abundance were causing a change in the particle size structure of the estuary. Nutrient enrichment of aquatic systems has been reported to change the size distributions of the biomass of those systems (Kerr and Ryder, 1998).

6.5 Conceptual Model of Water Quality

Water quality in Berowra Creek is influenced by inflows from the catchment, from water exchange with the Hawkesbury River and by bio-geochemical cycling within the estuary.

Urbanisation of the catchment has led to an increase in suspended sediment and nutrient loads to the estuary. During rainfall events these pollutants are washed into the creek from diffuse sources such as urban and rural areas. They may also be discharged directly into the creek from point sources such as from the two sewage treatment plants within the system. During intense rainfall events, bypassing of the STPs contributes directly to the bacterial load to the estuary.

Nutrient enriched water and suspended sediments from the Hawkesbury River affect the lower reaches of the creek where water from the Hawkesbury River mixes with the waters in the creek. Fine sediments are delivered to the Creek from the Hawkesbury River. It is not clear whether the Hawkesbury water contains higher concentrations of nutrients than the Berowra Creek water and hence whether the Hawkesbury water is a source of nutrients.

These water quality impacts also influence the estuarine processes within Berowra Creek.

Pollutant and hydraulic loads affect the bioavailable nutrients, which are integrated by and therefore affect the macroalgae, phytoplankton and zooplankton communities. Phytoplankton, as well as being dependent on bioavailable nutrients, are also affected by environmental factors such as light and temperature. The major source of nutrients to the Berowra Creek are the STP discharges. As the nitrogen reduction program is implemented the relative contribution of the STP's will decrease and the catchment load will become more significant.

Phytoplankton blooms respond relatively quickly to changes in water quality and are therefore a good indicator of the condition of the estuary. The observed chlorophyll-a concentrations in the vicinity of the Berowra Ferry are typical of eutrophic systems.

6.6 Microbiological Pollutants

There are several sources of microbial pollutants that could effect waters within the study area. These include; effluent from the STP's, sewage overflows, sewage from residential or commercial on-site sewage systems, effluent discharge from boats, and animal faecal runoff from the catchment, notably from dogs.

Effluent from the two STP's in the study area is of reasonable quality with both plants using an activated sludge process with tertiary treatments including filtration, disinfection by chlorination, nitrification, and chemical phosphorus removal.

Sewage overflows can include exfiltration through broken pipes, reticulation surcharges, blockages etc, and generally only occur after medium to large rain events As sewerage overflow is untreated it can contain $10^7 - 10^8$ cfu/100ml. High levels of faecal coliforms have been measured downstream near Calabash Bay but generally do not extend to the lower reaches. Primary contact recreation (i.e. swimming) is not recommended until four days after rain, and near Crosslands swimming is generally not recommended.

Pollution from on-site systems such as septic tanks can also occur within the catchment. Although septic tanks only provide a low quality of treatment, which generally does not adequately reduce bacteria or pathogens, their low discharge means that effects are localised. Effluent discharge from boats however, can be problematic in areas where boating occurs close to swimming areas or oyster beds. These issues do not appear to be significant in Berowra Creek but should be monitored to assess their potential.

A number of human viruses and parasitic protozoa throughout the world are associated with possible swimming related illness, such as Enterovirus (*polioviruses, coxsackie*), hepatitis A., adenviruses and reoviruses, as well as *Cryptospoidium parvium* and *Giardia lamblia*. These viruses and protozoa are derived from faecal coliform pollution within the water. Treatment of sewage before discharge, however generally results in a shorter life span of such coliform bacteria.

Webb, Mckeown & Associates (1996) found that faecal coliform levels were highly variable across the Berowra Creek catchment, the highest levels were measured at upstream sites, whilst the lowest were measured at the estuarine sites. The upstream sites exceeded the ANZECC (1992) guideline for primary contact in recreational waters, 150 cfu/100ml, more often than the STP sites into which sewage effluent is discharged (Webb, Mckeown & Associates, 1996). The maximum levels measured at the estuarine sites did not exceed the guideline value for primary recreation, however, with the exception of the creek entrance area, the maximum recorded levels exceeded the ANZECC (1992) guidelines for the Protection of Fish, Crustacea and Shellfish (14 MPN/100ml)

It should be noted, that the faecal coliform levels from August and October 1995 were lower than the corresponding mean 1992/1993 values, particularly at the upstream sites, these 1995 levels were generally less than 200 cfu/100ml, with several non-detect values recorded Webb, Mckeown & Associates (1996).

