An Investigation into the Treatment Efficiency of a Primary Pond in the Barker Inlet Stormwater Wetland System, South Australia.

A Thesis submitted to the University of Adelaide
for the Degree of
Master of Engineering Science

by

Sarah E. Murphy

Submitted in fulfilment of the requirements of
the Degree of Master of Engineering Science

Department of Civil and Environmental Engineering

December 1999

Corrigenda

An Investigation into the Treatment Efficiency of a Primary Pond in the Barker Inlet Stormwater Wetland System, South Australia

The following amendments should be made to the text contained in this thesis in accordance with Examiners comments:

Page 3, second sentence:	replace with: "The resulting data set is one of the most comprehensive in existence in Australia and may play an importance role in further understanding water quality behaviour and interception processes."
Page 5, numbered item 1:	replace with "1. Deoxygenating substances, including organic material."
Page 12, Section 2.1.3.5:	replace final sentence with: "Anthropogenic sources of iron include rusting of steel (Gupta *et al.*, 1981), iron/steel industry emissions and landfill leachate (Canadian Council of Resource and Environment Ministers, 1987). The major (natural) source of iron in stormwater is soil."
Page 15, Section 2.1.5:	insert statement after the second sentence: "Photosynthesis and soil/rock properties are the major determinants of stormwater pH, with soil pH being influenced by vegetation."
Page 19, final paragraph:	replace "Litkowski (1997)" with "Litkowski *et al.* (1997)".
Page 47, Section 3.3.3:	add statement following first sentence: "It should be noted that macrophytes represent only a minor phosphorus sink (1–3%) compared to wetland sediments (90–95%)."
Page 48,Section 3.4:	following numbered items, add statement: "Many metals have a strong affinity for adsorbing to the surface of suspended sediment particles and, as a result, a majority of metals in stormwater are found in particulate form. The dominant removal pathway for total metal loads is therefore sedimentation."
Page 63:	add statement at bottom of page: "The catchment contributing to this Pond is the North Arm East Catchment with an area of 2200 hectares. The surface area of Pond 4 is therefore only approximately 0.5 percent of the contributing catchment area."
Page 158:	add comment after Figure 6.45: "By disregarding the large event on October 30 1997, 50 percent of the annual flow is still contributed by only 11 events out of the 54 recorded."
Page 163:	add statement before Figure 6.51: "The significant increases in ammonia encountered are indicative of high reducing conditions and are consistent with the small surface area of the pond compared to the size of the catchment (0.5% area ratio). These findings have important implications in the design of stormwater wetlands with respect to their size relative to the catchment area."
Pages 172–175	Replace all reference to "initial concentration" with "inflow concentration".:
Page 187:	replace Equation 7.6 with:
Page 190, first paragraph:	replace "calibrating" with "verifying".
	$L = C_{pd} V \left(1 - \frac{1}{\left(1 + \left(0.5 V_s A / Q\right)\right)^{0.5}} \right)$ where: Q = inflow (ML/day)
Page 205, first bullet point:	Replace reference to "initial concentration" with "inflow concentration".
Chapter 9 omitted references:	Chiew, F.H.S. (1995) "An Overview of Urban Water Research Studies in Australia." Report prepared for the Australian Science and Technology Council, 35pp. pH Environment (1995) "Patawalonga Catchment Sediment Quality." prepared for the Department of Environment and Natural Resources, Water Resources Group.

Abstract

The research undertaken for this thesis formed part of a multidisciplinary study initiated by the Government of South Australia (through the former MFP Development Corporation) at the Barker Inlet Wetlands in South Australia. For a period of 18 months the quality of stormwater flowing into and out of a primary pond in the system was monitored and analysed to gain an understanding of the pollutant removal efficiency of the system. Pond efficiency was examined both on an event basis and over the longer term. The length of the data set was such that annual loads and mean annual concentrations could be determined. Where data were missing due to instrumentation malfunctions, likely loads were estimated using a simple regression modelling technique which related event loads to event volume.

Results indicated that sediment and metal loads were significantly reduced by the pond both on an event basis, and over the longer term. Nutrient removal was variable on an event basis, although total nitrogen and total phosphorus loads over the long term (12-month period) were reduced substantially.

Removal efficiency demonstrated a dependence on initial concentration, a phenomenon noted by other researchers. Removal efficiency was also greater for smaller events, although for an extreme event recorded during the study period (50 year ARI) a significant removal was still achieved. This is contrary to the belief that high flows should be diverted or bypassed to prevent re-suspension of previously deposited sediments.

An attempt was made to estimate the longer term performance of the system using a simple model developed by the Cooperative Research Centre for Freshwater Ecology (CRCFE). Although good calibration was obtained for total phosphorus and total nitrogen, total suspended solids were not modelled as well. The model generally predicted a higher removal efficiency than was determined from monitoring.

The research has provided insight into the pollutant removal ability of a Primary Pond in the Barker Inlet Wetland System, which is one of the largest constructed wetlands in the World. Results have shown that storm size influences pollutant removal, however significant removal was achieved across a wide range of flow conditions.

Statement of Originality

This work contains no material which has been accepted for the award of any other degree or diploma in any University or other institution and, to the best of my knowledge, contains no material previously published or written by another person, except where due reference has been made in the text. I give consent to this copy of my thesis, when deposited in the University Library, being available for loan and photocopying.

Sarah E. Murphy

December 1999

Acknowledgements

Firstly I would like to thank my supervisors Dr David Walker and Mr Trevor Daniell for their invaluable assistance, support and patience throughout the duration of my degree. I also need to thank the former MFP Development Corporation for funding the data collection and McMahon Holdings for their Postgraduate Scholarship, without which this project would not have been possible.

Much gratitude is extended to the other members of the MFP team, in particular Rachel French for being such a great field and research partner and for never being too bust for chips & gravy or a game of pool. Also big thanks to Bryan Williams for all the hard work that went into the initial stages of the project.

A million thanks goes to Doug McCarty for his many hours of help in the field and for getting us out of many sticky (or is that muddy.....Rach?) situations. Also to Stan and Neil Woithe for their assistance with instrumentation and for fixing any bugs in the monitoring network at short notice, particularly on Friday afternoons!

Within the Department of Civil and Environmental Engineering, thanks to Greg and David in the labs and to Dianne, Bernie and the other general office staff for day to day support. Outside the Department, thanks to Paul Manning for directing the project, Civil Skills for help in the construction of the Henschke Street station, and Ben for the computer assistance and program writing.

Finally the biggest thanks of all to my family, friends, Ben and Syrah who gave me the love and support to see me through to the end. Especially thanks to Mum and Dad who sacrificed so much to give their children the best education imaginable.

Table of Contents

Index of Figures

Index of Photographs

Index of Tables

Index of Appendices

Glossary of Terms

Listed below are the meanings of the acronyms and abbreviations used in this thesis:

AEP	Annual exceedance probability
AHD	Australian height datum
ARI	Average recurrence interval
BOD	Biochemical oxygen demand
COD	Chemical oxygen demand
DO	Dissolved oxygen
EC	Electrical Conductivity
EMC	Event mean concentration
GPT	Gross pollutant trap
HENS.	Henschke Street
HLR	Hydraulic loading rate
HRT	Hydraulic residence (or retention) time
NAE	North Arm East
NEA	Northern Ephemeral Area
NH_3-N	Ammonia-N
NO_3^-N	Nitrate-N
NTU	Nephelometric turbidity units
OP	Orthophosphorus
PAH	Poly-cyclic aromatic hydrocarbon
SAR	Surface to area ratio
SRC	South Road Connector
TDS	Total dissolved solids
TKN	Total Kjeldahl nitrogen (ammonia-N + organic N)
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
TTU	Tain turbidity units

1

Introduction

Stormwater pollution is a byproduct of urbanisation. Not only has urbanisation led to an increase in the amount of runoff generated from catchments, but it has also resulted in a greater diversity of pollutants in stormwater than would normally be found in natural water.

Healthy, natural waterways are to some degree self-purifying. This is due to the array of physical, chemical, and biological processes that take place within them. Unfortunately during the process of urbanisation, a majority of natural waterways were replaced by engineered concrete drains. These were designed to remove stormwater as quickly as possible with little regard for downstream receiving waters.

The combination of larger stormwater flows and higher levels of pollution caused by urbanisation and the lack of treatment provided by engineered drainage systems has over time, resulted in the degradation of receiving waters. It has only been in the last decade or so that the problems associated with stormwater pollution have been acknowledged and remedial measures have been put in place.

The Barker Inlet is situated off the Gulf of St Vincent in South Australia and is adjacent Torrens Island. The aquatic reserve represents one of the World's southern-most stands of the Grey Mangrove *Avicennia marina* which, along with the native seagrasses, act as a spawning ground for many species of fish (MFP Australia, 1995). This spawning ground is vital to the recreational and commercial fishing industry in the Barker Inlet and the Gulf of St. Vincent waters.

Each year more than 4,500 ML of stormwater flows into North Arm Creek at the inland extremity of the Barker Inlet. This originates from a large band of catchments covering the Northern suburbs of Adelaide. Until recently stormwater was allowed to flow into the North Arm Creek untreated, and has resulted in damage to the fragile marine habitat in the area. It is of vital importance that this unique ecosystem now be protected, which can be achieved partly through the management of stormwater from the Barker Inlet catchment.

In 1994 the State Government of South Australia, through the former MFP Australia, commissioned the development of the Barker Inlet Wetlands at Dry Creek. The Wetlands are located around 20 kilometres to the North of the Adelaide metropolitan area and were constructed on degraded low-lying coastal land adjacent the Barker Inlet North Arm.

The system covers an area of approximately 172 hectares and forms an integral link in the chain of wetlands stretching from Greenfields Wetlands at Mawson Lakes to the Magazine Creek Wetlands in Gillman. Together, these wetlands form the largest group of constructed wetlands in the world and treat approximately 40 percent of Adelaide's stormwater runoff. The Barker Inlet Wetlands alone intercept and treat stormwater runoff from around 26 percent of the Adelaide metropolitan catchment area (MFP DC, 1996)

While the popularity of wetlands for stormwater quality control has escalated in recent years, there is still much unknown about their efficiency, or indeed the reasons why they are effective. Any research being conducted on them is therefore providing invaluable information which can be used to further refine design and management principles.

The former MFP Development Corporation initiated a comprehensive study of the Barker Inlet Wetland system in 1994, which brought together the expertise of researchers from a number of institutions and organisations including:

- The University of Adelaide
    - Department of Civil and Environmental Engineering
    - Department of Botany
- CSIRO
    - Land and Water
- The University of South Australia
    - Urban Water Resources Centre
    - Aquatic Toxicology Unit, School of Pharmacy and Medical Sciences
- Ecomanagement Services

A number of concurrent studies have been carried out on the site over the last five years, of which the study presented in this thesis is just one. Other research has focussed on groundwater, salinity, terrestrial vegetation, aquatic vegetation, sediment chemistry and ecotoxicology.

The Department of Civil and Environmental Engineering's involvement in the study began in 1994 with the establishment of water quality monitoring stations in each of the major inflow drains (North Arm East Drain, HEP Drain, Dunstan Road Drain, and South Road North Arm West Drain) to the Wetland system. The number of monitoring stations maintained grew to 7 in 1997, with the installation of the Henschke Street inflow monitoring station and the South Road

Connector and Bund outflow monitoring stations. This enabled the quality of stormwater at every inflow location to be monitored (both continuously and during events) as well as the outflow from the largest of the Southern ponds and the outflow from the entire system at the Bund station.

The resulting data set is one of the most comprehensive in existence in Australia.

The study presented in this thesis has focussed on the largest of the Southern ponds in the system, referred to as Pond 4. Monitoring stations at the inlet and outlet of the pond enabled a continuous assessment of performance to be made over the study period.

The objectives of the study were to determine loads entering and exiting the Pond for a period sufficient to determine its performance. In addition, it was hoped that inflow and outflow loads could be characterised according to flow conditions, to provide a model suitable for further predictions of performance, without the need for monitoring. The information gathered during this research period and the conclusions drawn from it will provide invaluable insight into the enhanced design and management of urban stormwater treatment ponds.

The first two chapters of this thesis provide a background to water quality and previous studies, and the underlying dynamics of wetland systems. Following these are descriptions of the study site and monitoring network used to collect data.

Chapter 6 presents the details of the results obtained from the monitoring program and providies estimates of performance. Performance of the Pond has been discussed both on an event basis and over a 12-month period. The data has also been used to develop simple regression equations to predict inflow and outflow quality based on event volume.

The penultimate chapter, Chapter 7, presents results of the water quality computer modelling which was conducted using a model developed by the CRC for Freshwater Ecology.

Finally, a summary and the conclusions of the study are presented in Chapter 8.

2

Literature Review

In the past stormwater was viewed as a waste product of society and conveyed away to downstream rivers and oceans as quickly as possible. There is now a growing acceptance that stormwater has the potential to be a valuable resource if managed correctly as well as the realisation that untreated stormwater can have a detrimental effect on downstream receiving waters.

Stormwater control facilities such as wetlands and retention/detention basins are becoming common features of the urban landscape due to the recognition that they can significantly improve the quality of urban stormwater as well as retarding storm flows. The design of these facilities was in the past based purely on hydraulic performance. Now design is focusing more and more on the incorporation of features to enhance pollutant removal, with much research on optimum design techniques still being undertaken.

Livingston (1991) stated that monitoring of stormwater wetland systems is essential to determine relations between design variables and pollutant removal efficiency. Unfortunately, the data base on wetland performance in Australia at present is seldom sufficient for rigorous design protocols (Raisin *et al.*, 1997). While hydrological data has been collected continuously for over 100 years at some sites, it has only been realised relatively recently that water quality data are also important (Cordery *et al.*, 1997).

Computer modelling is a popular technique used to overcome deficiencies in hydrologic and water quality data, and indeed to eliminate the need for extended monitoring. All current hydrological and water quality modelling involves many assumptions and as a result the models must be calibrated for the particular situation to which they are to be applied if they are to be credible. Experience has shown that models developed in the absence of local data for verification can provide poor estimates of reality (Cordery *et al.*, 1997).

The following chapter discusses the important aspects of stormwater quality, its monitoring and modelling. The design of stormwater treatment facilities is also addressed, along with a review of how these facilities are functioning in other parts of Australia and the world.

2.1 Water Quality – The Important Parameters

Healthy natural waterways are self-purifying through complex interactions involving physical, chemical and biological processes (Tomlinson *et al.*, 1993). When natural waterways are replaced by urban drainage schemes these processes become ineffective, allowing changes in the biology and chemistry of the water to take place. Combined with the effect of increased runoff by the creation of large impervious areas, and increased water pollution by the use of environmentally degrading substances, disposal of untreated urban stormwater runoff can have significant effects on receiving waters.

In aquatic areas, animals and plants are adapted to the specific environmental conditions in which they exist and to each other (Connell, 1993). Although there may be diurnal or even seasonal variations, these occur according to some pattern of natural relationships. It therefore follows, that when a system is subject to change due to pollution, a resulting change in the array of aquatic biota, or plants and animals, would also be expected (Hynes, 1974).

The biological changes that occur are related to the characteristics of each pollutant. As a broad generalisation, pollutants have been placed in the following categories (Connell, 1993):

1. Deoxygenating substances;
2. Toxic substances (e.g. Metals);
3. Plant nutrients and fertilisers;
4. Suspended solids;
5. Energy pollutants such as heat and radioactivity; and
6. Biota (such as micro-organisms) which can cause disease.

With the growing realisation of the environmental problems arising from urban development comes the need to change the way in which we dispose of stormwater in order to protect, or even restore, our fragile receiving waters.

Each year more than 4500 ML of stormwater flows into the Barker Inlet which was, until recently, untreated and laden with pollutants potentially toxic to the natural ecosystem. The Barker Inlet represents one of the worlds southern-most stands of the Grey Mangrove *Avicennia marina* which, along with the native seagrasses, act as a spawning ground for many species of fish (MFP Australia, 1995). This spawning ground is vital to the recreational and commercial fishing industry in the Barker Inlet and the Gulf of St. Vincent waters. It is of vital importance that this unique ecosystem be protected, which can be achieved partly through the management of stormwater from the Barker Inlet catchment.

The pollutants of concern in this area have been outlined below along with their associated risks, common concentrations and sources. These have been identified as the most appropriate parameters to monitor at the Barker Inlet Wetland site.

2.1.1 Total Suspended Solids and Turbidity

The total suspended solids (TSS) concentration of water consists of the total amount of organic and inorganic particulate matter suspended in the water column. Not only do high concentrations of suspended solids compromise the aesthetic appeal of water, they can also create a threat to public health as pathogens [illegible] and other toxic substances, for example heavy metals, are commonly associated with the sediment particles (Smith *et al.*, 1994). In addition to this, suspended solids inhibit the respiration and feeding of biota, reduce light transmission necessary for plant photosynthesis, promote infections (US EPA, 1986), and when sediment is deposited it can suffocate benthic organisms.

Makepeace *et al.* (1995), in a review of literature published throughout the world on stormwater contaminants, reported that total suspended solids in stormwater have been found in the concentration range of 1 to 36 200 mg/L with mean values of 4 to 1223 mg/L. Williams (1997) reported a mean suspended solids concentration in stormwater entering the Magazine Creek Wetland in South Australia of 1535 mg/L which is slightly higher than the value published by Makepeace *et al.* (1995).

TSS

Because the quantity of sediment entering any given stream or pond depends greatly on natural factors, it is difficult to establish criteria for suspended sediment concentration. Smith *et al.* (1994) point out that in many western areas, stream ecosystems are naturally adapted to suspended sediment concentrations that are periodically many times greater than those detrimental in other areas. For this reason both ANZECC (1992) and the US EPA make recommendations in terms of light penetration, or turbidity, and recommend a maximum of 10 percent variation in seasonal mean turbidity (in Nephelometric Turbidity Units). As mentioned previously, high turbidity of water reduces the light penetration and hence primary production. The impacts are most pronounced on waters with very low suspended solids concentrations (and turbidity) where increases of as little as 5 mg/L can measurably reduce photosynthesis (Ryan, 1991).

The main source of suspended sediment is soil erosion. Although organic particles also form suspended sediment, the majority is inorganic by weight. Rates of erosion vary greatly and are site specific. It is dependent on factors such as soil characteristics, precipitation frequency and intensity, slope of land, and catchment activity such as construction or agriculture (Smith *et al.*, 1994).

2.1.2 Nutrients

Nutrients are the elements essential to the survival of all life forms. Of the many known nutrients, six of these, known as the macro nutrients, comprise 95 percent of the mass of all living organisms. These nutrients are carbon, oxygen, hydrogen, phosphorus, nitrogen, and sulphur (Miller & Armstrong, 1982). C O H P N.

S

P N

Literature regarding water quality generally focuses on phosphorus and nitrogen as the nutrients of concern. This is simply because nitrogen and phosphorus normally have the lowest levels of all the nutrients required by living organisms and are often termed the limiting nutrients of a system.

Plant growth (via photosynthesis) is primarily dependent on sunlight and inorganic nutrients, and can be summarised by the following (simplified) equation (ANZECC, 1992)

$$\textbf{Sunlight} + 106CO_2 + 16NO_3^- + HPO_4^{2-} + 122H_2O + 18H^+ + \textbf{(trace elements)} \rightarrow \{(CH_2O)_{106}(NH_3)_{16}(H_3PO_4)\}\ \textbf{(plant biomass)}$$

光合作用

The input of light or the availability of nitrogen and/or phosphorus usually limits biomass production as seens by the above equation. The most bioavailable form of phosphorus is orthophosphate (PO_4^{3-}) and the most bioavailable forms of nitrogen are ammonia (NH_3) and nitrate (NO_3^-).

2.1.2.1 Nitrogen

富营养化的

Nitrogen compounds are one of the main constituents of concern in stormwater because of their role in eutrophication, their effect on the oxygen content of receiving waters and their toxicity to some aquatic life forms (Kadlec & Knight, 1996). The most important inorganic forms of nitrogen are ammonia (NH_3), nitrite (NO_2^-), nitrate (NO_3^-), nitrous NO oxide, and dissolved elemental nitrogen. Nitrogen can also be present in wetlands in many organic forms including urea, amino acids, amines, purines and pyrimidines. Table 2.1 from Makepeace *et al.* (1995) lists the forms of nitrogen commonly studied in stormwater and their respective concentration ranges.

Ammonia normally exists in water in the ionised form, NH_4^+. It is important because:

1. it is the preferred nitrogen form for most wetland plants (see Section 3.2);
2. ammonia is chemically reduced and can be readily oxidised resulting in the consumption of oxygen; and
3. un-ionised ammonia is toxic to many life forms at very low concentrations.

显著的,占优势的

It should be noted that ammonia is more predominant in wastewater than stormwater.

Table 2.1 Nitrogen forms and concentrations in stormwater

Form of Nitrogen	**Concentration Range (mg/L) [1]**	**Concentration Range (mg/L) as N [2]**
Total Nitrogen	0.32 - 16.0	0.61 - 64.11
Inorganic Nitrogen	0.09 - 5.44	NA
Organic Nitrogen	0.32 - 16.0	0.34 - 56.0
Nitrate	0.01 - 12.0	Nitrate + Nitrite
Nitrite	0.02 - 1.49	0.01 - 0.7
Ammonia	0.01 - 4.3	0.01 - 5.9
Total Kjeldahl Nitrogen	0.32 - 16.0	0.44 - 64.1

(1) Makepeace et al. (1995)
(2) Williams (1997)
NA – Data not available from this study

Nitrite is an intermediate oxidation state of nitrogen and is therefore chemically unstable and found in very low concentrations in water. Nitrate on the other hand, is chemically stable and would persist unchanged if not for biological interaction (see Section 3.2). Nitrate is an essential nutrient for plant growth and can lead to eutrophication (Kadlec & Knight, 1996).

Organic nitrogen, as already mentioned, is made up of a variety of compounds including amino acids, urea, uric acid and purines and pyrimidines. Common expressions of nitrogen content in stormwater include *total Kjeldahl nitrogen (TKN)*, which is the sum of ammonia and organic-N, *total inorganic nitrogen (TIN)*, which is the sum of ammonia and nitrate (+nitrite)-N, and *total nitrogen (TN)* which is the sum of the organic and inorganic fractions.

Sources of nitrogen in stormwater include fertilisers, industrial cleaning operations, feed lots, animal excrement and the combustion of fossil fuels (Makepeace *et al.*, 1995).

2.1.2.2 Phosphorus

Phosphorus is a nutrient required for plant growth and is frequently a limiting factor for vegetative productivity. This is most easily understood by considering the Redfield ratio which is the mass proportion of nutrient elements in ecosystem biomass.

$$C:N:P = 40:7:1$$

Stormwater rarely has this ratio and therefore can result in a nutrient imbalance when introduced to receiving waters. As the ratio indicates, the introduction of trace amounts of phosphorus can have significant effects on the structure of ecosystems.

In total phosphorus, the majority (>90%) is organic and of this, more than 70 percent is particulate organic matter and the rest is dissolved or colloidal (Wetzel, 1983). Water generally consists of dissolved and particulate phosphorus in the forms listed below.

Particulate phosphorus

- phosphorus in organisms;
- phosphorus adsorbed to inorganic complexes such as clays and carbonates; and
- phosphorus adsorbed to dead particulate organic matter.

Dissolved Phosphorus

- orthophosphates (soluble reactive P);
- polyphosphates (detergents); and
- organic colloids.

Table 2.2 summarises the concentration ranges for forms of phosphorus found in stormwater.

Table 2.2 Phosphorus forms and concentrations in stormwater

Form of Phosphorus	Concentration Range (mg/L) (1)	Concentration Range (mg/L) as N (2)
Total Phosphorus	0.01 - 7.3	0.12 - 14.2
Dissolved Phosphorus	0.038 - 3.52	0.01 - 3.93
Particulate Phosphorus	0.014 - 2.85	0.01 - 13.29

(1) Makepeace et al. (1995)
(2) Williams (1997)

The most important form for plant nutrition is ionised inorganic orthophosphate (PO_4^{3-}) which is usually around five percent of the total phosphorus in natural waters.

Natural sources of phosphorus, such as erosion of rocks, is not the primary provider of phosphorus in urban environments. Sources in stormwater include tree leaves (Dorney, 1986), fertilisers, industrial waste, detergents and lubricants (Canadian Council of Resource and Environment Ministers, 1987).

2.1.3 Metals

Although the ions of some metals such as iron, calcium, zinc and copper are essential for life, some other metal ions are poisonous and can accumulate in certain body tissues (Laidler, 1991). These are the so called "heavy metals" including; lead, mercury and cadmium. They are found towards the bottom of the periodic table and have a high atomic weight. Many shellfish accumulate heavy metals in their bodies and if eaten regularly poisoning can result. Oysters, for example, can accumulate 100 000 times more mercury in their body than is present in the environment.

The Nationwide Urban Runoff Program (NURP – USEPA, 1983) in the United States concluded that metals, in particular copper lead and zinc, were by far the most prevalent priority pollutant constituents in urban runoff (Yousef *et al.*, 1986). Copper appeared to be the most toxic to aquatic life in some areas of the country.

Metal concentrations reported in literature can be misleading, as they tend not to reflect the metals 生物有效性 bioavailability. Free metal ions and weak inorganic complexes are usually in the most bioavailable form and hence are the most toxic to aquatic life (Makepeace *et al.*, 1995). As metals tend to bind to small particles in water, waters high in suspended particulate matter are often also high in metal concentrations. Generally the proportion of metals in the bioavailable form in stormwater is quite small and will change according to other parameters such as pH (Makepeace *et al.*, 1995). In addition to this, certain mixtures of heavy metals can have a toxicity greater than the added toxicities of individual species (synergism) and other combinations a reduced toxicity (antagonism) (ANZECC, 1992). Unfortunately water quality guidelines do not consider these effects and should be adjusted according to each specific ecosystem to which they are applied.

Some of the more common metals found in stormwater are discussed briefly below.

2.1.3.1 Aluminium

The bioavailability of aluminium is generally greatest in acidic waters with maximum toxicity found to be at around pH 5-5.2 (Schofield & Trojnar, 1980). The inorganic single unit aluminium species ($AlOH^+$) is thought to be the most toxic (Driscoll *et al.*, 1980).

A review carried out by Makepeace *et al.* (1995) found aluminium concentrations in the range 0.1 to 16 mg/L. It is interesting to note that a study by Williams (1997) on the Eastern Parade drain in the Magazine Creek Wetland Catchment in South Australia recorded a maximum aluminium concentration of 41.3 mg/L.

Aluminium in stormwater runoff may be due to natural sources (Dannecker *et al.*, 1990) or anthropogenic sources such alum used in water treatment, emissions from coal combustion, and various other industrial processes.

2.1.3.2 Cadmium

In surface waters cadmium occurs predominantly in the divalent form, comprising several organic and inorganic compounds (Reeder *et al.*, 1979). Although extremely toxic in the bioavailable form, cadmium is less toxic in natural waters than pure distilled water due to its affinity for silt particles (Golterman, 1975). The acute toxicity of cadmium is affected by hardness, pH, and water temperature (Makepeace *et al.*, 1995).

Makepeace *et al.* (1995) report a range of cadmium concentrations in stormwater of 0.00005 to 13.73 mg/L with the mean range being 0.003 to 0.011 mg/L. The highest mean found by Williams (1997) was 0.01 mg/L in the Magazine Catchment, South Australia.

Sources of cadmium in stormwater include combustion, wear of brake pads and tyres, corrosion of galvanised metal, and emissions from metal finishing plants (Makepeace *et al.*, 1995)

2.1.3.3 Chromium

Chromium in stormwater is mainly found in the trivalent Cr^{3+} and hexavalent Cr^{6+} forms and is often associated with suspended solids. Cr^{6+} is soluble, mobile and can be very stable in waters low in organic matter. Cr^{3+} has an affinity to form stable complexes such as chromium hydroxide, this is thought to be the major removal path of trivalent chromium from water (Makepeace *et al.*, 1995; ANZECC, 1992).

The Cr^{3+}:Cr^{6+} ratio in natural waters has been found to be dependent on the amount of organic matter and the dissolved oxygen concentration (Benes & Steinnes, 1975). Chromium is bioaccumulated by aquatic organisms with chromium (VI) being considered the more toxic form due to its ability to penetrate cell membranes to a much greater extent than chromium (III).

Makepeace *et al.* (1995) report chromium concentrations in stormwater between 0.001 and 2.3 mg/L with the means ranging from 0.01 to 0.23 mg/L.

Sources of chromium include corrosion of welded metal plating (Ward, 1990), wear of bearings and bushes (Gupta *et al.*, 1981), dyes, paints, ceramics, paper, fire sprinkler systems, pesticides and fertilisers (Canadian Council of Resource and Environment Ministers, 1987).

2.1.3.4 Copper

Copper is commonly found in the Cu^{2+} form in natural waters but speciation depends on pH and the presence of organic and inorganic ligands in the water (ANZECC, 1992). The toxicity in water increases with decreasing water hardness and dissolved oxygen concentration. High suspended solids concentrations lead to lower copper toxicities, presumably a result of complexation producing less bioavailable forms of copper (Spear & Pearce, 1979). Copper is considered the major aquatic toxic metal in stormwater (Makepeace *et al.*, 1995). As an essential element copper is readily accumulated by plants and animals.

Makepeace *et al.* (1995) report copper concentrations in stormwater between 0.00006 and 1.41 mg/L with means ranging from 0.0065 to 0.15 mg/L. The toxicity of copper on aquatic life is between 0.017 and 10.24 mg/L at a hardness of 50 mg/L. Williams (1997) found a mean copper concentration in the Magazine Creek Catchment of 0.48 mg/L with a maximum value of 1.23 mg/L.

Sources of copper in the urban environment include wear of tyres and brake linings (Ward, 1990) and metallurgical and other industrial emissions (Dannecker *et al.*, 1990).

2.1.3.5 Iron

The most common oxidation states of iron are the ferrous (Fe^{2+}) and ferric (Fe^{3+}) states. In surface waters, ferric iron is the predominant species although the ferrous form can persist in anaerobic reducing waters (ANZECC, 1992). Iron is usually associated with suspended solids and often acts as a site for the adsorption of phosphorus.

Iron is an essential trace element for plants and for animals as part of the haemoglobin in red blood cells (Laidler, 1991). Acute toxicity to insects however, has been reported at iron concentrations between 0.32 and 16 mg/L (Warnick & Bell, 1969). Makepeace *et al.* (1995) reported iron concentrations in the range 0.08 to 440 mg/L with means ranging from 0.998 to 12 mg/L. Williams (1997) found a maximum concentration of 102 mg/L with a mean of 30.1 mg/L.

Sources of Iron include rusting of steel (Gupta et al., 1981), iron/steel industry emissions and landfill leachate (Canadian Council of Resource and Environment Ministers, 1987).

2.1.3.6 Lead

Lead has been identified as the most important contaminant of concern in stormwater research. In stormwater lead is mainly associated with suspended solids, it adsorbs to particles and forms carbonate precipitates. Changes in pH affect the speciation (Makepeace *et al.*, 1995). In fresh water the main species is $PbCO_3$ and lead-organic complexes, with much smaller concentrations of free lead ions (ANZECC, 1992).

In humans, lead is a cumulative poison which builds up in bones and mostly affects children (Laidler, 1991). Organic lead compounds also interfere with the functioning of the central nervous system. Australian freshwater animals experience acute toxicity at concentrations between 0.18 and 0.5 mg/L, although this is influenced by water hardness (Bacher & O'Brien, 1990). Rainbow trout show spinal deformities in soft water at 0.031 mg/L but in hard water can tolerate concentrations in excess of 0.19 mg/L (Biesinger & Christen, 1972). Makepeace *et al.* (1995) reported lead concentrations in the range 0.00057 to 26 mg/L with means from different reports ranging from 0.0209 to 1.558 mg/L.

The main source of lead is emissions from petrol-powered motor vehicles and petrol additives such as tetrahedral lead. Although the increasing use of un-leaded petrol and LPG in vehicles is reducing these emissions, the ever increasing use of the automobile is counteracting the decrease at present. Lead enters the stormwater system through fallout and washout of lead in the air that contaminates roadside soils, vegetation and ultimately water.

2.1.3.7 Mercury

In the aquatic environment mercury exists mainly as complexes of mercury (II) and as organomercurials (Hart, 1982). The major concern in aquatic environments is that the inorganic forms of mercury which have *relatively* low toxicity and availability may be converted under anaerobic conditions into organomercury complexes such as methylmercury which are highly toxic and bioaccumlate (ANZECC, 1992; Laidler, 1991). Equation 2.1 is the bacteriological conversion that takes place and involves anaerobic methane bacteria.

$$Hg^{2+}(aq) \xrightarrow{\substack{\textit{anaerobic}\\ \textit{bacteria}}} Hg(CH_3)^{+}(aq)$$

Equation 2.1

toxic
short residence time

very highly toxic
long residence time

The most sensitive aquatic plant species are generally less sensitive than the sensitive animal species to both mercury (II) and methylmercury (CCREM, 1991). Fish tend to be more resistant than molluscs and crustaceans (ANZECC, 1992). Any guidelines are generally aimed at limiting bioaccumulation in seafood that may be eaten by humans (ANZECC, 1992). Mercury poisoning in humans affects the central nervous system causing slurred speech, loss of coordination and general mental degeneration (Laidler, 1991).

Makepeace *et al.* (1995) report mercury concentrations in stormwater between 0.00005 and 0.067 mg/L. Acute toxicity ranges from 0.0022 to 2.0 mg/L for Hg^{2+}. For methylmercury acute toxicity for rainbow trout is 0.024 mg/L, chronic toxicity ranges from 0.00004 (*D. magna*) to 0.00052 mg/L (brook trout) (Canadian Council of Resource and Environment Ministers, 1987)

Sources of mercury include emissions from the chlor-alkali industry, coal combustion, paint industry, and dental amalgam (Makepeace *et al.*, 1995).

2.1.3.8 Nickel

More than 90 percent of the nickel in the aquatic environment is associated with particulate matter or sediments, and only 7 percent is in the dissolved phase (Hart, 1982). Nickel in solution generally exists as a divalent cation, Ni^{2+}.

US Environmental Protection Agency (1986) compiled data on the acute toxicity of nickel to 21 freshwater species. The values ranged from 1.1 to 43 mg/L. Data available indicates chronic toxicity decreases as water hardness increases (ANZECC, 1992). Makepeace *et al.* (1995) found nickel concentrations in stormwater ranging from 0.001 to 49.0 mg/L, with means in the range 0.006 to 0.15 mg/L.

Sources of nickel include corrosion of welded metal plating (Ward, 1990), electroplating and alloy manufacturing and food production (Canadian Council of Resource and Environment Ministers, 1987).

2.1.3.9 Zinc

In stormwater zinc is mainly associated with dissolved solids (Morrison *et al.*, 1984) although it will also adsorb to suspended solids and colloidal particles (Makepeace *et al.*, 1995). The toxicity of zinc is influenced by water hardness and pH. Generally the acute toxicity of zinc is lower in waters with higher water hardness and lower pH (Mount, 1966; Holcombe & Andrew, 1978).

Makepeace *et al.* (1995) reported zinc concentrations in stormwater between 0.0007 and 22 mg/L with means between 0.0166 and 0.58 mg/L. Williams (1997) found a maximum concentration of 54 mg/L in the industrial Magazine Creek Catchment in South Australia, means were 0.51 and 0.6 in two residential catchments and 0.79 and 14.3 mg/L (median of 2.8 mg/L) for two industrial catchments.

Sources of zinc include wear from tyres (filler material), brake pads (Gupta *et al.*, 1981; Ward, 1990) and corrosion of metal objects and building materials (e.g. galvanised steel).

2.1.4 **Dissolved Oxygen** (COD· BOD).

Dissolved oxygen (DO) is essential to the respiration 呼吸 of aquatic organisms with its concentration being a major determinand of the species composition 组成 of biota 生物群 in the water and underlying sediments (Smith *et al.*, 1994). The dissolved oxygen concentration of water also plays an important role in determining the biochemical reactions that take place, which in turn affect many other aspects of water quality, including the solubility of toxic elements (particularly metals) and the aesthetic qualities of odour and taste.

Dissolved oxygen concentrations in stormwater can range from 0 to 14 mg/L (Makepeace *et al.* (1995). In the absence of substances that cause its depletion, ① the DO concentration in stream water approximates the saturation level for oxygen in water in contact with the atmosphere and decreases with increasing temperature from about 14 mg/L at freezing to about 7 mg/L at 30 degrees Celsius (Smith *et al.*, 1994). ② DO concentrations in stormwater tend to fluctuate significantly on a daily basis during periods of base flow and during storm events tend to drop 1 to 1.5 mg/L below the normal base flow concentrations (Keefer *et al.*, 1979). According to Keefer *et al.* (1979) there is a good correlation 相关性 between the DO concentration and flow for stormwater.

The main sources for DO depletion are the decomposition 分解 of organic material and the oxidation 氧化 of some inorganic compounds 混合物 (Canadian Council of Water Resource and Environment

Ministers, 1987). Although there is a lack of appropriate guidelines in Australia for stormwater quality, freshwater aquatic guidelines in Canada range from 5 mg/L for warm water biota to 9.5 mg/L for cold water biota in early stages of life. While extensive data exist on the effects of depleted dissolved oxygen levels on fish elsewhere in the world, similar Australian data are scarce. Koehn and O'Connor (1990) have reviewed data on freshwater fish in Victoria, and suggest that levels below 5 mg/L are stressful to many species. According to ANZECC (1992) there are no published data on the effects of dissolved oxygen on Australian aquatic invertebrates.

2.1.5 pH

Most natural fresh waters have a pH close to 7.0 and marine waters close to 8.2 (ANZECC, 1992). In many waters the pH is controlled by the carbonate-bicarbonate buffer system with marine waters being very strongly buffered. Even the slightest change in pH in marine waters indicates a major change to the system (ANZECC, 1992). Generally a pH in the range of 5-9 is not lethal to fish, although the toxicity of several pollutants, for example ammonia and cyanide, is strongly influenced by pH changes.

2.1.6 Bacteriological Contaminants

The concentration of faecal coliform bacteria is generally considered an important indicator of water quality because the presence of these organisms is a reliable indicator of faecal contamination from warm blooded animals (US Environmental Protection Agency, 1976). The correlation between infectious disease and specific species of faecal coliform bacteria such as *Escherichia coli* (*E. coli*) is well established, but total concentrations of faecal coliforms are easier to measure than individual species and, thus have for many years been used as an indicator. The total coliform analysis detects the presence *of E. coli, Citrobacter, Enterobacter, and Klebsiella* (Makepeace *et al.*, 1995). In stormwater however, quite high concentrations of bacteria can be experienced along with other microorganisms of pathogenic, non-human and non-enteric origin. Faecal indicators will therefore not always adequately assess the waters potential health risk (O'Shea & Field, 1991).

Makepeace *et al.* (1995) report total coliform concentrations in stormwater between 7 and 1.8×10^7 colony forming units (CFU)/100 ml. Faecal coliform concentrations have been detected in the range 0.2 to 1.9×10^6 CFU/100 ml.

2.1.7 Petroleum Products (Hydro carbons).

At times stormwater runoff from urban catchments can have a characteristic multicolored sheen on the surface (Williams, 1997). The source of this oily film is predominantly engine oil which has leaked and been washed off the road into the stormwater system. Despite the appearance, the amount of oil in stormwater is usually small with little threat to aquatic life. The more toxic components of this film, namely the Polycyclic Aromatic Hydrocarbons (PAHs), can however exist in stormwater in significant concentrations and are highly toxic to aquatic organisms. The high molecular weight PAHs have a very low water solubility and are easily adsorbed onto suspended and bed sediment and aquatic biota (Makepeace *et al.*, 1995).

Many of the PAHs were detected in the NURP Priority Pollutant Study, and, in a comparison between catchments of different land use, Williams (1997) found PAHs to be significant in a catchment with a high proportion of industrial land use and only during base flows. The main sources of Polycyclic Aromatic Hydrocarbons are petroleum, lubricating oils, bitumen, car exhaust fumes and decomposing organic matter emissions (Snelder & Trueman, 1995).

Table 2.3 lists the PAHs detectable in stormwater and the range of concentrations from the literature search of Makepeace *et al.* (1995) and from the data collection of Williams (1997).