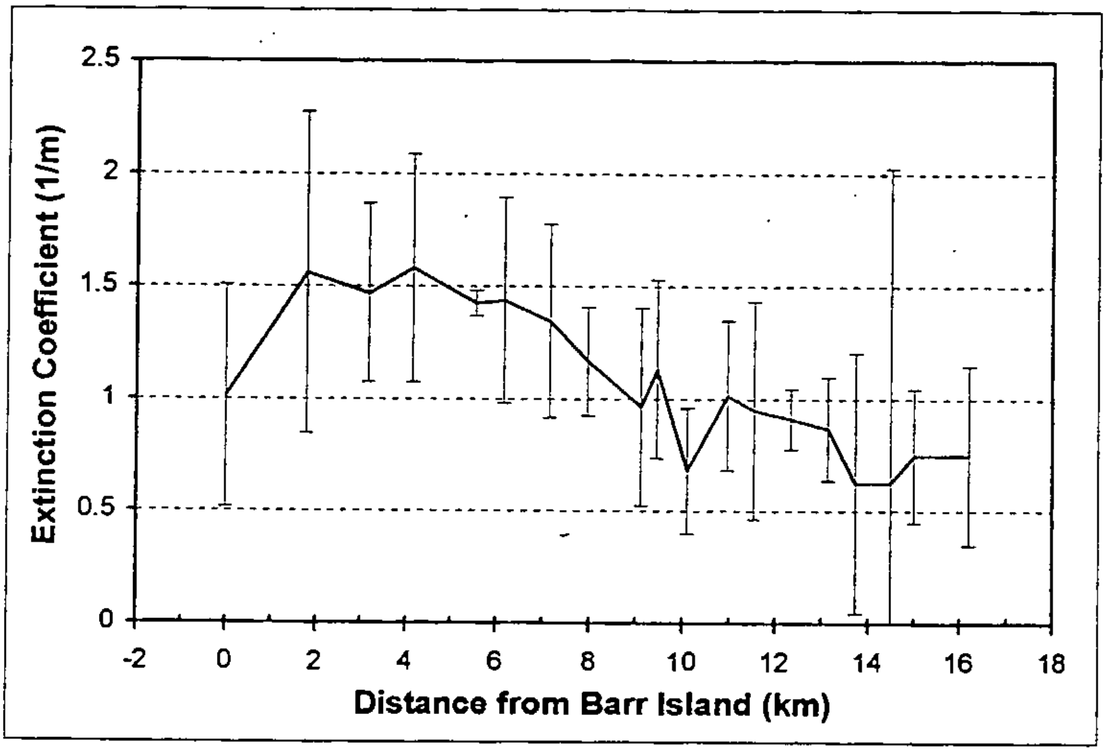
6.6.1 Commercial Interests

Commercial fishing and oyster leases in Berowra Creek are located at the downstream end of the estuary, away from possible causes of organic pollution (from the STP's and urban runoff). The proximity of the Hawkesbury River ensures that the oyster beds are well flushed by tides. Dinoflagellate blooms recorded in Berowra Creek would not appear to have been so severe as to affect oyster beds or estuarine fish habitats in Berowra Creek (Webb, Mckeown & Associates, 1996).

Oysters feed on micro-organisms from the water through their gills, subsequently pollution can greatly affect the oyster. Purification of oysters before sale is required by law and has proved to be highly effective. All oysters from Berowra Creek undergo this cleaning or 'depuration' process where the oysters are held in ozonated water for 36 hours before marketing. Berowra Creek oysters are depurated at Sandbrook Inlet, off the main Hawkesbury River. Given that the water used in the depuration process itself is free from toxic dinoflagellates, this procedure has generally been successful in reducing the occurrence of poisoning due to the ingestion of cultivated oysters (MHL-855, 1998b).