Table 2.3 Polycyclic aromatic hydrocarbons in stormwater

Polycyclic Aromatic Hydrocarbons	Range of values or mean and standard deviation (μg/L) (1)	Range of concentrations experienced (μg/L) (2) → 经验值
Anthracene	9×10^{-3} – 10	<0.5 – 5.4
Benzo(a)anthracene	0.3×10^{-3} – 10	<0.5 – 0.6
Benzo(*b*)fluoranthene	3.4×10^{-3} – 1.9	< 0.5
Benzo(*k*)fluoranthene	1.2×10^{-3} – 10	< 0.5
Benzo(*g,h,i*)perylene	2.4×10^{-3} – 1.5	< 0.5
Benzo(a)pyrene	2.5×10^{-3} – 10	< 0.5 – 3
Benzo(e)pyrene	0.4 – 0.609	NT
Chrysene	3.8×10^{-3} – 10	< 0.5
Dibenzo(a,*h*)anthracene	0.6×10^{-3} – 0. 9	< 0.5
Fluoranthene	30×10^{-3} – 56	< 0.5 – 5.1
Fluorene	96×10^{-3} – 1	< 0.5 – 17
Indeno(*1,2,3-c,d*)pyrene	0.31 – 0. 5	< 0.5
Methylphenanthrenes	2.9 ± 3.4	NT
2-Methylanthracene	10×10^{-3} – 1.6	NT
9,10-Dimethylanthracene	1 ± 1.4	NT
Napthalene	36×10^{-3} – 2.3	< 0.5 – 30
Perylene	50×10^{-3} ± 0.5	NT
Phenanthrene	45×10^{-3} – 10	< 0.5 – 16
Pyrene	45×10^{-3} – 10	< 0.5 – 11
Total PAH	0.24 – 1.3	NT

(1) values reproduced from Makepeace et al. (1995)
(2) data taken from Williams (1997)
NT – parameter not tested in this study

传导率

2.1.8 Dissolved Solids and Conductivity

There is some confusion in the reporting of total dissolved substances in water as many different terms, such as total dissolved salts, total dissolved solids (TDS), salinity, conductivity, and filterable residue are all used, often interchangeably (ANZECC, 1992). Dissolved solids, measured in milligrams per litre, or parts per million (ppm), refers to the sum of all dissolved constituents in a water sample (Smith *et al.*, 1994). In most cases the major components of TDS are the ions or salts of calcium, magnesium, sodium, potassium, bicarbonate, sulphate, and chloride. Electrical conductivity (EC) is a term commonly used to characterise the salinity of water and is an approximate measure of the concentration of dissolved ions. The units of conductivity are micro siemens per centimeter (µS/cm). The relationship between EC and TDS is usually linear, or in other words

$$\text{TDS} = k \times \text{EC}$$

The value of k varies between 0.55 and 0.75 depending on the location, ANZECC (1992) assume a k value of 0.68. Analysis at the Barker Inlet site has showed an average correlation between EC and TDS (k-value) of 0.552. Figure 2.1 shows the relationship for the site.

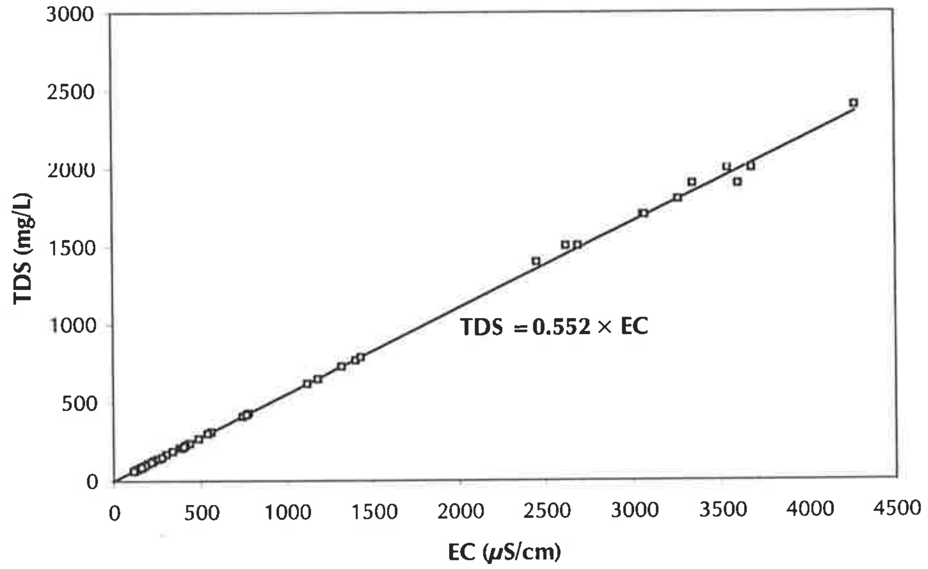


Figure 2.1 TDS vs. EC relationship for Barker Inlet Wetland and catchment

The main source of dissolved solids in stormwater is dissolution of minerals naturally found in soil and rocks (Smith *et al.*,1994), the TDS concentration of rainwater is typically very low, in the order of just a few milligrams per litre (Berner & Berner, 1987). Makepeace *et al.* (1995) in their review of stormwater contaminant data throughout the world reported TDS concentrations in stormwater between 75.9 to 2792 mg/L. The mean in this study was 178 mg/L compared to 100 mg/L found by Berner and Berner (1987).

The major significance of these ions in stormwater is the restrictions it imposes on the potential re-use for irrigation or industrial purposes rather than ecological significance. Some problems have been found at the Barker Inlet Wetland site with colonisation of aquatic plants due to the very saline water in some ponds.

2.1.9 Temperature

Water temperature has a substantial effect on the functioning of aquatic ecosystems and the physiology of biota. Physiological processes have optimum temperatures and any alteration to the ambient temperature may affect certain species in a variety of ways (ANZECC, 1992). Changes in ambient temperature regimes may affect growth and metabolism, timing and success of reproduction, migration patterns and mobility. The effects may be direct, such as metabolic changes, or indirect, such as changes in the solubility of oxygen in water.

There are two activities that may cause substantial changes in water temperature; discharge of cooling waters or heated effluent and discharge of cold water from reservoirs. The former has received some attention in Australia, but the latter has largely been ignored. In Australia there is relatively little information on the thermal tolerances of Australian aquatic organisms or their response to temperature changes.

2.2 Stormwater Quality Data

Traditionally, research in the field of stormwater has concentrated on the quantity of water produced and the development of methods to safely handle the runoff. Only in the last decade or so have the contaminants in the stormwater become recognised as a cause for concern.

In the early 1980s the US Environmental Protection Agency funded the Nationwide Urban Runoff Program (NURP) as a means of obtaining a comprehensive data set to evaluate and control the effects of polluted urban stormwater runoff. During the study a large number of structural and non-structural pollution abatement facilities were investigated at 29 sites across the United States to determine their effectiveness at removing potentially harmful substances from stormwater. This program provided a substantial advance in the base of data pertinent to support decisions and effectively engineer systems for stormwater runoff control (Driscoll, 1986).

A number of authors have published literature combining data collected from urban stormwater studies published throughout the world. These studies have provided insight into not only "typical" concentrations, but also the variation between studies. Makepeace *et al.* (1995) presented a comprehensive literature review of all international literature published in the last 25 years to identify and quantify contaminant data available on stormwater. In order to assess impacts and put into perspective the importance of the concentrations reported in the review, the values obtained from studies around the world were compared to relevant guidelines and regulations. The Cooperative Research Centre (CRC) for Catchment Hydrology undertook a similar study in the form of a bibliography, or chronological review, of urban stormwater quality literature (Duncan, 1995b). Rather than tabulating data in the form of concentration ranges or effects like Makepeace, the CRC report focused on stormwater processes. The review of over 700 publications led to the emergence of the view that stormwater quality processes are very complex and that many factors influence the generation of pollutant loads. He also found that there are large spatial and temporal variations present within and between locations which persist down to small scales (Duncan, 1995a).

2.3 The Need for Data Collection

Much use has been made of that overseas stormwater data and results, but the realisation has emerged during this study and in many other studies, that this is not always appropriate due to differences in rainfall and runoff patterns and catchment characteristics.

There is now a considerable amount of published and unpublished data that has been collected by the water boards throughout Australia, in particular by Sydney Water (part of the Clean Waterways Program) Melbourne Water (Streamwatch and Backyard to Bay programs) and urban planning authorities in the ACT. The Brisbane City Council is also co-ordinating a major study of pollutant generation and transport into the Brisbane River and Moreton Bay (Scott & Davis, 1996).

In South Australia a number of studies have been carried out over the last decade or so. The major studies include

- The Paddocks Wetland (Tomlinson et *al.*, 1993),
- The Glenelg Quantity and Quality project (Argue & Scott, 1996),
- Minkara Wetland at Happy Valley (Gamble, 1997; Jacobi & Murphy, 1996; Daniell & McCarty, 1994)
- Dry Creek Stormwater Study (Passfield & Phillips,1996),
- Greenfields Wetland (Salisbury Council)
- Patawalonga Catchment Management Plan (MFP Australia, 1995)
- Onkaparinga Estuary Stormwater Wetlands (Manning & Stevens, 1991)

Despite these recent studies, the need for further data collection within Australia is still noted by a number of authors. The relative scarcity of monitoring programs capable of providing useful data was discussed by NSWEPA (1997) who also identified a number of processes within stormwater treatment devices that cannot be quantified due to data shortage. Litkowski (1997) identified a lack of general performance data while Wong and Somes (1997) suggested that more comprehensive monitoring programs were required to overcome the high variations experienced in what performance data is available, to establish long term behaviour (Bowmer, 1993; Duncan, 1995b) or investigate basic wetland processes (Breen, 1992). Other researchers (Wong & Geiger, 1997; Cheiw, 1995) have identified a requirement for data to reduce the uncertainty of computer model parameters. In reference to models used to estimate pollutant loads generated from urban catchments, Chiew (1995) came to the following conclusions.

"It is conceivable that there may be sufficient data to provide rough estimates of pollutant loads generated from large urban areas. However, because of the large variability in pollution characteristics, specific monitoring in the area of interest may always have to be carried out if detailed information on the pollution characteristics for the area is required."

The study presented in this thesis is the first that has attempted to determine the performance of the Barker Inlet wetland system, with regard to water quality, and part of the largest stormwater monitoring project carried out in South Australia to date.

2.4 Removal Efficiency of Detention Ponds and Wetlands

The previous two sections have dealt with the problems associated with urban stormwater runoff and studies that have investigated stormwater runoff and likely pollutant concentrations or loads. The next logical step once the problem has been identified is to investigate solutions.

Sources of stormwater runoff result from wet and dry atmospheric deposition, traffic emissions, land erosion and point source pollution (Novotny *et al.*, 1995). An effective way of reducing the degradational effects of pollution associated with stormwater runoff is the utilisation of wetlands, ponds and detention basins (Holler, 1989). These ponds and basins have the potential to trap a significant percentage of pollutants present in urban stormwater runoff by processes that will be discussed in further detail in Chapter 3.

Investigations of wetlands, ponds and basins constructed for stormwater treatment, although becoming more popular, were in the past limited and involved different pond types with different stormwater characteristics. The data available indicates the following (Livingston, 1991):

1. nutrient removal varies widely;
2. flow and seasonal factors influence pollutant removal capability; and
3. removal is consistently better for BOD, suspended solids and metals.

These three points were also noted in the present study.

It is difficult to compare results obtained from different studies as they are often conducted on varying types of ponds in locations of very different climate and hydrology. Sampling methods may vary from event based sampling to base flow sampling or even routine sampling, sometimes testing for different parameters. The structure of the study is usually designed to suit the outcomes required, for example examining the relationship between removal efficiency and surface to area ratio or removal efficiency and hydraulic residence time. The National Urban Runoff Program (NURP) in the United States involved the study of nine ponds with surface to area ratios between 0.01 and 2.85 percent. The pond types included oversized sections of drains below street level, ponds or small lakes on streams, flood control basins, a farm pond, and a golf course pond (Wu *et al.*, 1996). The basic outcomes of the NURP study were that for an adequately sized pond, removal efficiencies greater than 90% for total suspended solids and lead could be obtained. Removal efficiencies for pollutants with significant soluble fractions tended to be lower and more variable with around 65% for total phosphorus and 50% for biochemical oxygen demand (BOD), chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN), copper and zinc (Wu *et al.*, 1996). These findings were supprted by Yousef, et al. (1986). In a study of a wet detention pond at Maitland Interchange (Florida) they found that the particulate fraction of metals from highway runoff were removed to a larger degree than the dissolved fraction. Results of this study are summarised in Table 2.4.

Although these are commonly quoted figures when it comes to pollutant removal, it is less clear whether the removals are based on event removal, long term removal, or indeed how the efficiency has been determined.

Wu *et al.* (1996) reported that the degree of urban runoff contamination may be highly variable and site specific as it is dependent on catchment characteristics and rainfall intensity and duration among other factors. It is not uncommon to experience complete or negative removals at the same site at different times and therefore performance should be based on long term removals of pollutant mass loadings rather than performance on a single event basis. Mulhern and Steele (1988) for example, found that during certain times of the year (September – December) there was a net increase in phosphorus concentrations at the outlet of a wet pond near the Cherry Creek Reservoir in Denver. As stated in Duncan (1995a) pollutant removal is also highly dependent on the initial loads applied to the system. Higher input concentrations of suspended solids in particular, leads to higher removal rate (Randall *et al.*, 1982), as these higher loads tend to include larger particles which settle out more rapidly (Ferrara & Witkowski, 1983).

Wu *et al.* (1996) conducted a study of three urban wet detention ponds in Charlotte, North Carolina to investigate long-term pollutant removal efficiency as a function of surface to area ratio. Surface to area ratio (SAR) is defined as the ratio of pond surface area to the area of the contributing catchment. Four different SARs were investigated; 0.6, 0.8, 2.3 and 7.5 percent. All catchments were predominantly of residential land use. Sampling consisted of automatic sampling at the outlet of each pond and manual sampling at the inlets. Sampling frequency varied depending on whether samples were being taken during or after a storm. Eleven storms were sampled over a 13-month period. Samples were tested for total suspended solids (TSS), total phosphorus (TP), orthophosphorus (OP), total Kjeldahl nitrogen (TKN), ammonia nitrogen (NH_3-N), and total and dissolved lead, zinc, copper and iron. The concentrations of lead and copper were consistently below the limits of detection. The results of the study by Wu *et al.* (1996), which are summarised in Figure 2.2, generally supported the findings of the NURP study that for an adequately sized pond significant efficiency can be achieved. The results of this study are also summarised in Table 2.4 appearing at the end of this section.

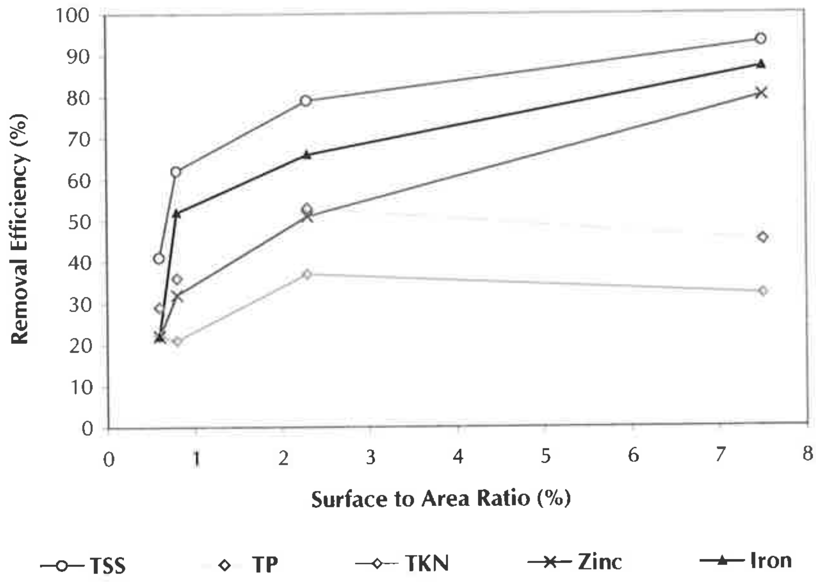


Figure 2.2 Relationship between removal efficiency (%) of various parameters and surface to area ratio as found by Wu *et al.* (1996)

Lawrence (1986) used a different variable to describe basin performance. He reported that hydraulic retention time (HRT) provided the most significant basis for correlating determinand (pollutant) decay values and presented 'Retention of Determinand – Hydraulic Retention Time' curves based on data for a range of lakes and ponds. Rushton *et al.* (1995) also tested the effect of hydraulic residence time on removal of pollutants in a constructed wetland in Tampa, Florida. They found that for a 14-day residence time the TSS and TP loads were reduced by 94 and 90 percent respectively. With a 5-day retention however, TSS was only reduced by 64 percent and the TP load by 57 percent. Similarly, Reinelt and Horner (1995) observed that TP removal was only 8 percent for 3.3-hour retention, but 82 percent for 20-hour retention. Although HRT is likely to be a significant parameter in the description of basin performance, descriptors such as this which incorporate runoff information are not readily available for many sites to allow a comparison of performance (Duncan, 1997).

Pope and Hess (1988) examined the load-detention efficiency of eleven water quality parameters in a dry detention basin in Topeka, Kansas. The basin was a rectangular grass lined pond with a surface area of 4100 m^2 and the catchment area was a five-hectare residential area.

Sampling in this study involved automatic sampling at two inflow and two outflow stations and nineteen storms were sampled over a 14-month period. The sampling method and preparation of a flow weighted composite sample for each event was very similar to that used in this project described in detail in Chapter 5. Results indicated negative removals of some pollutants such as total dissolved solids (TDS) and TKN. Other pollutants were removed satisfactorily, for example; ammonia nitrogen 69%, lead 66% and zinc 65%. Some other pollutants are summarised in Table 2.4. The authors warned that although the results may be indicative of their particular ponds performance, other types of ponds may behave in a very different manner.

Holler (1989) undertook an assessment of the nutrient removal efficiency of a combination grassed swale/wet detention stormwater management system at Springhill, Florida. The pond was one hectare in surface area with a 15.2 hectare catchment largely pervious with low density residential areas. This study also used automatic samplers to monitor storm events, in addition routine bi-weekly samples were used to establish background concentrations of nutrients, and groundwater monitoring to determine the groundwater interaction. Generally it was found that for the parameters tested (TSS, TP, OP, TKN, NO_x, NH_3):

routine outflow < routine inflow < event outflow < groundwater < event inflow

The exceptions were ammonia and TKN which showed strong groundwater influence. For purely surface flows the results were impressive with 64% total phosphorus removal, 98% NO_x, and 77% TKN. Ammonia showed negative removals and TSS removal was negligible due to the very low inflow concentrations (EMC ~ 2.5 mg/L). When groundwater and rainfall was taken into account the results were not as impressive (see results in Table 2.4) probably due to groundwater interaction. Groundwater is not often accounted for in removal calculations, although Holler demonstrated that the interraction should perhaps be quantified.

The Paddocks wetland in South Australia is a similar system to that studied by Holler (1989) in that it is a combination grassed swale/wet detention system. The pond is a similar size with a surface area just under one hectare, the difference is the size of the catchment being 60 hectares, fully developed (residential land-use) with 61% pervious area. The pond is also much shallower with a maximum depth of 1.2 metres compared to 5 metres in Springhill. Both storm and base flows were monitored and the results combined to give the removal percentages over a 150-day period. The efficiency was 92% for TSS, 73% TP, 64% TN, 14% zinc, 91% lead and 33% for iron. Removal was generally higher than in the study by Holler (1989), however no groundwater interaction was investigated.

Ferrara and Witkowski (1983) examined the effect of yet another variable influencing the performance of pollution control ponds. They examined the influent and effluent concentrations as well as the particle size distribution of stormwater in a detention basin in Hillsborough Township, New Jersey. The aim was to develop a relationship between particle size distribution, settling velocity and trap efficiency. They found that the basin was effective at reducing solids, TP and COD, however TKN concentrations and loadings were generally increased. The basin effectiveness appeared to be related to the processes of equalisation and sedimentation. As an equalisation basin the influent water was diluted or mixed with the pre-storm basin contents and as a result the dry weather water quality in the basin is important. As a sedimentation basin, significant removals of particulate material were obtained and the study provided an estimate of the fraction of pollutant load that is settlable. Generally it was found that removal of sediment was greatest for the fraction greater than 105 μm in size. Much lower removals were obtained for the less than 10 μm size range which is reflected in the removal efficiencies obtained for TP. It is these very fine particles which phosphorus, along with many species of metals, tend to adsorb to. This study, along with that of Duncan (1998) highlights the importance of initial concentrations and how the apparent effectiveness of a system may be influenced by the sediment characteristics.