The processes that control the phytoplankton blooms are schematised in Figure 6.2. Nutrient inputs provide a continuous food source for phytoplankton and when there exists sufficient light and optimal temperature the phytoplankton populations can multiply very quickly. This rapid growth is then countered by the flushing effect and zooplankton grazing. The existence of the deep holes influences the tidal flow such that a typical water residence time within this region is approximately seven days whilst upstream and downstream the residence times are considerably shorter.

The concentrations of phytoplankton have an effect on small zooplankton which respond to changes in food availability. Increased zooplankton abundance can result in grazing pressure (top down control) of phytoplankton. In Berowra Creek zooplankton only appear to exert top down control when phytoplankton productivity is slowed by changes in other environmental factors (such as reduced light).



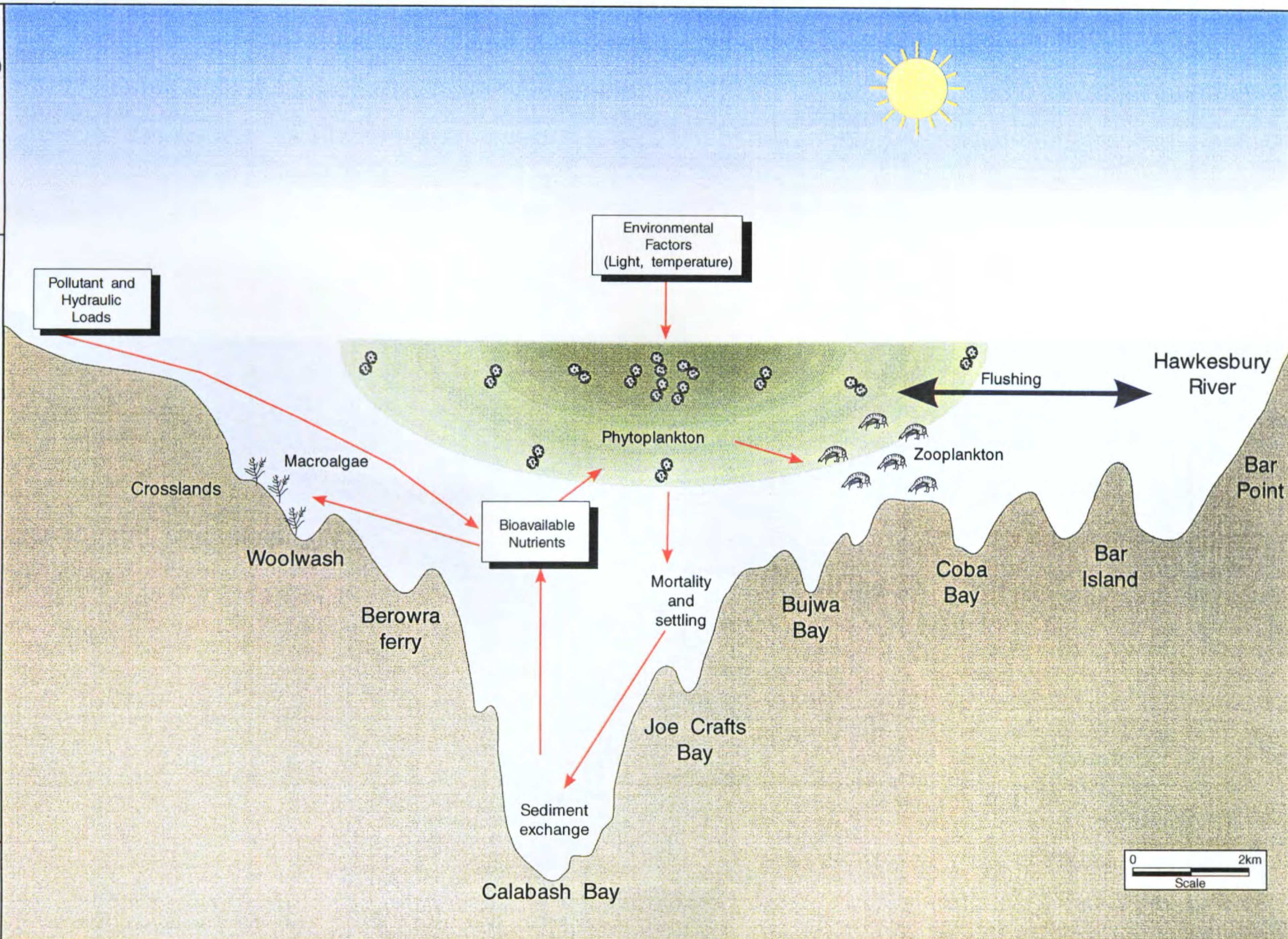
NSW DEPARTMENT OF PUBLIC WORKS AND SERVICES
MANLY HYDRAULICS LABORATORY

EXTINCTION COEFFICIENT VARIABILITY ALONG BEROWRA CREEK ESTUARY - SUMMER 97-98

MHL Report 928

Figure 6.1

DRAWING 928_6_01.DRW



NSW DEPARTMENT
OF PUBLIC WORKS
AND SERVICES
MANLY HYDRAULICS LABORATORY

ALGAL BLOOM PROCESSES

MHL
Report 928

Figure
6.2

DRAWING 928BER62.CDR

7. References

Anderson, D.M., S.B. Galloway, and J.D. Joseph 1993, 'Marine Biotoxins and Harmful Algae: A National Plan', *WHOI Technical Report* 93-02, Woods Hole Oceanographic Institution, Woods Hole, MA. (44 pp).

Anderson, D.M. (Chair) 1994, 'ECOHAB The Ecology and Oceanography of Harmful Algal Blooms A National Research Agenda', *A workshop sponsored by the Division of Ocean Sciences of the National Science Foundation and by the Coastal Ocean Program of the National Oceanic and Atmospheric Administration,* Woods Hole Oceanographic Institution, Woods Hole, MA. (44 pp).

ANZECC 1992, *Australian Water Quality Guidelines for Fresh and marine Waters,* Australian and New Zealand Environment and Conservation Council.

AWT Ensight 1997a, *Berowra Creek Model Recalibration* Report No 97/43 prepared for Inland Waste Water, Sydney Water.

AWT Ensight 1997b, *Summary of Water Quality Modelling Results for Berowra Creek EIS* Report No 97/44 prepared for Inland Waste Water, Sydney Water.

AWT Ensight 1997c, *Pollutant Loads to Berowra Creek from Pyes, Tunks and Waitara Creeks, 1995-1997* Report No 97/219 prepared for Hornsby Shire Council.

Bays, J.S. and Crisman, T.L. 1983, Zooplankton and trophic state relationships in Florida lakes, *Can. J. Fish. Aquat. Sci.* **40**, 1813-1819.

Hallegraeff, G.M. 1993, A review of harmful algal blooms and their apparent global increase, *Phycologia*, 32, 79-99.

Hallegraeff, G.M. 1995, Species of the diatom *Pseudonitzschia* in Australian waters, *Botanica Marina,* 37, 397-411.

Harvey, H.W., Cooper, L., Lebour, M.V. and Russel, F.S. 1935, Plankton production and its control, *J.Mar.Biol.Ass.UK.*, 20, 407-442.

Lund, J.W., Kipling, C. and Le Cren, E.D., 1958, An inverted microscope method of estimating algal numbers and the statistical basis of estimations by counting. *Hydrobiologia*, 11, 143-170.

Kerr, S.R. and Ryder, R.A. 1998, *A review of environmental issues for management.* Unpublished Report by Hornsby Shire Council.

Manly Hydraulics Laboratory 1998a, *Berowra Creek Water Quality Data Collection November 1997 to February 1998*, Report No. MHL917, NSW Department of Public Works and Services.

Manly Hydraulics Laboratory 1998b, *Berowra Creek Estuary Processes Study Review and Interpretation of Existing Data* MHL Report 855, undertaken for Hornsby Shire Council and the Department of Land and Water Conservation.

Manly Hydraulics Laboratory 1998, *Berowra Creek Estuary Processes Study: Hydrodynamics and Flushing*, NSW Department of Public Works and Services, Manly Vale, NSW.

Martens, D.M. and Warner, R.F. 1994, *Impacts of On-site Domestic Wastewater Disposal in Sydney's Unsewered Areas*, Department of Geography, University of Sydney, Camperdown, NSW.