Duncan (1998) used a multiple regression approach in an attempt to identify relationships between water quality improvement in storage and the large range of explanatory variables such as surface to area ratio, hydraulic residence time and hydraulic loading rate (HLR). He examined data from 76 sites in Australia and around the world reported in urban quality literature, including many of those mentioned above. It was found that, for suspended solids, hydraulic loading and input concentration were the most important explanatory variables in determining basin performance while hydraulic loading and design index (a rating of likely effectiveness according to current design methods (Duncan, 1998)) were important for nitrogen and phosphorus. This suggests that area is more important than volume in the removal of these pollutants. The generally accepted impression that wetlands perform better than open ponds is related to the dominance of surface area in dictating pond efficiency. A shallower wetland usually does perform better than a deeper pond of the same volume, however, statistically a wetland is often indistinguishable from a pond of the same area (Duncan, 1997). This is not surprising since many of the important removal processes such as settling, biological action in sediments, solar radiation effects and aeration are all dependant on surface area more than volume (Duncan, 1998).

Table 2.4 Removal efficiency experienced in various wetlands and ponds

Study	Pond Type	Suspended Solids	Total Phosphorus	Total Nitrogen	Total lead	Total Zinc	Total Iron	Notes
Ginninderra, Canberra (1)		2 %	10 %	7 %				
The Paddocks, South Australia (2)	Wetland (wet detention & plants)	88 %	62.3 %	43 %	89 %	93 %	12.5 %	
Katoomba, N.S.W. (3)		59 %	30 %	29 %				
Burke, Washington DC (4)	Wet Pond (33 ML)	36.8 %	59.2					
	Wet pond (36 ML)	86.8 %	69.8					
	Dry Pond (3.5 ML)	77.3 %	26.2					
Maitland Interchange, Florida (5)	Wet detention basin		90.1 %	35.7 %	54.5 %	88.3 %		← dissolved fraction
			11.4 %		95.1 %	96.2		← particulate fraction
Springhill, Florida (6)	Grassed swale/wet detention basin	0	64 %					← event
		-1 %	43 %					← routine
Montgomery County, Maryland (7)		60 %	15 %		80 %	60 %		Medians of 33 storm events
Hillsborough, New Jersey (8)	Wet detention pond	42 %	27 %					Weighted mean of 3 storms
Bundoora, Victoria (9)	Billabong/lake	94 %	85 %	78 %	88 %	93 %		
	Swamp 1	70 %	50 %	28 %	20 %	79 %		
	Swamp 2	> 82 %	40 %	52 %	0	28 %		
Charlotte, North Carolina (10)	Wet detention ponds	41 %	29 %			22 %	22 %	← SAR = 0.6
		62 %	36 %			32 %	52 %	← SAR = 0.8
		79 %	53 %			51 %	66 %	← SAR = 2.3
		93 %	45 %			80 %	87 %	← SAR = 7.5
Topeka, Kansas (11)	Dry detention basin	2.5 %	18.5 %		66 %	65 %		

(1) *Department of Territories (1986)*
(2) *Tomlinson et al. (1992)*
(3) *Swanson (1994)*
(4) *Randall (1982)*
(5) Yousef et al. (1986)
(6) Holler (1989)
(7) *Grizzard et al. (1986)*
(8) *Ferrara and Witkowski (1983)*
(9) *Graham (1990)*
(10) *Wu et al. (1996)*
(11) Pope and Hess (1988)

2.5 Wetland Design

Highlighted in the last section was the fact that constructed wetlands and detention basins are increasingly being used as a means of improving the quality of stormwater. Often the aim is to treat to a standard suitable for discharge to the marine environment (meeting water quality guidelines such as ANZECC, 1992), or for introduction into local aquifers for storage and re-use (Dillon & Pavelic, 1995). While the potential re-uses require strict water quality standards, the design of wetlands is, more often than not, based on simple guidelines or rules of thumb, commonly based on average retention times.

Australia is subject to high variability, both spatially and temporally, in rainfall and therefore runoff characteristics (McMahon & Finlayson, 1991). Accompanying the variable runoff patterns are associated variations in the washoff characteristics of catchments. It has been discussed by a number of authors (Cullen, 1989, Hensel *et al*, 1998, Raisin *et al.*, 1997; Somes & Wong, 1994 and Murphy *et al.*, 1998) that a significant proportion of a catchment's annual pollutant export is often contained in a small number of runoff events. In addition to this variability in pollutant loads from event to event, there is also a variation in pollutant concentrations within each event. For any given runoff event, a significant proportion of the pollutant load is often transported in the early part of the storm. Design guidelines for constructed stormwater wetlands for water quality improvement therefore need to address both the variable pollutant concentrations in stormwater and the ephemeral nature of stormwater flow (Somes & Wong, 1994).

Fisher and Stricker (1992) have noted that although constructed wetlands have been shown to be feasible in the control of water pollution, to date their performance in the treatment of wastewater has been variable. They stated (in 1992) that "*this indicates that the current state of knowledge is insufficient to design and operate constructed wetlands to achieve reliable and efficient treatment with minimal risk of failure and poor performance*" and that "*further work is required to fully establish the long term viability of constructed wetlands in water pollution control and to optimise design parameters....*"

2.5.1 Early Design Guidelines

Somes and Wong (1994), in a review of practices in the utilisation of wetlands for stormwater pollution control found design guidelines, particularly with regard to hydrodynamic performance, to be *ad hoc.* Most of the design parameters were based on an adaptation of wastewater treatment wetlands and in most cases it was not clear how the variation in hydrologic and pollutant loading characteristics between stormwater and wastewater had been incorporated into the design guidelines.

The design guidelines for stormwater wetlands constructed for water quality improvement are aimed at determining the most appropriate dimensions and hydrologic regime to promote the removal of pollutants via the many physical, chemical and biological mechanisms operating within the wetland. However, in most cases examined by Somes and Wong (1994), the adopted approaches varied and tended to be site specific. The storage volume of the wetland defines the retention time of the system for any given inflow rate (Somes & Wong, 1994) and is just a combination of the nominated depth and surface area. Some recommendations for storage area, for example those reported in Livingston (1991) are aimed at trapping the first flush of the stormwater runoff. According to Livingston (1991), in Florida the first flush equates to the first 25mm of runoff, which carries up to 90 percent of the pollution load from a storm event. The Florida Stormwater rule implemented in 1982 required all newly constructed stormwater discharges to use best management practices (BMPs) to treat the first flush of runoff (Livingston, 1986) which range from a minimum volume of 25mm runoff up to 2.5 times the percent of impervious catchment area. In the same paper Maryland legislation relating to stormwater management is quoted as requiring a detention time of 24 hours for the one-year storm, or a surface area of three percent of the contributing drainage area.

Generally fundamental parameters related to the site such as hydrologic variability of runoff and characteristics of pollutants are neglected which limits their application to other sites. In a recent study by Somes and Wong (1998), the authors used pluviograph data to run a continuous simulation of wetland storage behaviour in variously sized storages located in seven Australian capital cities. The study found that the volume required to contain a similar proportion of runoff for a given period varied significantly from site to site. These findings challenge many of the current guidelines for sizing wetlands (such as those based on average retention times) which generally ignore rainfall characteristics and assume that a constant volume is required to contain runoff regardless of the location of the storage.

Most of the design guidelines used are based on wetland storage volume, surface area, depth, retention time and length to width ratio (Somes & Wong, 1994). A majority of these parameters are of course interrelated and can be expressed in terms of the fundamental criteria of surface area and depth. The combination of these criteria gives the storage volume, which in turn can be expressed as retention time. The specification of these two parameters, and hence the storage volume and retention time, are generally related to the desired removal of pollutants, or in other words, the wetland efficiency.

Anderssen *et al.* (1990) reported that initially the design of water quality ponds was based on purely experimentally derived information. In fact the *modulus operandi*, as they put it, was as follows; decide on the efficiency (for example, percentage sedimentation) to be achieved, then use experimentally derived "master curves" relating percentage removal of a particular pollutant to average residence time to determine the average retention time required. The curves developed by Lawrence (1986) were, and still are, commonly used in this process. The next step is the assumption that the global retention time is equivalent to the average retention time so that the volume of the pond can be determined. This is very similar to the "three step method" recommended by the NSW Department of Housing (1993)

Sedimentation, although not the only removal mechanism involved, is the major pathway by which pollutants are removed from stormwater. The following equation (Fair & Geyer, 1954) is for sedimentation in a wastewater wetland and relates sediment removal to wetland area.

$$R = 1 - \left(1 + \frac{1}{n} \frac{v_s}{Q/A} \right)^{-n} \qquad \textbf{(Equation 2.2)}$$

where: R = fraction of initial solids removed
v_s = settling velocity of particles
Q/A = rate of applied flow divided by the surface area of the basin or wetland
n = turbulence or short-circuiting parameter

As Equation 2.2 describes, the rate of sedimentation is dependent on the settling velocity of the particles, which is in turn related to the particle size distribution of the sediment. The unsteady nature of stormwater flow, unlike wastewater flow, is also likely to mean there will be a significant variation in the value of Q from event to event and during each individual event. It is therefore unlikely that experimentally determined curves could be successfully applied to other locations. Wong and Walker (1998) collated data from various studies relating to particle size distribution of street runoff and found the sediment characteristics of Australian catchments to be finer than elsewhere in the world. A study of stormwater runoff in Sydney by Ball and Abustan (1995) found approximately 70 percent of sediment to be less than 62 µm. This is significantly different from studies in America and European catchments where typically only 20 percent of sediment is less than 100 µm (Lloyd *et al.*, 1998). These findings re-enforce the suggestion that guidelines developed in the United States are not directly transferable to Australian catchments. Australian application of American guidelines would more than likely lead to inadequacies in the design of pollution abatement facilities owing to the finer nature of Australian sediment and the associated slower settling characteristics.

2.5.2 Recent Developments in Stormwater Wetland Design

With many of the shortcomings in wetland and pond design being identified, much research is being undertaken in an attempt to improve the design and transferability of guidelines.

Simple empirical models for the design of stormwater ponds and wetlands have been used in Australia for over 15 years (Lawrence & Breen, 1998). Concern, however, is growing about the appropriateness of these models when applied to different locations with different climates, hydrology and soils, raising questions about the long-term performance of these ponds and wetlands with respect to pollutant removal ability. Even the earliest studies on constructed wetlands (Weir, 1976; Finlayson and Mitchell., 1982; Finlayson and Chick, 1983; Finlayson, 1983) stressed that the performance of wetlands could be improved if the internal pollutant removal mechanisms were better understood and this knowledge then incorporated into appropriate design and operational management techniques (Fisher & Stricker, 1992). In 1994 the CRC for Freshwater Ecology initiated a research program aimed at gaining a better understanding of the physical, chemical and biological processes involved in pollutant interception. Arising from this research was a model and guidelines for the design of pollution control ponds and wetlands.

The CRC Guidelines guide the user through a number of principles leading to the selection of a treatment train appropriate to the site hydrology and pollutant characteristics (Lawrence & Breen, 1998b). Each of the major interception, transformation and transfer processes are addressed and a model provided to estimate the significance of each component in order to fine-tune the design process. The guidelines also include a computer-based water quality model consisting of five sub-models – mass balances/advection; adsorption and sedimentation; sediment redox and pollutant release; algal growth; and mixing/oxygen transfer. The model is run on a daily time step basis, with the capacity to include historical runoff and water quality data for the specific site (Lawrence & Breen, 1998b).

Anderssen *et al.* (1990) point out that although mathematical modelling can assist directly in the design of ponds as pollution control devices, the final layout of the pond actually constructed must comply with the site specific local constraints such as topography and geology. This is a step in the design process which modelling can only accommodate through simulation (Anderssen *et al.*, 1990). For the purpose of pollution control, the simulation involves studying the flow and sedimentation dynamics in terms of appropriate two-dimensional elliptic partial differential equation models.

Somes and Wong (1994) note that owing to the highly variable nature of catchment runoff and the associated pollutant concentrations the appropriate storage volume should be based on a continuous simulation approach. This would involve a behavioural analysis of the storage volume using historical or stochastically generated streamflows. The storage volume could then be selected based on long term performance rather than prescribed performance for a single event. In situations where rainfall/runoff data is limited or unavailable, Somes and Wong (1998) recently have outlined a method to size wetland storages using pluviograph data typically available from the Bureau of Meteorology. Their method is based on the annual rainfall depth, the sites location and the monthly distribution of rainfall and dry days. This approach highlighted the hydrologic variations even with Australian catchments.

2.6 Urban Stormwater and Wetland Modelling

The modelling of urban stormwater has long been identified as a required research interest. The development of models to simulate the behaviour of a runoff hydrograph and its associated quality "pollutograph" under varying rainfall conditions have been studied since the early 1970s (Chen & Shubinski, 1971).

There are two basic reasons for developing water quality models:

1. To increase the understanding of the system in the management of water systems;
2. To use this better understanding of the system in the management of water systems (Thomann, 1982).

An accurate, or reliable, model can eliminate the need for rigorous sampling which can be both expensive and time consuming (Daniell & McCarty, 1994). However, as James and Elliot (1991) indicated, increased amounts of data collection through automatic water sampling needs to be undertaken to provide the information necessary for calibration of these models (Jewel & Adrian, 1981). Jewel and Adrian also suggest that the period over which the data are collected to calibrate the models should span at least one year, so that storms are sampled over each season.

A number of hydrologic models have been developed both in Australia and overseas to predict runoff volumes and flow rates in drainage systems. Many of these commercially available models are also capable of predicting water quality by estimating build-up and wash-off of pollutants within the catchment.

2.6.1 Stormwater Modelling

The Stormwater Management Model, commonly referred to by the acronym SWMM, is a comprehensive model capable of simulating runoff quantity and non-point source quality problems in urban areas (Ball & Ferguson, 1994). The model was originally developed in 1969-71 by a consortium of organisations however since then the model has continuously been modified and improved by the original developers as well as users of the model.

SWMM is partitioned into a number of modules, which perform tasks linked to the generation, the collection, the transport, and the disposal of stormwater runoff. While in some models, for example ILSAX (O'Loughlin, 1990), the components combine to form a complete model, in SWMM the various components are designed to operate independently or together. Each block in the model controls a different aspect of the simulation; the RUNOFF block simulates the collection of stormwater and pollutants while the TRANSPORT and EXTRAN blocks simulate the motion of the water and pollutants within the drainage system (Ball & Ferguson, 1994). The integration of the blocks is shown in Figure 2.3.

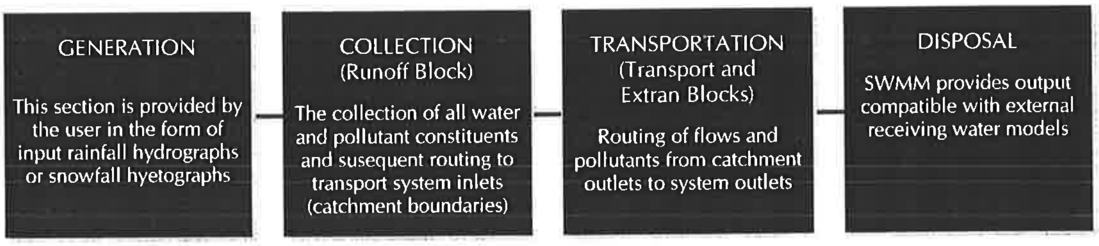


Figure 2.3 **Conceptual representation of SWMM**
(Source: Ball & Ferguson, 1994)

While SWMM is widely used in both in Australia and particularly overseas, White and Cattell (1992) point out that deterministic models such as SWMM and STORM are very complex and require rigorous calibration. They go on to suggest that they have limited use in Australia as there are few catchments which have been monitored extensively enough to provide the data necessary for calibration. An alternative to these complex data intensive models is AUSQUAL (Gamtron, 1990) which obtains results from a limited data base. AUSQUAL uses two main programming components; the hydrologic model and the water quality model in much the same way as AQUALM-XP (discussed later). Figure 2.4 shows the process interactions involved in the model.

For each land use type (it allows up to five) a coefficient for imperviousness, continual rainfall loss and pollution availability is allocated. For each sub-catchment weightings can be allocated for runoff continual loss, loadings for pollutants (up to four) and percentage of different land uses. The results of this simple type of modelling have been used effectively in six urban catchments in Sydney for calculating hydrology, gross export of pollutants and the shape of pollutographs (White & Cattell, 1992).

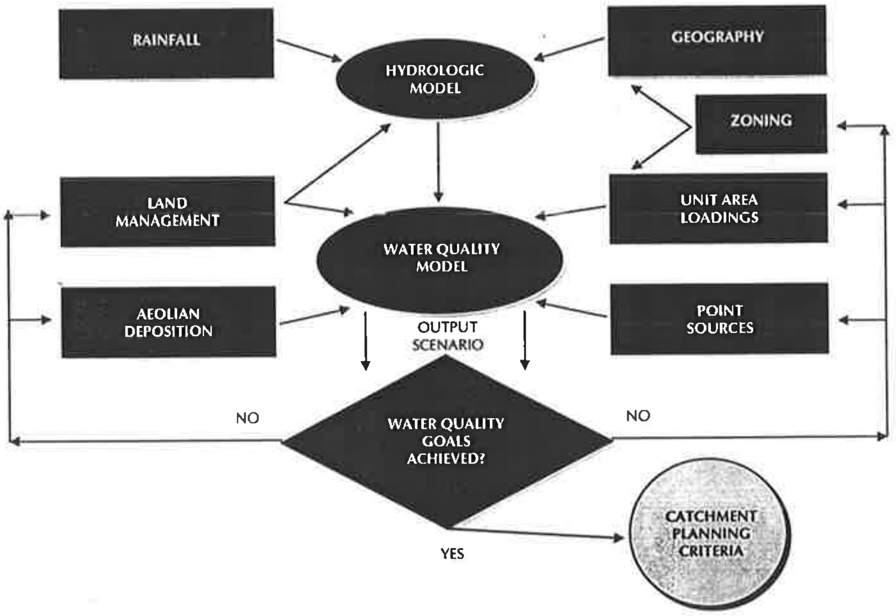


Figure 2.4 Interactions considered by AUSQUAL
(Source: White & Cattell, 1992)

While AUSQUAL is effective at predicting runoff and pollutant export, there is no facility to simulate pond storage behaviour. AQUALM-XP (WP Software, 1992) is a networked system which generates runoff and pollutant exports on a continuous daily basis and routes these flows through gross pollutant traps (GPTs) and/or water pollution control ponds (WPCP). The model has two modes of operation; the EXPORT mode generates runoff and pollutants and routes them through the GPT or WPCP while the RIVER mode routed the runoff and pollutants generated from the EXPORT mode through lakes and rivers (Phillips *et al.*, 1992). AQUALM-XP comprises a series of sub-models including:

1. Rainfall/runoff model;
2. Non-point source pollution export model;
3. GPT model;
4. Water pollution control pond model;
5. Lake loading model (adaptation of Vollenweider lake loading model); and
6. River loading model.

The data requirements are appropriate considering the general lack of good quality data available in Australia. AQUALM-XP has been successfully applied to catchments in South Australia (Jacobi & Murphy, 1996; Milne & Clarke, 1991) as well as subtropical catchments in Brisbane (McAlister *et al.*, 1995).

Simple empirical models of ponds, based on hydraulic retention time/pollutant retention correlations have been in use in Australia since the early 1980's (Lawrence & Baldwin, 1997). In recent years there have been considerable advances in the understanding of water quality and ecological processes in ponds and wetlands and research attention has been focusing on developing models which incorporate the many complex physical, chemical and biological processes which occur in these ecosystems. These models are capable of predicting wetland or pond behaviour and performance under a range of conditions by simulating the ecology of the systems and have the potential to become an invaluable design tool when combined with catchment hydrology and pollutant export models.

2.6.2 Ecological Modelling of Wetlands and Pond Systems

In recent years a number of authors have developed ecological process models which simulate the ecology of wetlands and behaviour of macrophytes as well as the hydrology. Brown *et al.* (1994) for example, developed a model to simulate the transport of pollutants into a wetland in an effort to predict effluent concentrations, removal efficiency and long term bioaccumulation. Moreno-Grau *et al.* (1996) developed a similar model, which included thermal and biochemical sub-models. In this model linear and differential equations describing heat and mass balances were solved using numerical methods to simulate wetland performance.