Martin, J.J. 1965, Phytoplankton-zooplankton relationships in Narragansett Bay, *Limnol. Oceanog.*, 10, 185-191.

Rissik, D., Suthers, I.M. and Taggart, C.T. 1997, 'Enhanced zooplankton abundance in the lee of an isolated reef in the south Coral Sea. The role of flow disturbance'. *J. Plankton. Res.*, 19, 1347-1368.

Steele, J.H. 1965, Notes on some theoretical problems in production ecology, in Primary Production in Aquatic Environments, C.R. Goldman (ed.) University of California Press, Berkeley, California, pp393-8.

Sprules, W.G., Casselman, J.M. and Shuter, B.J., 1983. Size distribution of pelagic particles in Lakes. *Canadian Journal of Fish and Aquatic Science*, 40, 1761-1769.

Tsuruya H. and Hibino T. 1998a, 'A Study of Anoxic Water Structures Generated in Deep Bay Enclosed by Tsunami Breakwaters'.

Tsuruya H. and Hibino T. 1998b, 'Water Circulation System in Enclosed Bay and its Utilisation for Water Quality Management', *The First Joint Meeting of the CEST Panel of the UJNR.*

Uye, S. 1994, Replacement of large copepods by small ones with eutrophication of embayments: cause and consequence, *Hydrobiologia*, 192/293, 513-519.

van Senden D, Howells L and Anderson P 1996, 'Lake Ainsworth Processes Study' Australian Water And Coastal Studies Pty Ltd Report 96/07, for Ballina Shire Council, Department of Land and Water Conservation and Department of Sport and Recreation.

Wallingford Software 1992, SALMON-Q User Manual (Beta Release), HR Wallingford, Wallingford, UK.

Webb, McKeown & Associates Pty. Ltd., 1996, *Overflows from West Hornsby – Berowra Creek Sewerage System – Approved Draft Options Report*, Sydney Water.

Appendix A

Zooplankton Size and Biomass Analysis

Appendix A Zooplankton Size and Biomass Analysis

Methods

Study Site

Berowra Creek estuary is a drowned river valley which joins the Hawkesbury River estuary approximately 24 km from the Pacific Ocean (Figure 1). Berowra Estuary has an approximate waterway area of 13 square kilometres and drains a catchment area of approximately 310 square kilometres. The estuary is the receiving water for tertiary treated sewage discharge from two sewage treatment plants. A number of stormwater drains also discharge into the estuary. Environmental data for surface water at each sampling period are listed in Table 1.

Field Sampling

Sampling took place during eight cruises on the estuary during December 1997 and February 1998. Plankton were captured using a 100 µm mesh net with a 20 cm diameter opening towed at the surface at approximately 1.5 m/s. A General Oceanics flow meter was used to assess the volume sampled during each tow. The average volume filtered during each tow was 2.5 cubic metres. Three sites were sampled within the estuary. These were at Arcadia (upstream of the phytoplankton bloom location), at Calabash (phytoplankton bloom) and downstream of the bloom at Kimmerikong (Figure 1). Three replicate samples were obtained at each site. Sampling took place during daylight at low tide slack to prevent contamination of samples with organisms from the Hawkesbury River Estuary (Fig 1) that may have been transported upstream during high tides. Samples were immediately preserved in 5% formalin/95% estuarine water.

Three random algal samples were collected af each station for counts and identification. These were immediately preserved in 3% Gluteraldehyde and 97% estuarine water. A Seabird Conductivity Temperature and Depth recorder was used to collect water quality data (temperature, salinity depth and dissolved oxygen) from selected stations along the estuary (Fig. 1). The instrument was also fitted with a Seatech Fluorometer which was used to collect data of chlorophyll a biomass. Water samples were collected at random throughout each cruise and assessed for chlorophyll a concentration in the laboratory following the methods of Strickland and Parsons (19xx). These data were used to calibrate the fluorometer data.

Laboratory

Phytoplankton

The 100 mL phytoplankton samples were concentrated to 10 mL by sedimentation. Concentrations were obtained using a calibrated Lund Cell (Lund 1958) viewed under a compound microscope.

Zooplankton

Plankton samples were gently rinsed through a 90 µm mesh sieve. The remaining sample was rinsed into a beaker and standardised to 100 mL with the addition of water. The sample was stained with lactophenol cotton blue for 48 hours. This stained all particles a deep blue colour enabling a greater contrast between particles and the white background of the microscope to be obtained. Three 1 mL sub-samples were taken from each of the 100 mL samples and placed into individual petri dishes. These were diluted with 9 mL of water.

Images of each sample were captured using a Phillips CCD camera with a macro lens and a framegrabber card. Particle size analysis was conducted using NIH Image software through a Macintosh computer. Seven fields of view were analysed from each petri dish. This method was conducted on each of the three sub-samples and then averaged.

A procedure, written for the NIH Image program enabled the total number of particles in each field of view to be counted and the area of each particle to be calculated. Areas were converted to equivalent spherical diameter (esd). Esd is the diameter across a sphere which has the same area as the area of the organism being studied. Data within the size range of interest in the experiment (90-900 µm esd) were grouped into 20 equal particle size classes (eg Sprules et al. 1983, Rissik et al. 1997). The concentrations of each particle size class were standardised to number of organisms per cubic meter. Zooplankton biomass was estimated by calculating using the volume of each particle size class. It is assumed that as zooplankton are neutrally buoyant, the biomass can be calculated relating volume to mass.

To assess the diversity of the zooplankton community, Samples were thoroughly homogenised and two replicate subsamples were taken. The first 400 particles were identified and counted. Counts included detritus which enabled an estimation of the amount of detritus, an important component of estuarine water, to be obtained. The diversity and the percentage composition of each community.

To enable the particle size spectrum to be characterised, the first 50 individuals of each taxon were identified and areas were measured using the image analysis system. The areas were converted to equivalent spherical diameter.

Appendix B

Berowra Creek Catchment Pollutant Load Predictions

Appendix B Berowra Creek Catchment Pollutant Load Predictions

B1 Introduction

Water quality in the Berowra Creek Estuary is heavily influenced by pollutant loads coming from the upstream catchment. Phosphorus and nitrogen loads into the estuary are of particular concern, as they may force the estuary to enter a eutrophic state, hence increasing the likelihood of harmful algal blooms occurring.

Pollutant loads from catchments are subject to a very large number of influences which may be difficult to quantify. Probably the two most important of these are rainfall and land use.

From MHL (1998) previous water quality information for Berowra Creek of relevance here include the following.

- Monthly measurements of phosphorus and nitrogen concentrations in Berowra Creek and its tributaries undertaken by Hornsby Shire Council.
- Water quality modelling undertaken by AWT-EnSight (1997a, 1997b) for the whole Berowra Creek catchment.
- Pollutant load data obtained by AWT-EnSight (1997c) over the period 1995 to 1997 in three tributaries of Berowra Creek (Pyes, Tunks and Waitara creeks).

Of these sources, the water quality modelling performed by AWT-EnSight provides the most comprehensive assessment of water quality in the catchment; drawing on the data gathered by the other sources listed above, and estimating loads where no data is available, based on the available data.

The aims of the present report are to:

- Compare the model results directly against pollutant load data;
- Determine if there are any simple relationships that can be derived that reasonably reproduce the model results; and
- Determine values for phosphorus and nitrogen loads from the Berowra Creek sub-catchments over the period December 1997 to February 1998 to allow water quality modelling of the estuary to be performed.

B2 Methods

B2.1 Comparison of Loads - Model vs Data

Calibrations undertaken by AWT-EnSight were performed by varying model coefficients within HSPF to ensure that flow rates and concentrations of total phosphorus and total nitrogen gave good agreement with data collected for these parameters at Waitara, Pyes and Tunks creeks.

The calibrations performed gave reasonable results; however, when considering the impact of pollutants from a catchment on a downstream water body (in this case the Berowra Creek estuary), it is necessary to ensure that the nett fluxes (loads) of total phosphorus and total nitrogen to that water body are in good agreement.