Christensen *et al.* (1994) developed a first generation wetland ecosystem model for four constructed freshwater marshes at the Des Plaines River Wetland in Illinios, USA. The model included hydrology, sediment and phosphorus algorithms to predict wetland function under varying hydrologic and chemical loadings. The authors described the model as a "dynamic and casual model" (Jørgensen, 1988) that presented an integrative approach to understanding wetland processes without specifically taking into account internal spatial movements of water and chemicals.

Dørge (1994) developed a more specific model targeting the simulation of nitrogen dynamics. A general simulation model (MIKE 11 WET) was developed for freshwater wetlands to determine the retention and removal of nitrogen in wetlands as water flows through them. The model consisted of a simple hydrological sub-model and a more complex biological sub-model including heterotrophic nitrogen dynamics and plant uptake. The whole biochemical pathway from mineralisation of organic matter to ammonia and further to nitrate in the oxic zone before denitrification was explicitly modelled (Dørge, 1994).

As Kadlec and Hammer point out, the dynamics of wetlands have been represented by a variety of ecological models which describe or predict certain features, such as nutrient cycling (Dørge, 1994) or biomass productivity. One popular modelling technique partitions the wetland systems into compartments such as litter, sediments, live and dead biomass, and accounts for the transfers between them (Mitsch *et al.*, 1993).

Material balance calculations are done most easily using a compartmental model such as that shown in Figure 2.5 which can be used to describe the quantities and transfers associated with physical wetland processes. While many detailed compartmental models have been developed, sometimes of great complexity, few spatially distributed models exist (Mitsch 1983). The mobility of waterborne components is more often than not neglected, however the authors suggest that water flow models should be incorporated into the overall analysis. Kadlec and Hammer (1988), for example, developed a simple mathematical model, which permitted the dynamic simulation of wetland hydrology and nutrient driven interactions between wastewater and the wetland ecosystem. A hydrology model was used to predict the overland flow while ecosystem phenomena were represented using a one-dimensional spatially distributed compartmental model. In terms of mathematics, including water flow in the model converts the model from a set of ordinary differential equations to one which also contains partial differential equations.

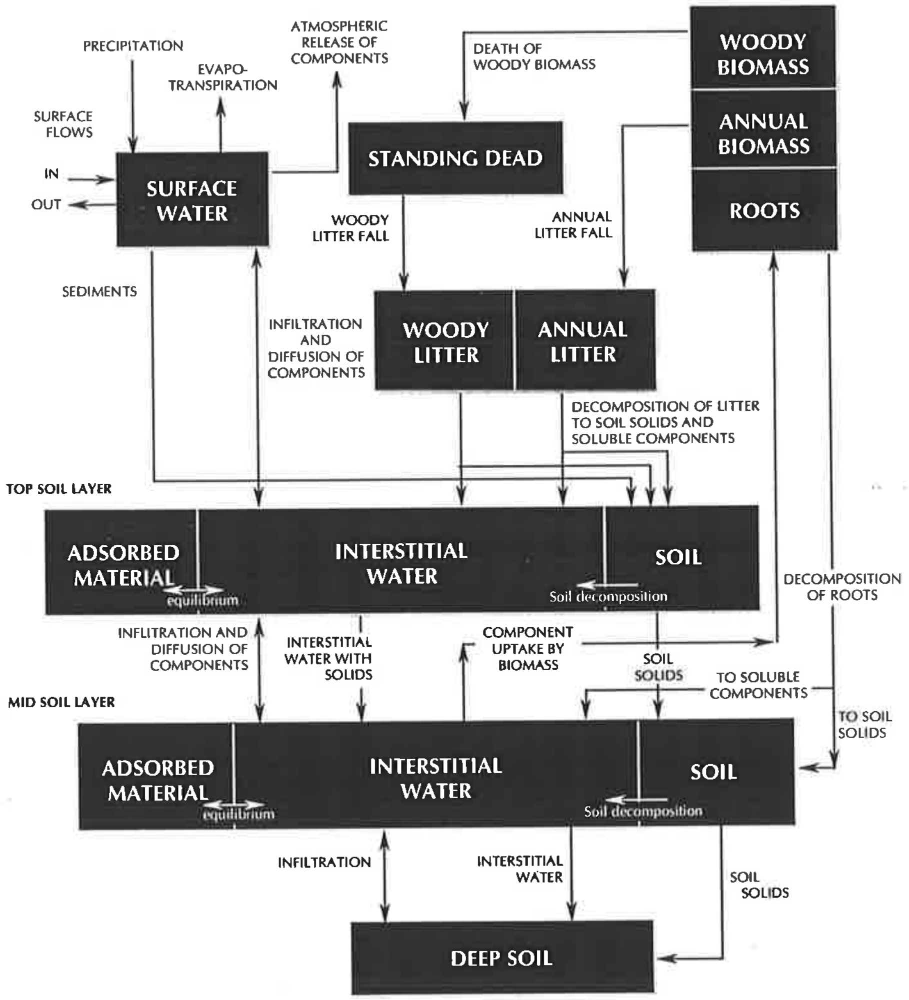


Figure 2.5 **Wetland ecosystem model**

(Source: Kadlec & Hammer, 1988)

Lawrence and Baldwin (1997) have also recognised that simple models are no longer adequate for modelling water pollution ponds and that a more rigorous model that can provide for a range of catchment and climatic conditions is necessary. The recent model developed by the CRC for Freshwater Ecology (Lawrence & Breen, 1998b) contains the following system components:

i) Particulate settling processes;

The component describing the process of particle setting and the influence of particle size distribution.

ii) Redox processes;

This is the component describing the process of sediment reduction during which chemically bound and adsorbed nutrients and metals are released into the water column. This involves three sub-components:

a) loading of organic matter

b) transfer of O_2 from the atmosphere

c) composition of the sediment in relation to the six dominant reduction phases (DO depletion, denitrification, nitrate, sulphate and iron reduction, and methane fermentation);

iii) Pond mixing processes;

Describes the pond mixing primarily as a result of wind and heat adsorption but also includes sub-components for oxygen transfer and algal composition.

iv) Role of macrophytes;

Describes the process whereby macrophytes influence the particulate interception processes and sediment redox processes.

v) Role of algal biomass;

The algal biomass can influence sediment redox processes as well as responding to available nutrients and light.

Development is proceeding on the incorporation of this model into existing models such as AQUALM (Lawrence & Baldwin, 1997).

2.6.3 Model Selection

It is commonly thought that a more complex model should more accurately account for the reactions taking place in a real system, but Jørgensen (1995) points out that this is not necessarily true. Developing a more complex model, and thus introducing many more parameters increases the level of uncertainty as all parameters are estimated to some degree either by field observations, laboratory experiments or calibrations. Parameter estimation is never completely error-free, and the level of uncertainty is increased as these errors are carried through into the model. The problem of the appropriate model complexity is of particular interest in ecological modelling as was put so well by Lung and Light (1996).....

"Like most modelling of natural ecosystems, wetland modelling is a formidable task: the goal of accurately characterising all of the key parameters and processes in natural systems immediately becomes a trade-off between completeness and the availability of data and time."

For the purposes of this study the simple ecological model developed by the CRC for Freshwater Ecology (as described by Lawrence & Breen, 1998) was selected. This model was readily available and simple to operate using Microsoft® EXCEL 97. The model represented a recent development in numerical models, had a reasonable complexity and would benefit from field verification. The model is described in more detail in Chapter 7.

△3

Wetland Processes

Constructed ponds and wetlands are widely recognised as being effective in the treatment or control of pollution, as well as for restoring urban stream values, for their recreational and aesthetic qualities, and for conserving flora and fauna (Lawrence & Breen, 1998). Ponds and wetlands are particularly attractive with their ability to substantially reduce the discharge of stormwater pollutants to receiving waters.

Treatment facilities tend to be either predominantly open water systems (termed ponds) with associated macrophyte 大型植物 zones, or predominantly macrophyte systems (termed wetlands) with some pondage to adsorb variations in flow. One of the main functions of the water plants in wetlands, in addition to slowing down the flow of water, is the provision of a suitable habitat for microorganisms. Microbes are associated with the plant shoots, roots (rhizosphere) and litter decomposition. Effective microbial activity is dependent on the environment at the base of the plants as this is where most of the water purification, aside from sedimentation, takes place (Sainty *et al.*, 1994).

Sedimentation has been identified by many authors as being the major process by which pollutants, including TSS and metals, are removed by constructed wetlands and ponds. These facilities can, however, also be effective at nutrient removal. The extent to which this happens depends on the area, type and age of the wetland, water quality, and the time taken for the water to move through the wetland (detention time) as was seen in Section 2.3. Phosphorus removal, in particular, is a difficult task during any water treatment, and wetland technology is no different. In general, on a per unit area basis, wetlands are not efficient at removing phosphorus.

Although the focus of pond and wetland design has traditionally been the reduction in the mass of pollutants exported from catchments, it is now appreciated that the form of these potential pollutants is just as important as the quantity. A pond or wetland that transforms even a small portion of the intercepted nutrients, metals or organics into a more bioavailable form, as a result of sediment reduction processes, may seriously undermine an otherwise harmless pollutant export condition (Lawrence & Breen, 1998). An understanding of the various cycles in wetlands is therefore important when designing wetlands and ponds as treatment systems. Some of the earliest studies on constructed wetlands stressed that the performance of wetlands could be improved if the internal pollutant removal mechanisms were better understood (Fisher & Stricker, 1992).

Below is a brief description of the internal processes at work in pond and wetland systems and their role in the treatment of stormwater.

3.1 Sedimentation

A major function of wetland ecosystems is the removal of suspended solids from the water column. The removal is the result of a rather complicated set of internal processes, shown in Figure 3.1. [After the suspended material reaches the wetland, it joins large amounts of internally generated suspendable materials, and both are transported through the wetland.] Sedimentation and trapping, as well as resuspension, occur en-route as does the generation of suspended solids through a number of activities occurring both above and below the water surface. Low water velocities, coupled with the presence of aquatic macrophytes, promote the fallout and filtration of solid materials. This transfer from the water column to the sediment not only has important consequences for water quality, but also for the properties and function of the wetland ecosystem (Kadlec & Knight, 1996).

在途中

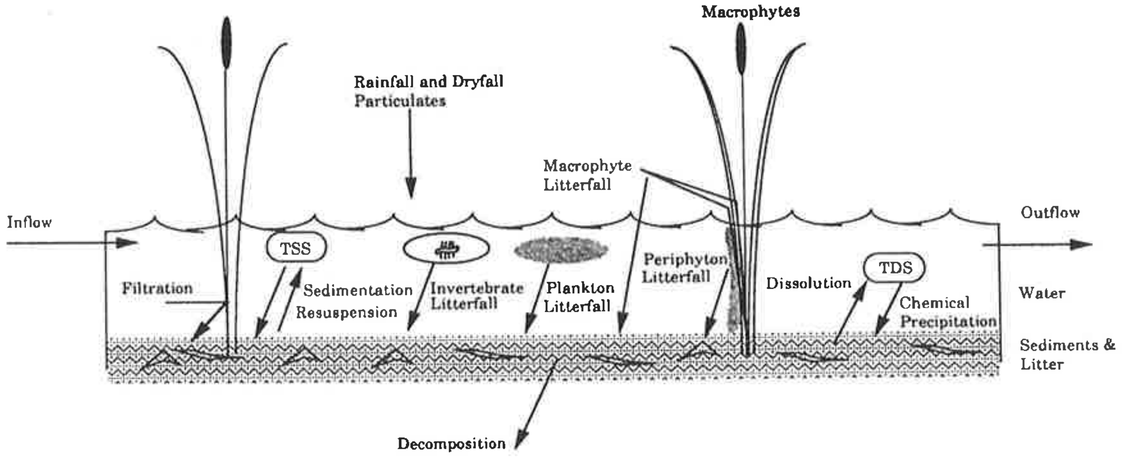


Figure 3.1 Suspended solids storages and transfers in wetland environments.

(Source: Kadlec & Knight, 1996)

Many pollutants, such as metals, organic chemicals and phosphorus, are associated with the incoming suspended material and partition strongly to suspended matter. The removal of suspended solids is therefore important in the removal of many of the pollutants associated with urban stormwater.Settling of Particulates

The slow moving water velocities in ponds and wetlands often permit time for the physical settling of TSS. Figure 3.2 demonstrates the effect of residence time (the time taken for water to flow through the wetland) on the removal of suspended solids for two different systems. As can be seen, the wetland system with its associated macrophytes and typically shallower water depths performs much better than the straight pond system. It should be noted at this point that the line representing the straight pond system was based on a limited data set.

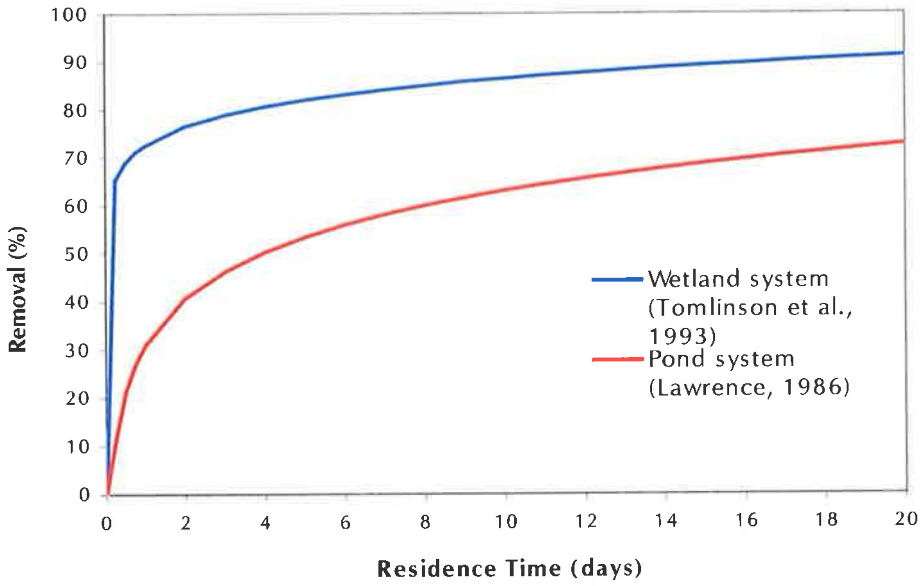


Figure 3.2 **The effect of residence time on the removal of suspended solids for two different systems**

Settling is also dependent on the physical characteristics of the particles. The particle size and density influences the settling velocity which in turn influences the amount of solids removed in a given time as demonstrated by Equation 3.1 (Fair & Geyer, 1954).

$$R = 1 - \left(1 + \frac{1}{n}\frac{v_s}{Q/A}\right)^{-n} \qquad \textbf{(Equation 3.1)}$$

where: R = fraction of initial solids removed

v_s = settling velocity of particles

Q/A = rate of applied flow divided by the surface area of the basin or wetland

n = turbulence or short-circuiting parameter

Colloidal material present in the inflow is very stable and extremely slow to settle. The slow settling of fine material is of concern as this is the size fraction that other pollutants tend to adsorb to.

3.1.1 Chemical Precipitation

There are several chemical reactions that can produce sediment and promote settling under the right circumstances. Some of the more important are the oxyhydroxides of iron, calcium carbonate, and divalent metal sulphides. The hydroxides are typically flocs with the possibility of coprecipitates (Kadlec & Knight, 1996).

3.1.2 Resuspension

In treatment wetlands physical resuspension is usually not a dominant process, as water velocities are normally too slow to dislodge sediment from the pond bed. The exception to this is under extreme flow conditions when souring can occur, particularly in the inlet zone. Three other mechanisms for resuspension can occur in wetland environments (Kadlec & Knight, 1996).

3.1.2.1 Wind driven turbulence

In open water areas, wind-driven currents cause surface flow in the direction of the wind and return flows along the bottom in the opposite direction. Sometimes these return flows can be greater than the net velocity through the wetland and can be enough to cause resuspension.

3.1.2.2 Bioturbation

Most animals, of all types and sizes, can cause resuspension. Feeding carp have been known to cause a particular problem both in South Australia, and elsewhere in the world (Kadlec & Hey, 1994). Human sampling activities can also result in local resuspension of sediments.

3.1.2.3 Gas lift

Gas lift occurs when bubbles of gas become trapped in or attached to particulate matter. There are a number of gas generating reactions in wetland environments, the more important being photosynthetic production of oxygen by algae and the production of methane in anaerobic zones (Kadlec & Knight, 1996).

In heavily vegetated wetlands, the litter and root mats provide substantial stabilisation of soils and sediments which limits, but does not eliminate, resuspension.

3.2 Nitrogen Removal

In wetlands nitrogen is present in organic and inorganic forms (cf. Chapter 2), with organic nitrogen being the dominating form in many systems (Kemp *et al.*, 1982; Nixon, 1981).

The various forms of nitrogen are continuously involved in chemical transformations from inorganic to organic compounds and back again. Some of the processes require energy and some release energy which is then used by organisms for growth and survival. All of the transformations are a necessary component of a successful wetland ecosystem and most of the chemical changes are controlled by enzymes and catalysts produced by the organisms that they benefit (Kadlec & Knight, 1996).

Five principal processes occur in wetlands to transform nitrogen from one form to another. These are:

1. Nitrogen fixation;

2. Ammonification (or mineralisation of organic nitrogen to ammonium);

3. Nitrification of NH_4^+ to nitrate (NO_3^-) in aerobic environments;

4. Denitrification of NO_3^- to dinitrogen (N_2) in anaerobic environments; and

5. Assimilation of nitrogen species by plankton and other aquatic vegetation.

Coupled to these transformations are a number of physical processes, which transport or translocate nitrogen from one place to another without any molecular alteration. These transfer processes include:

1. Particle setting and resuspension;

2. Diffusion of dissolved forms;

3. Litter-fall;

4. Ammonia volitilisation;

5. Sorption and desorption of soluble forms of nitrogen (in particular ammonium ions) to soil particles;

6. Seed release; and

7. Organism migration.

These processes are shown in Figure 3.3, which is a simplified diagram of the nitrogen cycle showing the transformations and translocations taking place.

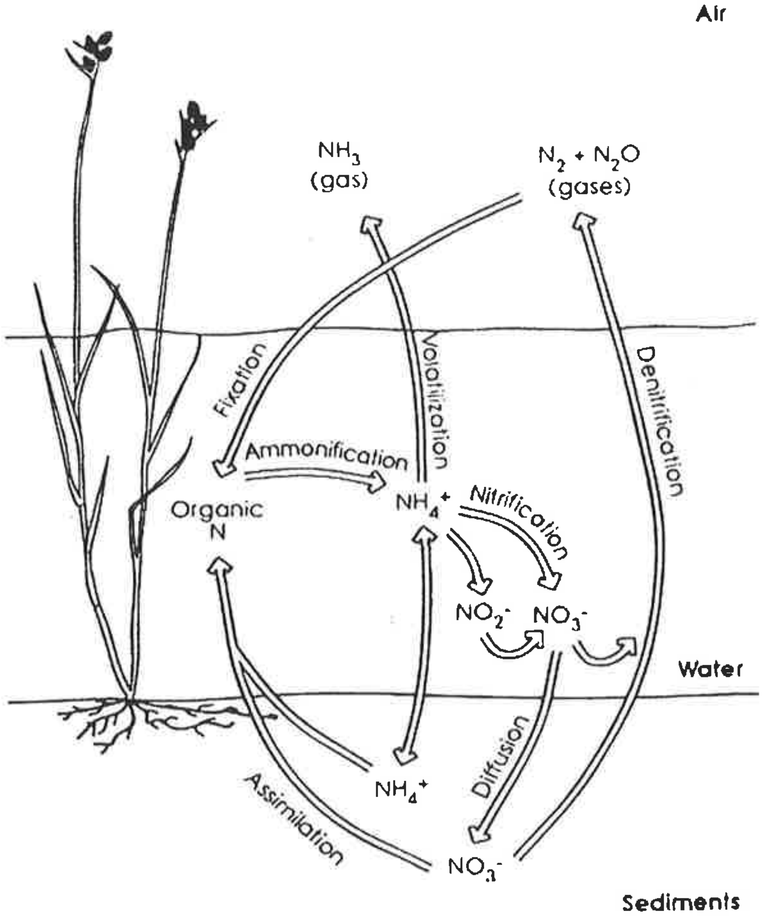


Figure 3.3 **Simplified wetland nitrogen cycle**

(Source: Kadlec & Knight, 1996)

3.2.1.1 Nitrogen Fixation

Nitrogen fixation is the first step in the nitrogen cycle and involves the reduction of dinitrogen to ammonium ions as shown in Equation 3.2.

$$3CH_2O + 3H_2O + 2N_2 + 4H^+ \rightarrow 3CO_2 + 4NH_4^+ \qquad \textbf{Equation 3.2}$$

For nitrogen fixation to take place, microorganisms are required which contain the enzyme nitrogenase. Species capable of nitrogen fixation include some types of algae, bacteria and plankton. The majority of the fixation is carried out by some species of cyanobacteria, also (incorrectly) known as blue-green algae, such as *Anabaena*. Plankton are also capable of fixing nitrogen, however this is dependent on the availability of light and adenosine triphosphate

(ATP) which is generated during photosynthesis. Some plant species are able to fix atmospheric nitrogen because of their symbiotic relationship with nitrogen fixing bacteria. An example is the floating fern *Azolla*, which houses the blue green alga *Anabaena azollae* in its rhizoid (Jones, 1994)

3.2.1.2 Nitrification

Nitrification is a two step process, which is mediated by two separate species of microorganisms known as aerobic chemoautotrophs; aerobic because they require oxygen, and chemoautotrophs because they derive their energy from an external source - the oxidation of inorganic compounds.

In the first step, the microorganism *Nitrosomonas sp.* oxidises ammonia to nitrite, which is an intermediate product. Equation 3.3 gives the reaction taking place.

$$2NH_4^+ + 3O_2 \xrightarrow{nitrosomonas} 2NO_2^- + 4H^+ + 2H_2O$$ **Equation 3.3**

In the second step the species *Nitrobacter sp.* oxidise nitrite to nitrate as shown in Equation 3.4.

$$2NO_2^- + O_2 \xrightarrow{nitrobacter} 2NO_3^-$$ **Equation 3.4**

The *Nitrosomonas* and *Nitrobacter* species are thought to be the main nitrifying species although other species may be capable of similar reactions (Mitchell, 1995).

Because nitrifying bacteria are chemoautotrophs, they are incapable of using complex organic material as a carbon source and instead use carbon dioxide. Nitrifiers are very sensitive to changes in environmental conditions and have very low growth rates, so in order to compete with chemoheterotrophs (which derive their energy from oxidation of organic rather than inorganic compounds) the nitrifiers require low BOD and high oxygen availability. Rates of nitrification are generally slow due to the limited supply of oxygen in the sediments where the bacteria live, however macrophytes are able to enhance this process as will be discussed in more detail later.

3.2.1.3 Denitrification

The biochemical processes that cause nitrogen fixation would ultimately deplete the atmosphere of nitrogen gas if the cycle were not closed by the microbial liberation of nitrogen via denitrification (Kadlec & Knight, 1996).

Denitrification involves the reduction of nitrate to gaseous nitrogen, which is then liberated from solution. It is only when the nitrogen gas is liberated from solution that the nitrogen is completely removed from the system (Mitchell, 1995). The reaction is given by Equation 3.5.

$$NO_3^- + 1.08CH_3OH + H^+ \rightarrow 0.065C_5H_7O_2N + 0.47N_2 + 0.76CO_2 + 2.44H_2O$$

(assuming methanol is the carbon source) **Equation 3.5**

The conditions required for denitrification are different from those required for nitrification as different bacteria, chemoheterotrophs, are involved. Denitrification requires nitrate, anoxic conditions (nitrate used as the terminal electron acceptor), and a carbon and energy source. Whereas nitrification required low BOD, denitrification requires a readily biodegradable source of carbon and energy.

Denitrification generally occurs in the sediments in the anaerobic/anoxic conditions. The rate of denitrification is faster than nitrification which results in minimal accumulation of nitrate in the water (Reddy & D'Angelo, 1994). Kemp *et al.* (1990) has shown that nitrification-denitrification is the major mechanism regulating nitrogen cycling processes and nutrient enrichment in wetlands and ponds.

3.2.1.4 Ammonification (Mineralisation)

Ammonification is the biological transformation of organic nitrogen to ammonia and is the first step in the mineralisation of organic nitrogen (Reddy & Patrick, 1984). This process occurs as a result of the microbial breakdown of organic tissues which contain amino acids, through excretion of ammonia from plants and animals directly, and by hydrolysis of urea and uric acid. The organic forms are converted to ammonia via a complex, energy releasing, multi-step biochemical process. In some cases the energy is used for growth by microbes, and the ammonia is directly incorporated into microbial biomass (Kadlec & Knight, 1996).

3.2.1.5 Nitrogen Assimilation

Nitrogen assimilation refers to the numerous biological processes that convert inorganic nitrogen forms into organic compounds (proteins and amino acids) which then serve as the building blocks for cells and tissues (Kadlec & Knight, 1996). The two forms of nitrogen generally used are ammonia and nitrate nitrogen. Kadlec and Knight (1996) report that ammonia is the preferred form as it is more reduced energetically than nitrate. These authors however, focus on wastewater wetlands, which tend to have a much higher loading of ammonia than stormwater wetlands such as the Barker Inlet system. Aquatic macrophytes utilise enzymes (nitrate reductase and nitrite reductase) to convert oxidised nitrogen to useable forms and the production of these enzymes decreases when ammonium is present. It therefore follows that in wastewater wetlands the use of nitrate as a nitrogen source may be limited by high ammonia concentrations, while in stormwater wetlands nitrate may become the favoured source as reported in Gamble (1997).

3.2.2 Nitrogen Reactions at the Soil-Water Interface

The soil in wetland systems acts as a major nitrogen reservoir and NH_4^+ is constantly regenerated in the soil during decomposition of organic matter. The ammonium-N produced undergoes ion exchange equilibrium between the soil solid phase and porewater resulting in an equilibrium concentration in the porewater which readily diffuses into the water column. Here it is either assimilated by algae or undergoes transformations according to Figure 3.3. A majority of the ammonium is nitrified at the soil-water interface where the redox potential is usually > 300 mV and nitrate formed in the surface aerobic soil zone diffuses downwards in response to the concentration gradient where it is denitrified (Kemp *et al.*, 1990).

During times of resuspension, any ammonium adsorbed to soil particles is rapidly desorbed into the water column and made available for uptake by plankton or for use as an energy source for nitrifying bacteria. In addition to this, relatively high pH conditions (pH>8) during active photosynthesis can result in the conversion of NH_4^+ to NH_3, which is then lost to the atmosphere by volitilisation.

3.2.3 The Role of Aquatic Plants

Vegetation can play a significant role in nitrogen removal by assimilating nitrogen into plant tissue and by providing an environment in the root zone which supports nitrification-denitrification. Efficiency of nitrogen utilisation by plants is highly variable and depends on the type of plant (Reddy & D'Angelo, 1994). Nitrogen assimilation by herbaceous plants such as aquatic macrophytes is normally short term and cycles within the system whereas uptake by wooded plants is longer term but is returned to the system during leaf litter-fall. Plant uptake typically accounts for 16-75 percent of nitrogen removal in wetlands (Reddy & DeBusk, 1987). Wetland plants also have the unique capacity to translocate oxygen to the root zone and thus create an oxidised rhizosphere. This aerobic rhizosphere supports nitrification, while the anaerobic zone adjacent supports denitrification. This nitrification-denitrification is recognised as a major pathway for nitrogen removal in wetlands (Neeley & Baker, 1989).

3.3 Phosphorus Removal

Phosphorus is utilised in wetlands and ponds in a complicated biogeochemical cycle which involves many pathways and diverse sinks and sources which are both temporary and permanent (Kadlec & Knight, 1996). Due to the typically scarce nature of this nutrient in most natural environments and the absence of a significant atmospheric source, wetlands have numerous adaptations to scavenge and sequester this nutrient. Consequently, phosphorus cycling is efficient and extensive in wetland systems. The phosphorus cycle is shown in Figure 3.4.

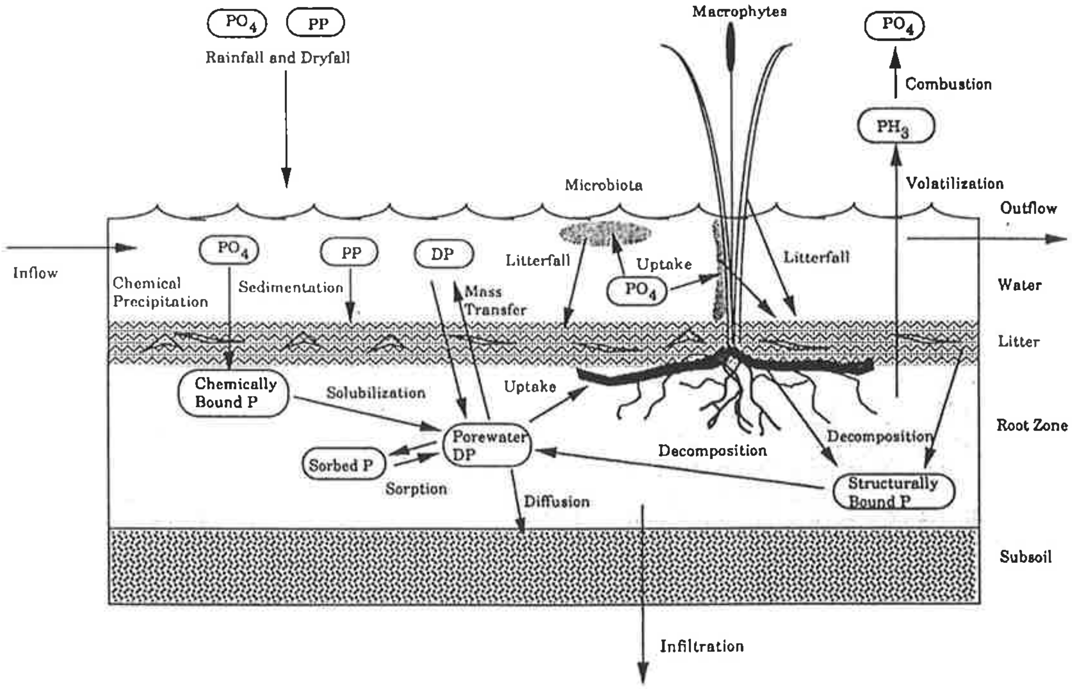


Figure 3.4 **Phosphorus storages and transfers in wetlands. PO_4 = orthophosphate, PP = particulate phosphorus, DP = dissolved phosphorus, PH_3 = phosphine.**

(Source: Kadlec & Knight, 1996)

Phosphorus is cycled rapidly, mostly in the particulate form, by 藻类 algae. Some of the 胶状的 colloidal fraction is lost by hydrolisation when it is converted to orthophosphate and assimilated by biota, while some is deposited in the sediments. Soluble phosphorus is recycled during decomposition of organic matter, release from sediments, and from anthropogenic inputs. Two important physical processes for phosphorus removal in wetlands are sedimentation of particulate phosphorus and sorption of 可溶化 soluble phosphorus.

3.3.1 Sediment Interactions

Phosphorus complexes are highly insoluble and easily adsorbed to particulate matter, therefore much of the phosphorus cycle is centred around exchange between the sediment and water body. There are a number of factors that influence the potential of sediments acting as a sink and the release of phosphorus back into the water column. Biota within the sediment, as well as environmental conditions, affect the transport of phosphorus from the sediment.

The critical area for phosphorus exchange is at the sediment-water interface. The important chemical variables that determine the adsorption-desorption of phosphorus from sediment are the oxidation-reduction (redox) potential, the pH, the calcium concentration, and the degree of agitation of sediment in the water.

The oxygen content in the sediment-water interface and surrounding zone is a key factor in phosphorus exchange, and this is influenced by the bacteria and planktonic invertebrates that live at the interface (Gamble, 1997). The DO content directly influences the redox potential, which in turn defines the iron (II)/iron (III) ratio. A large proportion of phosphorus is adsorbed to ferric hydroxides and oxides. When the redox potential decreases, for example during times of high oxygen demand associated with algal growth, ferric hydroxides and oxides dissolve and phosphorus is released (Kramer *et al.*, 1972). Phosphorus can also be bound as ferric phosphate minerals, and a decrease in redox potential can cause mineral solution. Although these mechanisms have been identified as the reasons behind the release of phosphorus since the early 1930s and 40s, little attention has been paid to release of phosphorus from sediments under aerobic conditions (Boström et al., 1982). Boström *et al.* (1982) have indicated that release of phosphorus under aerobic conditions can be significant and involves mainly the release of dissolved phosphorus existing in the porewater which is readily exchangeable and highly mobile. Essential transport mechanisms in this process are diffusion, wind induced turbulence, bioturbation and gas convection.

3.3.2 Interactions in the Water Column

Phosphorus interactions in the water column involve suspended particulate and colloidal, or adsorbed phosphorus, and dissolved phosphorus.

Phosphate anions, predominantly orthophosphate, can form insoluble salts with a number of metal ions in the water. The pH of the water and the relative concentrations of phosphate ions to metal ions determine the extent to which this happens. This removal of phosphorus from the water column occurs most markedly at pH conditions between 5 and 6 when the phosphate anions bond to positively charged clay particles.

3.3.3 The Role of Aquatic Macrophytes and Other Littoral Vegetation

In addition to the components of the phosphorus cycle already mentioned, there can be significant uptake of phosphorus by the roots of littoral vegetation, macrophytes and by phytoplankton. Organisms within wetlands require phosphorus for growth and incorporate it in their tissues (Kadlec & Knight, 1996). The most rapid uptake is by microbiota (bacteria, fungi, algae, microinvertebrates) because they grow and multiply rapidly. Macrophytes on the other hand, obtain and use phosphorus much more slowly. Some is obtained by roots located at the surface of the soil, but more is obtained from roots located below the soil surface which require phosphorus to move downward in the soil. Uptake response times are in the order of weeks (Richardson & Marshall, 1986).

All wetland biota undergo a cycle of growth, death and decomposition, as shown in Figure 3.5, and hence form an integral part of the phosphorus cycle within wetlands. Phosphorus is re-released into the system upon death and decay of biota. Up to 50 percent of the TP from macrophytes can be leached from leaves and root systems within hours (Wetzel, 1983).

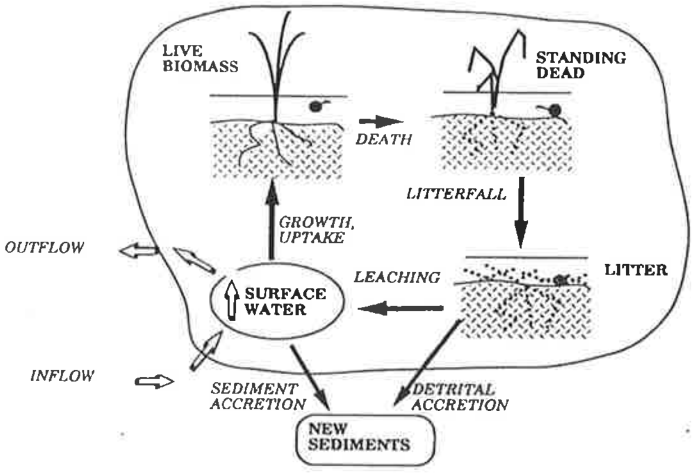


Figure 3.5 Phosphorus cycling in wetlands involving biota.

(Source: Kadlec & Knight, 1996)

3.4 The Removal of Metals

Wetlands interact strongly with metals in a number of ways and thus have significant removal capabilities. Three major mechanisms are operative (Kadlec & Knight, 1996):

1. Binding to soils, sediments, particulates and soluble organics;
2. Precipitation 沉淀 as insoluble salts, principally sulphides and oxyhydroxides; and
3. Uptake by plants, including algae, and by bacteria.

Uptake by plants and algae is for the purpose of growth enhancement or, at higher metal concentrations, for protective 保护的 purposes. The metals reach plants via their fine root structure and most is intercepted there with only small amounts finding their way to the stems, leaves and rhizomes (Zhang et *al*., 1990). When the roots die, some fraction of the metal content may be buried permanently, but there are little data on the re-release of metals during root decomposition. Algae react to metals in different ways. Some species will sequester a particular metal but not others, while some trace metals are toxic, or inhibitory to some algae and invertebrates. Cadmium, for example, is incorporated into the tissues of wetland plants. Uptake is strongly preferential to the roots and rhizomes where concentrations can reach 100 to 700 mg/kg in *Typha* spp.

Wetland soils bind metals by cation exchange and chelation, and humic substances (in solution and as solids) can form bonds with metals. It is this storage in the wetland sediments that is the only sustainable sink for metal uptake.

With most of the metals found in stormwater, the removal efficiency increases with increasing inflow concentrations (Kadlec & Knight, 1996).

3.5 Other Wetland Processes

In addition to the pollutants for which removal pathways have already been discussed, stormwater contains a number of other potentially harmful substances (see Section 2.1) which can also be successfully treated.

Removal of pathogens can be achieved by wetlands simply due to the environmental conditions. Factors such as natural die off, temperature, ultraviolet light, unfavourable water chemistry, predation and sedimentation can all contribute to the reduction in pathogen populations.

Hydrocarbons are a product of urban environments and in receiving waters can be toxic to plants and pose limitations on aerobic and anaerobic degradation. The major routes for the removal of hydrocarbons in wetlands are

1. Volitilisation;
2. Photochemical oxidation;
3. Sedimentation;
4. Sorption; and
5. Biological (microbial) degradation.

In addition, three types of microbial processes can contribute: fermentation, aerobic, and anaerobic respiration (Kadlec & Knight, 1996).

4

Site Description

The Barker Inlet Wetlands are located around 20 kilometres to the North of the Adelaide metropolitan area and were constructed on degraded low-lying coastal land adjacent the Barker Inlet North Arm.

The Barker Inlet Wetland is the largest in a series of constructed wetlands which form a "green band" designed to intercept and treat Adelaide's stormwater runoff. Together with the Range, Magazine Creek and Connector Wetlands, the Wetlands cover an area of approximately 340 hectares and treat almost 40 percent of Adelaide's stormwater runoff. The Barker Inlet Wetlands alone occupy an area of 172 hectares and intercept runoff from approximately 26 percent of the Adelaide metropolitan catchment area (MFP DC, 1996).

Planning for the Barker Inlet Wetland began in 1994, and contracts for the earthworks, structures and pipework were awarded in December 1995 (MFP Australia, 1995). Construction of the ponds began in January 1996 which involved the excavation of 500 000 cubic metres of earth and the importation of 200 000 cubic metres of topsoil. Planting within the wetland began in mid-1996 and was completed in early 1997.

4.1 The Catchment

Stormwater flowing into the Barker Inlet Wetland originates from a large catchment covering a significant portion of Adelaide's northern suburbs as seen in Figure 4.1. The catchment as a whole is predominantly urban with undeveloped pockets of land facing development in the near future. Photograph 4.1 taken in March 1999 shows a large pocket of vacant land in Regency Gardens (Oakden) already earmarked for residential development. A number of heavy industrial estates are located at the bottom end of the catchment.