The net fluxes of nitrogen and phosphorus respectively calculated from the HSPF model were checked against the data collected by AWT-EnSight for the period June 1995 to February 1996 in Waitara, Pyes and Tunks creeks. To perform these checks, loads in the creeks were obtained from the AWT flow rate and concentration data. Estimates were then obtained from the load data of the pollutant load at Galston Gorge using the formula shown in Equation 2.1.

$$L_G = (L_W + L_P + L_T)\frac{A_G}{A_W + A_P + A_T} + L_{STP}$$

where L_G is the pollutant load (phosphorus or nitrogen) at Galston Gorge
L_W is the pollutant load at Waitara Creek
L_P is the pollutant load at Pyes Creek
L_T is the pollutant load at Tunks Creek
L_{STP} is the pollutant load from the West Hornsby STP
A_G is the catchment area (phosphorus or nitrogen) at Galston Gorge
A_W is the catchment area at Waitara Creek
A_P is the catchment area at Pyes Creek
A_T is the catchment area at Tunks Creek

L_W, L_P *and* L_T were calculated as measured flow multiplied by average dry weather concentration through dry periods and as measured flow multiplied by event mean concentration for wet weather flows.

L_{STP} was calculated from the STP flow rates multiplied by the STP concentration of TP/TN. During dry weather periods, the average dry weather flow (10 ML/day) was taken for STP flow rates. For wet weather periods, the STP flow was assumed to be three times the average dry weather flow.

Figures 4.3 to 4.4 show that the estimated loads at Galston Gorge from the model and from the data collected are in good agreement, given the number of assumptions that had to be made regarding the flows and concentrations from the STP. The predictions that the model makes for the other sub-catchments can therefore be expected to be reliable. However, it is desirable to have a more easily tractable way to estimate catchment loads for Berowra Creek rather than using the model. To this end, two regression techniques were used to try to relate the daily loads to daily rainfall for each sub-catchment. The techniques used were simple linear correlation and linear correlation with a step function at low rainfall. These are discussed separately below.

B2.2 Linear Regression on AWT data

Table B1 shows details of a simple linear regression performed between total phosphorus and total nitrogen loads estimated by the HSPF model and daily rainfall data from the Bureau of Meteorology rain gauge at Berowra for the 10 model inputs to the Berowra Creek estuary for the period 1 June 95 to 30 November 95. It can be seen from the correlation coefficients in Table B1 that the linear regression relationship did not give reliable estimates of the TP and TN loads.

B2.3 Stepped Regression on AWT Data

Table B2 shows details of a stepped regression performed on the same data sets as above. The stepped regression relationship can be described as:

$$L(i) = \begin{cases} L_0 & i \le i_0 \\ L_1 + m(i - i_0) & i > i_0 \end{cases}$$

where: $L_j(i)$ is the daily pollutant load (the subscript j represents phosphorus or nitrogen) as a function of the daily rainfall i

L_{j0} is the dry weather daily pollutant load

L_{j1} is the minimum daily pollutant load for a wet weather event ($L_{j1} > L_{j0}$)

i_{j0} is an initial rainfall loss parameter

m_j is the rate of increase in daily pollutant load with change in rainfall.

This relationship can be justified on physical grounds, as for very small rainfall events there will be very little or no runoff generated, and therefore no pollutants will leave the sub-catchment, so up to some value of the rainfall i_{j0} the pollutant load entering the creek will not change from dry weather conditions. For rainfall events larger than i_{j0}, some increase in pollutant load may be expected above dry weather events, and an increase in rainfall should result in an increase in pollutant load.

To account for these factors, i_{j0}, L_{j0}, L_{j1} and m_j are treated as variables, and an optimisation was performed find the values of these parameters that would minimise the difference between pollutant loads predicted by the model, and predicted by this relationship for a given rainfall event. Table B2 shows that a reasonable relationship between pollutant loads and rainfall was able to be obtained using this technique.

Table B1 Simple Linear Regression

		Berow3# 2 Galston Gorge	Berow1# 15 Calna Creek (Confluence)	Berow2# 4 Crosslands	Berow1# 11 Cunio Pt	Berow1# 4 Coba Pt	Berow1# 2 Marramarra Confluence	Berow1# 9 Oaky Pt	Berow1# 6 Ants Nest Pt	Berow1# 8 Bujwa Bay	Calna2# 1 Calna Creek (Tidal Limit)
TP	y int	0.96	0.32	0.36	0.21	0.04	0.37	0.03	0.03	0.09	0.83
(kg/day)	grad	0.56	0.15	0.10	0.19	0.02	0.22	0.03	0.10	0.05	0.13
	R^2	0.24	0.26	0.17	0.25	0.37	0.46	0.28	0.36	0.24	0.27
Average daily load 1 June to 30 Nov 95		2.19	0.66	0.58	0.64	0.08	0.87	0.09	0.26	0.20	1.11
TN	y int	278.46	0.09	0.09	0.64	0.06	-0.44	0.09	0.12	0.39	206.42
(kg/day)	grad	21.56	0.10	0.10	1.06	0.06	1.58	0.09	0.22	0.37	11.09
	R^2	0.05	0.25	0.25	0.13	0.16	0.38	0.13	0.24	0.10	0.21
Average daily load 1 June to 30 Nov 95		325.83	0.31	1.61	2.98	0.19	3.06	0.30	0.62	1.20	231.00

Table B2 Stepped Linear Regression

		Berow3# 2 Galston Gorge	Berow1# 15 Calna Creek (Confluence)	Berow2# 4 Crosslands	Berow1# 11 Cunio Pt	Berow1# 4 Coba Pt	Berow1# 2 Marramarra Confluence	Berow1# 9 Oaky Pt	Berow1# 6 Ants Nest Pt	Berow1# 8 Bujwa Bay	Calna2# 1 Calna Creek (Tidal Limit)
TP	L_0	1.38	0.39	0.40	0.35	0.06	0.60	0.05	0.12	0.10	0.91
(kg/day)	m	0.87	0.14	0.09	0.19	0.03	0.48	0.04	0.19	0.05	0.10
	L_1	2.23	1.32	1.16	1.28	0.15	1.74	0.19	0.52	0.20	2.23
	i_0	13.17	4.00	4.00	4.00	22.00	23.15	14.00	18.07	1.00	4.00
	r^2	0.73	0.75	0.83	0.75	0.56	0.44	0.69	0.57	0.75	0.76
TN	L_0	294	0.17	0.82	1.51	0.10	1.08	0.17	0.28	0.68	216.29
(kg/day)	m	35	0.16	0.95	1.83	0.10	3.40	0.16	0.33	0.63	14.45
	L_1	294	0.17	3.21	5.96	0.38	6.12	0.60	1.24	2.41	462.00
	i_0	13	13.27	16.00	16.72	16.00	22.00	16.00	13.27	16.44	22.00
	r^2	0.93	0.70	0.83	0.83	0.80	0.52	0.84	0.73	0.87	0.76

B3 Estimates of Pollutant Loads,

Given the good results obtained using the stepped regression relationship, monthly loads were predicted using this relationship for the summer of 1997-98. Table B3 reports these results. These loads were calculated by determining the daily loads from the stepped regression relationship and summing over the month.

Table B3 Load Predictions, December 1997 to February 1998

Load	Predictions	Berow3# 2	Berow1# 15	Berow2# 4	Berow1# 11	Berow1# 4	Berow1# 2	Berow1# 9	Berow1# 6	Berow1# 8	Calna2# 1
Month	Load Type	Galston Gorge	Calna Creek (Confluence)	Crosslands	Cunio Pt	Coba Pt	Marramarra Confluence	Oaky Pt	Ants Nest Pt	Bujwa Bay	Calna Creek (Tidal Limit)
Dec 97											
Rainfall	TP (kg)	62	19	17	19	2.2	25	2.6	7	5.7	34
51 mm	TN (kg)	9861	8.7	45	82	5.3	79	8.4	16	34	7121
Jan 98											
Rainfall	TP (kg)	76	29	24	32	2.3	28	3.3	9	9.1	45
113 mm	TN (kg)	10390	11	56	104	7	98	10	23	41	7205
Feb 98											
Rainfall	TP (kg)	39	16	15	16	1.5	17	1.5	3.4	4.7	31
33 mm	TN (kg)	8251	4.8	23	42	2.9	30	4.7	7.9	19	6056