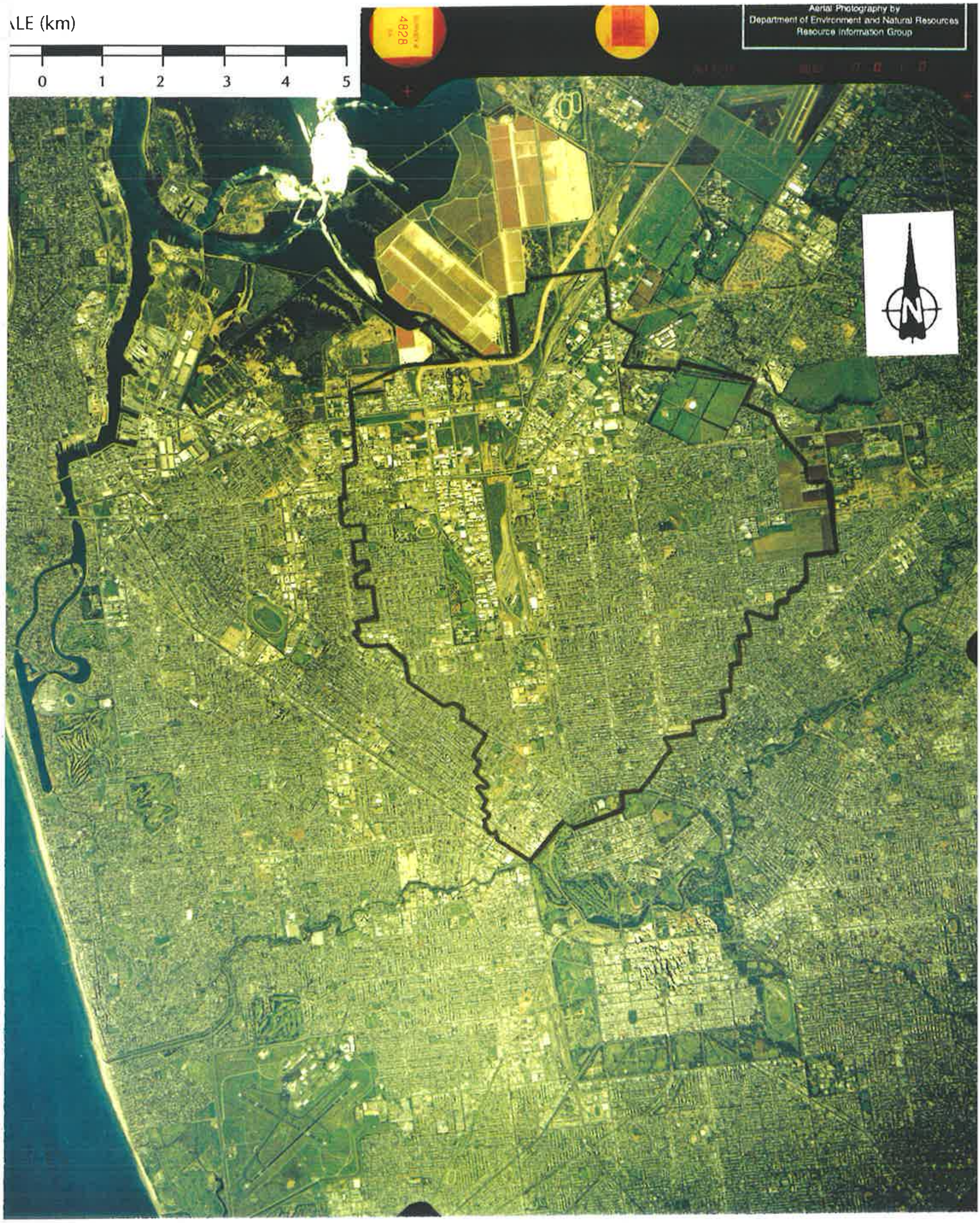


Figure 4.1 Aerial photograph of Adelaide showing location of the Barker Inlet Wetland catchment

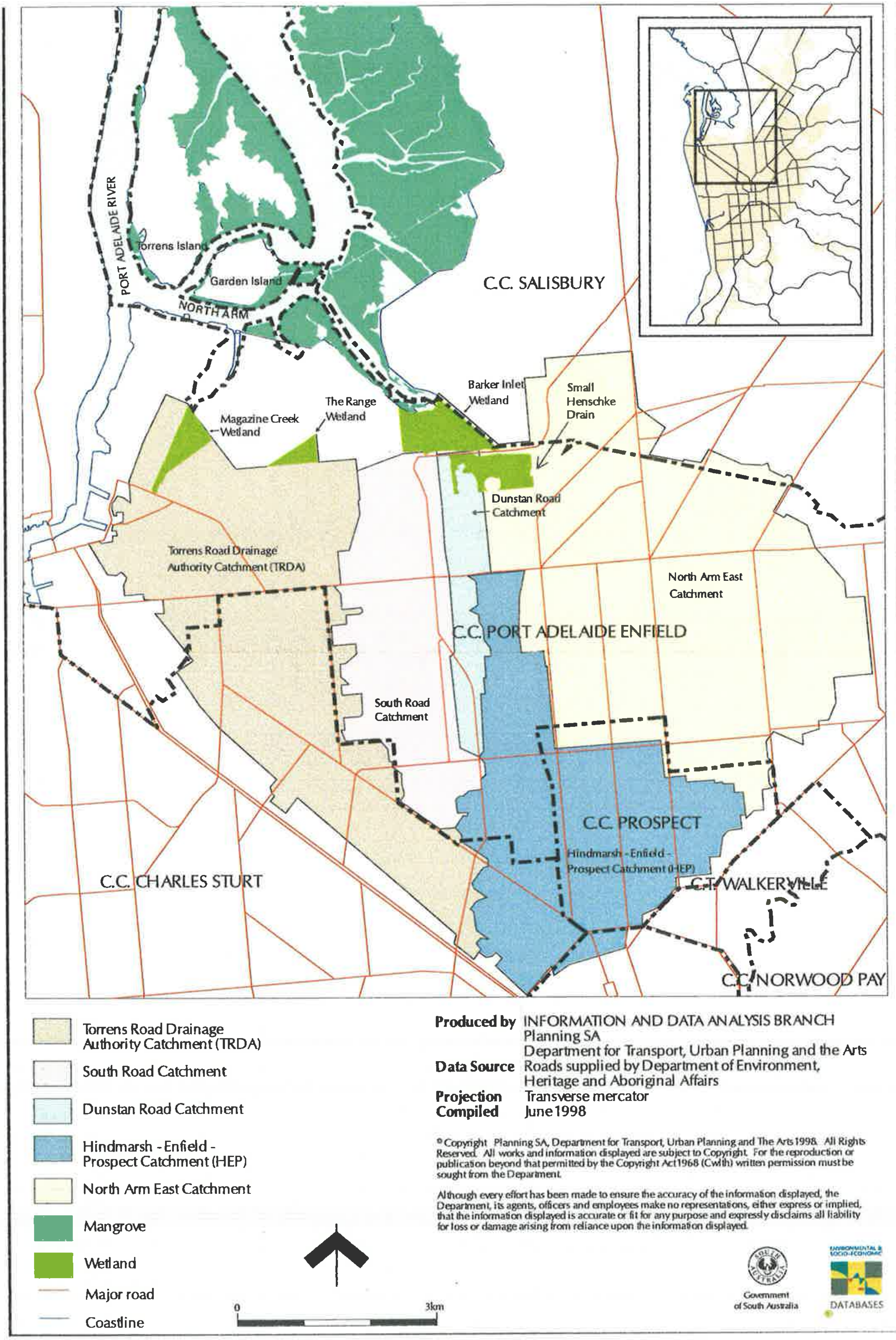
Photograph 4.1 **Large pocket of vacant land previously owned by the Department of Agriculture earmarked for residential development - corner Fosters Rd and Sir Ross Smith Boulevard, Regency Gardens (Oakden). Developments such as this will increase the yield of the North Arm East catchment.**

The catchment slopes gently from the Adelaide foothills in the east down to the western side, and from the Adelaide CBD down to the wetlands. The total catchment area is approximately 4475 hectares, which can be split into five sub-catchments each draining into different parts of the pond system (see Section 4.2). The location of each of these sub-catchments relative to the entire catchment is shown in Figure 4.2, while Table 4.1 shows the names and areas for each of the sub-catchments draining into the pond system.

Table 4.1 **Summary of catchment names and areas**

Catchment	Area (hecatres)
North Arm East Catchment	2170
Hindmarsh Enfield Prospect Catchment	1300
Dunstan Road Catchment	223
South Road North Arm West Catchment	782
Henschke Street Catchment	220

Figure 4.2 Location of Barker Inlet Wetlands and catchment, showing sub-catchments.



Produced by INFORMATION AND DATA ANALYSIS BRANCH
Planning SA
Department for Transport, Urban Planning and the Arts
Data Source Roads supplied by Department of Environment, Heritage and Aboriginal Affairs
Projection Transverse mercator
Compiled June 1998

Historically, the average annual rainfall for the Barker Inlet Wetland catchment ranges from 440 millimatres on the western boundary to 500 millimetres on the eastern boundary, with an average across the catchment of 470 millimetres (Williams, 1997). The higher rainfall to the east is caused by the Adelaide foothills, which yield a higher rainfall than the plains to the west.

Rainfall runoff response times vary between the various sub-catchments due to differences in the pipe networks and channel systems. The North Arm East and Dunstan Road drains have a rapid response, typically in the order of 30-40 minutes, due to their highly efficient concrete stormwater drains. The Hindmarsh Enfield Prospect Drain, on the other hand, has a much slower response due to the presence of a large grass swale drain at the bottom end of the catchment, which spreads out the flow and provides for some infiltration in the early part of an event.

This project was specifically concerned with just one of the ponds within the system, refered to as Pond 4, and therefore only those catchments draining into Pond 4 will be discussed further. The two catchments of concern are the North Arm East drain, the largest of the sub-catchments, and the Henschke Street catchment, the smallest of the sub-catchments.

4.1.1 North Arm East Catchment

The North Arm East Catchment, as mentioned above, is the largest of the sub-catchments with an area of approximately 2170 hectares (22 km^2). Of this, around 26 percent, or 560 hectares, is impervious. The catchment is roughly bounded on three sides by Churchill Road to the West, Regency Road to the South, and Fosters Road (Northfield) to the East, and covers the suburbs of Blair Athol, Broadview, Clearview, Enfield, Gepps Cross, Greenacres, Northfield, Sefton Park, and parts of Dry Creek and Pooraka.

The catchment is almost entirely urbanised and predominantly of residential landuse. A summary of the landuse types within the catchment are given in Table 4.2. The landuses are also shown in Figure 4.3 for the entire Barker Inlet Wetland catchment.

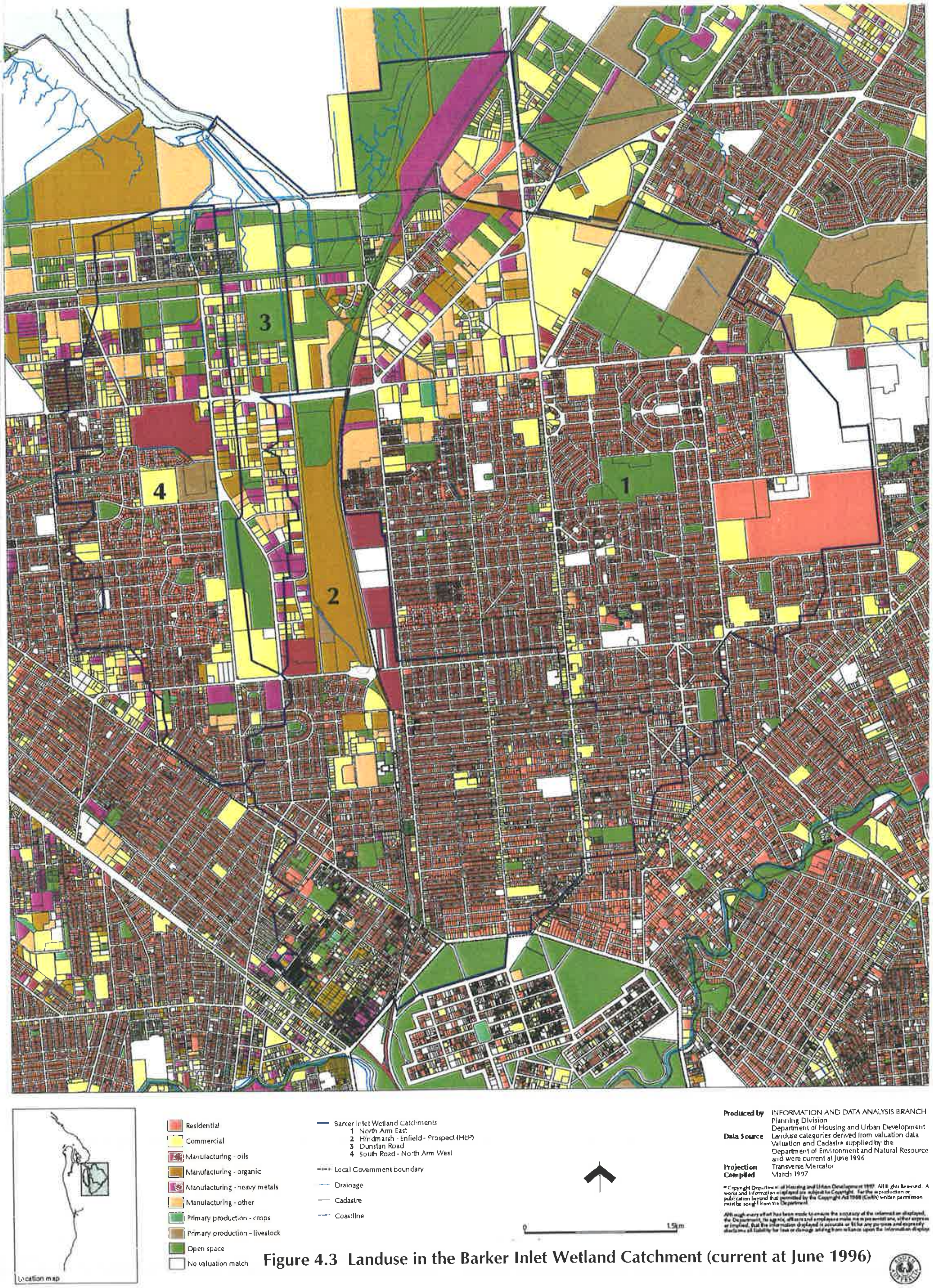
The landuse information given was current in June 1996 and was derived from Valuation and Cadastre supplied by the Department of Environment and Natural Resources (Williams, 1997). As the information suggests, the predominant landuse within the North Arm East catchment is for residential purposes (40 percent). The significance of this was demonstrated by Williams (1997) who compared the residential North Arm East catchment with the predominantly industrial Dunstan Road catchment. The Dunstan Road catchment produced stormwater runoff with significantly higher concentrations of sediment, heavy metals, and hydrocarbons.

Table 4.2 **Land use in the North Arm East Catchment**

Landuse	Percent of Catchment
Residential	40
Commercial	13.4
Manufacturing - oils	1.2
Manufacturing – organics	2.8
Manufacturing – heavy metals	5
Manufacturing – other	5.7
Primary production – crops	0.2
Primary production – livestock	2.4
Open space	18.7
No valuation match	9.9
Total	100

Photograph 4.2 and Photograph 4.3 are typical residential streets in the North Arm East catchment. Photograph 4.2 in particular, shows a very characteristic "leafy" street. These leafy suburbs contribute to the high organic load experienced at the bottom end of the catchment which are either trapped in trash racks or removed in the pond system.

Photograph 4.2 **Howard Street, Nailsworth – A typical "leafy" residential street in the North Arm East catchment.**



Produced by INFORMATION AND DATA ANALYSIS BRANCH
Planning Division
Department of Housing and Urban Development
Data Source Landuse categories derived from valuation data
Valuation and Cadastre supplied by the
Department of Environment and Natural Resource
and were current at June 1996
Projection Transverse Mercator
Compiled March 1997

Figure 4.3 Landuse in the Barker Inlet Wetland Catchment (current at June 1996)

Photograph 4.3 McKay Street, Broadview – typical residential street in the North Arm East catchment.

The stormwater drainage for the North Arm East catchment is a classically engineered system consisting of a complex network of underground concrete pipes branching out to the limits of the catchment. At the bottom end of the catchment the pipes feed into an open, concrete lined trapezoidal channel, which will be discussed further in section 4.3.1. The rainfall runoff response of the catchment is rapid and quantitively high. There is also a significant amount of baseflow through the concrete drain, thought to be supplied by industrial process and equipment cleaning activities.

4.1.2 Henschke Street Catchment

The Henschke Street catchment is a much smaller catchment to the North Arm East catchment. It is 220 hectares in area and lies to the north of the Barker Inlet Wetland site. The catchment is shown on Figure 4.2 as part of the North Arm East catchment jutting out to the north. It covers part of the suburbs of Cavan and Dry Creek.

As Figure 4.3 shows, the Henschke Street catchment is primarily an industrial/commercial catchment with a significant portion of open space.

4.2 The Pond System

As has been mentioned previously, the Barker Inlet Wetlands were constructed in 1996, with planting aquatic vegetation occurring in mid-1996. Photograph 4.4 show the ponds during construction and give an indication of the side slopes and water depths within the ponds.

Photograph 4.4 Barker Inlet Wetlands during construction phase

The ponds were designed to incorporate both deep and shallow water zones with cells varying in water type between fresh, saline and mixed. The pond system consists of a series of southern ponds and a major northern pond and marine inter-tidal area. Ponds on the Southern side of Salisbury Highway/South Road Connector typically have depths in the 0.6 - 2 metre range, while on the northern side they were designed to be much shallower, around 0.3 metres. Figure 4.4 gives a diagram of the entire pond system, and gives the nomenclature for each of the ponds. The surface areas and volumes of the ponds are given in Table 4.3.

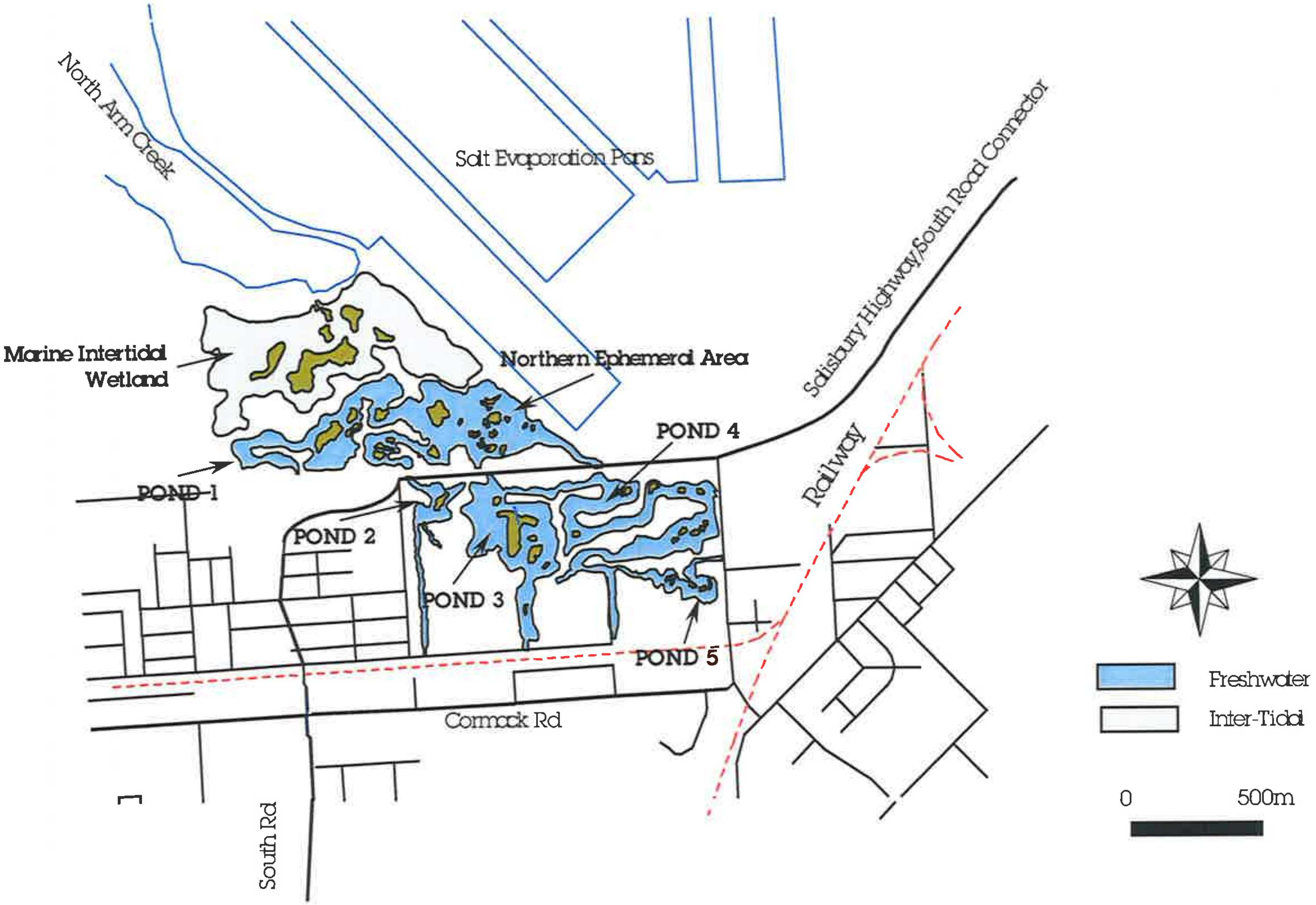


Figure 4.4 Diagram of Barker Inlet Pond system showing nomenclature of ponds

Table 4.3 **Surface areas and volumes (at operating level) for each of the ponds in the Barker Inlet Wetland System**

Pond	Surface Area (ha)	Estimated Volume (ML)
Pond 1	8.6	64
Pond 2	2.9	22
Pond 3	9.0	67
Pond 4	10.5	69
Pond 5	2	15
Northern Ephemeral Area	15.0	45
Marine Inter-tidal Wetland	28.3	-

The southern and northern ponds are discussed in further detail below.

4.2.1 The Southern Pond System

The five southern ponds (Ponds 1 through 5) shown on Figure 4.4 are fed directly from the major catchments listed in Table 4.1 via each of the inflow drains. Table 4.4 lists the catchments contributing to each of the southern ponds.

Table 4.4 **Catchment contributing to each of the southern ponds in the Barker Inlet Wetland System**

Pond	Contributing Catchment
Pond 1	South Road North Arm West Catchment
Pond 2	Dunstan Road Catchment
Pond 3	Hindmarsh Enfield Prospect Catchment
Pond 4	North Arm East Catchment
Pond 5	Henschke Street Catchment

Ponds 1 through 4 flow in parallel, with overflow from Pond 3 flowing into the tailwater pond of Pond 4. Pond 5 adjoins Pond 4 downstream of the North Arm East Drain inflow.

The ponds are in a relatively immature stage of development since the vegetation is only a couple of seasons old. Photograph 4.5 was taken shortly after construction was completed in 1996, while Photograph 4.6 is a similar shot taken in October 1998. This gives an indication of the extent of vegetation establishment that has occurred in just two years.

Photograph 4.7 and Photograph 4.8 show a similar comparison. It should be noted that the most recent photographs were taken in March 1999 (Autumn) while the reeds were in their annual senescence period which is why they appear brown.

Photograph 4.5 Barker Inlet Wetland Southern Pond system shortly after completion. Pond 4 to left and Pond 3 to right, adjoining tailwater pond in foreground (photograph late 1996)

Photograph 4.6 Taken from same location as Photograph 4.5 showing extent of reed growth as of March 1999.

Photograph 4.7 Barker Inlet Wetland Pond 3 taken from Salisbury Highway/South Road Connector looking towards Pond 2. Photograph taken late 1996.

Photograph 4.8 Taken from same location as Photograph 4.7 but in March 1999. Shows extent of vegetation growth.

Salinity of the southern ponds varies significantly, with Pond 4 being the freshest, and Pond 2 (fed by the Dunstan Road Catchment) being the most saline. The salinity of each of the ponds has affected the establishment of aquatic vegetation, which was planted on a trial basis in each of the cells. Photograph 4.9 of Pond 2, was taken in March 1999 and shows how little vegetation has established in this pond when compared to Photograph 4.6 or Photograph 4.8. Salt on the ground surface around the pond is also visible indicating the high salinity of the surface and/or ground water.

Photograph 4.9 Barker Inlet Wetland Pond 2. Little to no vegetation has established due to the visibly high salinity of the pond.

Pond 4, the largest of the southern ponds, has been the focus of this project. Details of this pond specifically are given below.

4.2.1.1 Pond 4

Pond 4 was chosen to be the focus of this project as it is the largest of the southern ponds and could be monitored relatively easily upon construction of an outflow monitoring station.

Figure 4.5 shows the layout of ponds 4 and 5 (which flows into Pond 4) and the contour depths of the ponds. The operating water level is 0.6 m AHD, at this depth Pond 4 has a surface area of 9.5 hectares. The average depth of the pond is 0.73 metres, with a maximum depth (at operating level) of two metres. The volume of the pond at operating level is 69 megalitres.

The aquatic plants planted on site are also shown on Figure 4.5, a list of the species planted is given below:

- *Bolboschoenus caldwelli*
- *Bolboschoenus medianus*
- *Cyperus gymnacaulis*
- *Cyperus vaginatus*
- *Eliocharis acuta*
- *Phragmites australis*
- *Schoenoplectus pugens*
- *Schoenoplectus validus*
- *Typha spp. (naturally occurring)*

Photograph 4.10 and Photograph 4.11 show the healthy stands of aquatic reeds that have established in Pond 4 since planting in mid-1996 . These photographs were taken in October 1998 near the outlet of Pond 4 facing South.

4.2.2 The Northern Pond System

The northern pond system consists of the Northern Ephemeral Area and the marine inter-tidal wetland. The northern ephemeral area is so named as it is shallow in depth and dries out during dry periods (typically summer). The marine inter-tidal area is separated from the NEA by a bund wall which lets stormwater out through a series of four bund gates, but does not allow salt water back in during high tides. The northern ephemeral area is characterised by high salinity water, originally thought to be caused by shallow saline ground water. In mid 1998, however, it was discovered that sea water was leaking back into the pond during high tides through the bund gates which were not sealing properly.

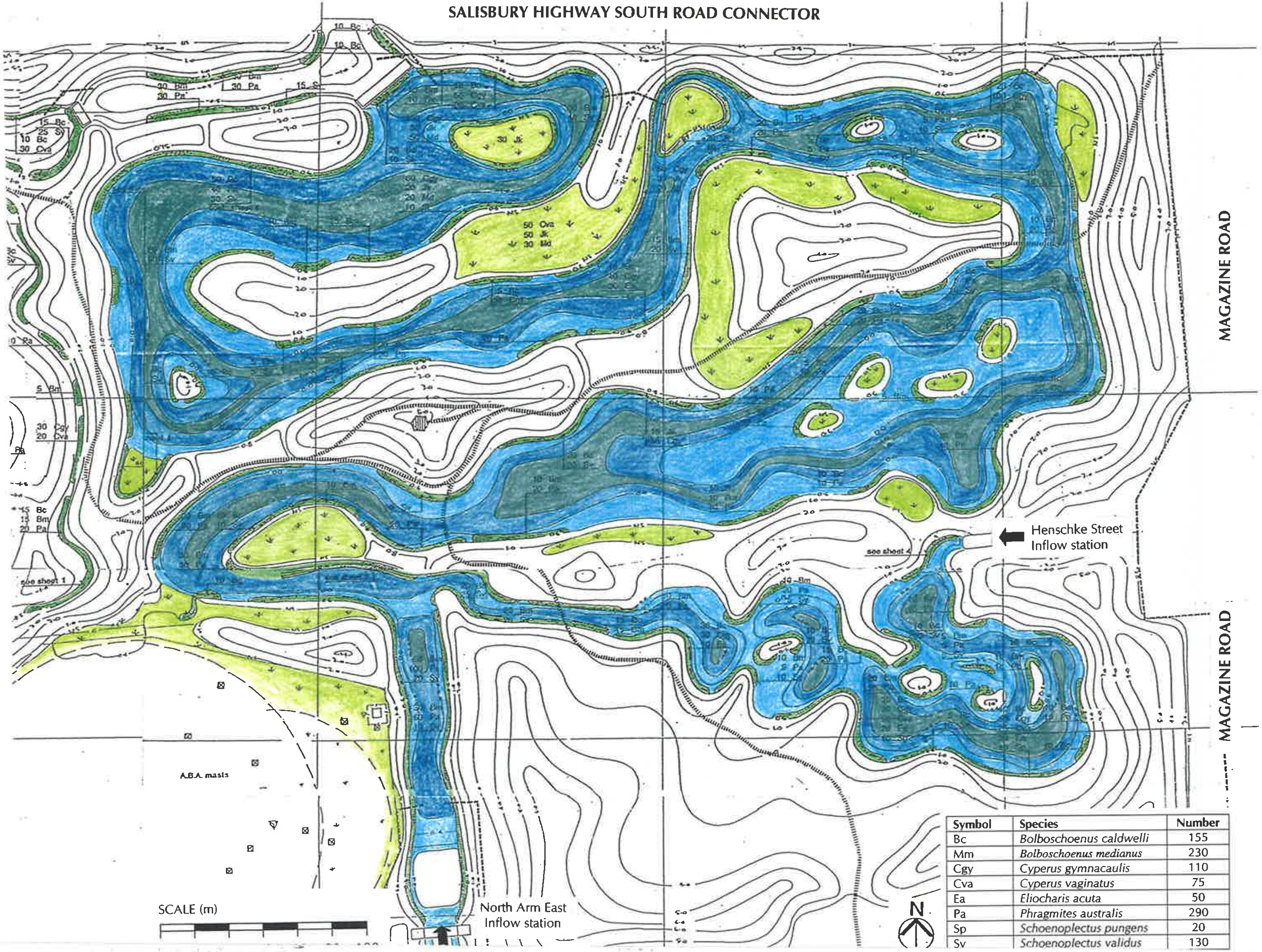
The shallow exposed nature of the NEA renders it susceptible to re-suspension of deposited sediments. An aquatic planting plan consisting of salt tolerant species is now in place to try and minimise the effects of strong winds on re-suspension.

Photograph 4.12 taken in September 1998 shows the Northern Ephemeral pond taken from the Bund facing South. The Bund monitoring station can be seen in centre of the photo and the beginnings of establishing vegetation are visible in the foreground.

The marine inter-tidal wetland is the interface between the relatively fresh Northern Ephemeral wetland and the Barker Inlet North Arm. This area is the home to a number of species of sea birds and wading birds, which feed in the shallow water. The Marine Inter-tidal wetland is pictured in Photograph 4.13.

Photograph 4.10 Barker Inlet Wetland Pond 4 taken near outlet facing South (October 1998). Shows healthy stands of aquatic reeds which have established in the Pond.

Photograph 4.11 Barker Inlet Wetland Pond 4 taken from outlet facing South (October 1998)



Symbol	Species	Number
Bc	*Bolboschoenus caldwelli*	155
Mm	*Bolboschoenus medianus*	230
Cgy	*Cyperus gymnacaulis*	110
Cva	*Cyperus vaginatus*	75
Ea	*Eliocharis acuta*	50
Pa	*Phragmites australis*	290
Sp	*Schoenoplectus pungens*	20
Sv	*Schoenoplectus validus*	130

Photograph 4.12 Barker Inlet Wetland Northern Ephemeral Area. Photograph taken from Bund facing South (October 1998).

Photograph 4.13 Marine inter-tidal wetland, Barker Inlet Wetlands

4.3 Monitoring Sites

In order to asses the performance of Pond 4 it was necessary to establish a number of monitoring sites to gauge all inflows and outflows of the pond. These monitoring stations were placed in the primary inflow drain, the North Arm East Drain, the secondary inflow drain, the Henschke Street Drain, and at the pond outlet just prior to the water flowing under the Salisbury Highway/South Road Connector. The locations of these stations are shown on Figure 4.6.

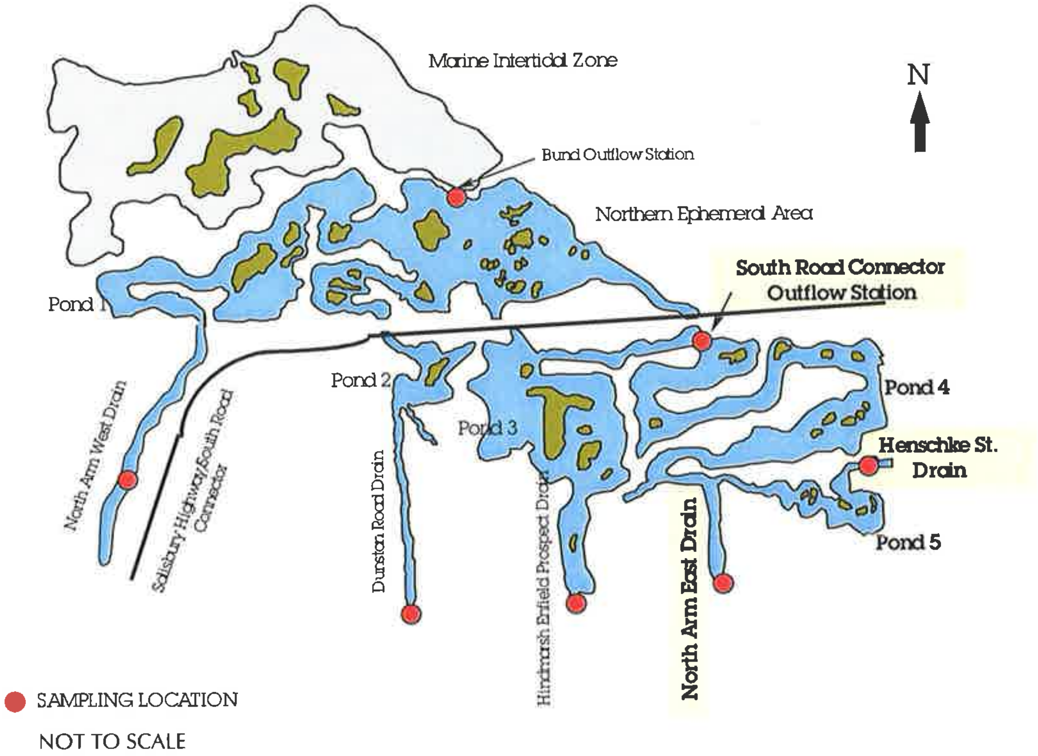


Figure 4.6 **Sampling locations within the Barker Inlet Wetland system**

4.3.1 North Arm East Drain

The North Arm East monitoring station was constructed in 1994. It involved the construction of a concrete broad crested weir across the channel and the placement of an instrumentation cage in the centre of the drain immediately downstream of the weir. Photograph 4.14 shows the construction of the North Arm East weir, while Photograph 4.15 shows the completed monitoring station. Details of the instrumentation are given in Chapter 5.

Photograph 4.14 The North Arm East monitoring station during construction

Photograph 4.15 The completed North Arm East monitoring station

The North Arm East drain has dimensions shown in Figure 4.7. The channel has a slope of 0.09 % south of the railway bridge, which is approximately 50 metres downstream of the weir. North of the bridge the channel slope is 0.07 %. Further downstream of the bridge trashracks were installed in January 1996. The installation of the trashracks significantly altered the rating of the broad crested weir in the channel, the effects are discussed in Section 5.2.1.

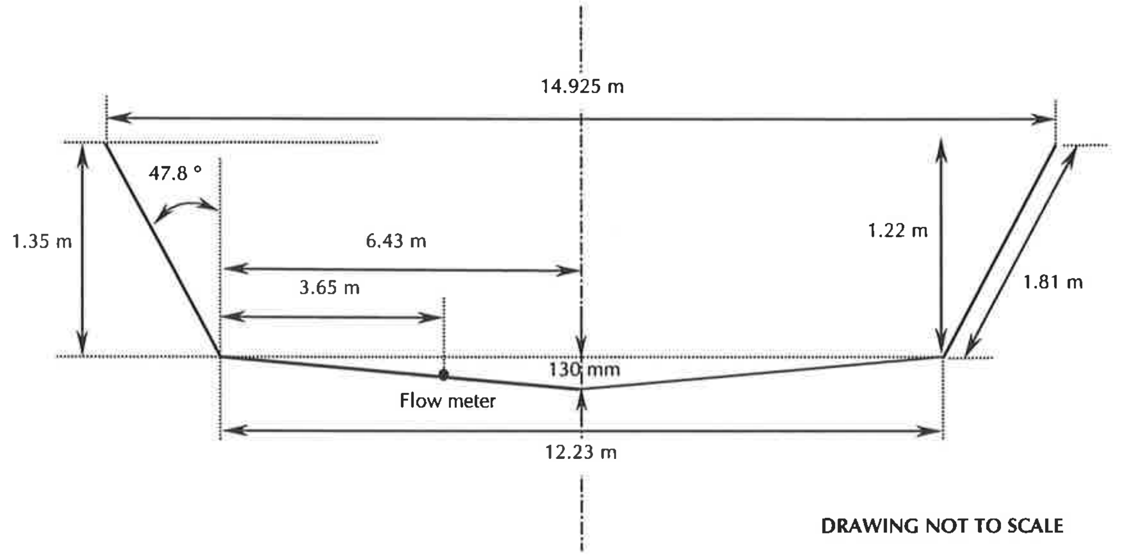


Figure 4.7 Dimensions of the North Arm East drain

4.3.2 Henschke Street Drain

The monitoring station in the Henschke Street Drain was constructed in early 1997 to complete the monitoring of the inflows to Pond 4. The drain is concrete lined and trapezoidal in cross section upstream of the culverts which direct the flow underneath Magazine Road. Downstream of the culverts the drain is unlined and heavily reeded, predominantly with *Typha spp.*

Construction of the Henschke Street Station took several months as the reeds had to be cleared in the section to be concreted, and a significant amount of earth moving was required. A concrete slab and vertical concrete side-walls were erected to provide a stable base for the cage to house the monitoring equipment as well as the pressure transducers and flow meter. A triangular profile flat v-weir, or Crump-weir, was constructed to act as a flow measurement device. The Crump-weir was constructed with a 1-to-2 slope on the upstream face, and a 1-to-5 slope on the downstream face. The cross slope was 1-in-20.

Dimensions of the Henschke Street station are given in Figure 4.8.

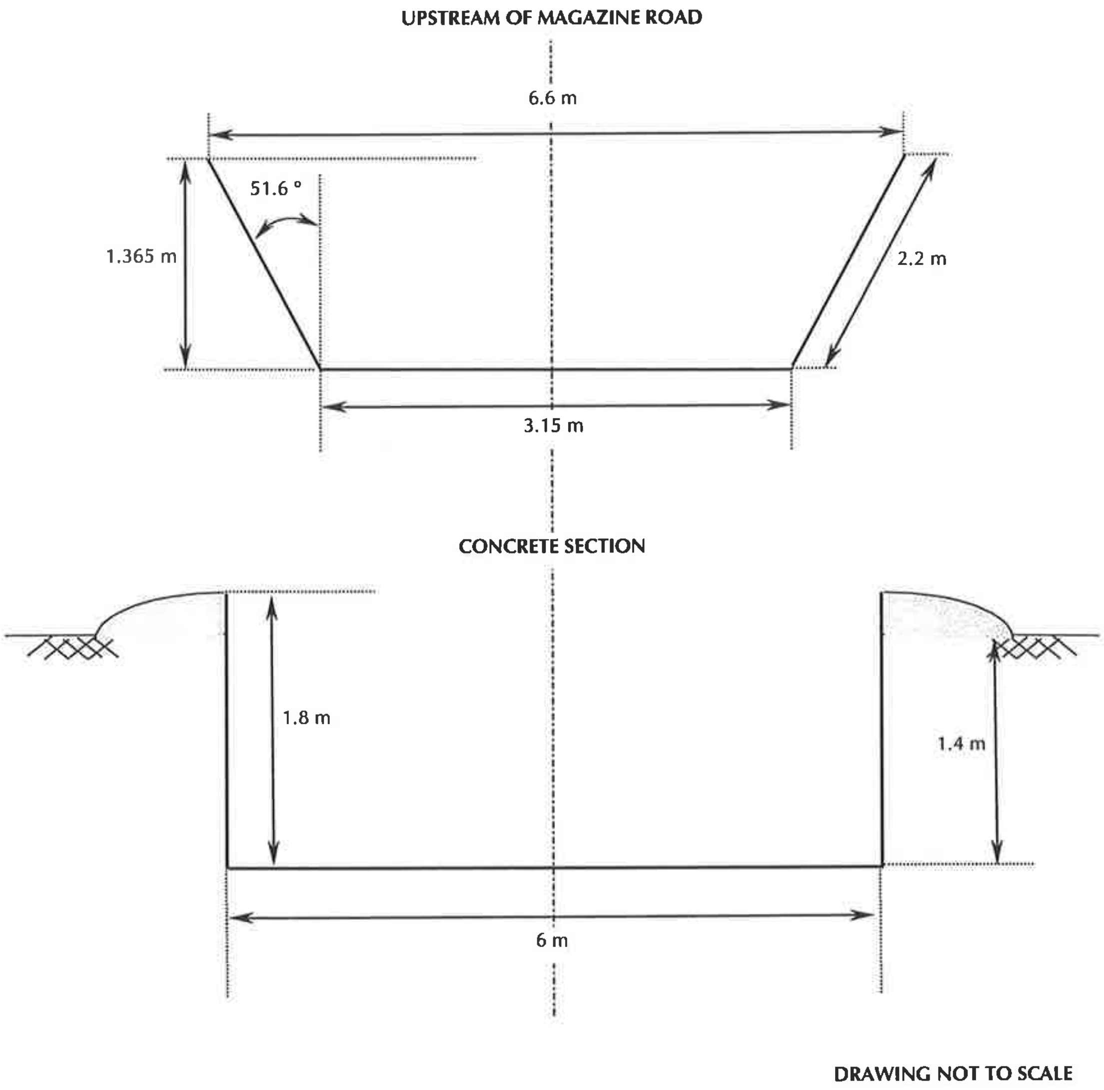


Figure 4.8 **Dimensions of the Henschke Street Drain; upstream of Magazine Road (top) and downstream of Magazine road (bottom).**

The Henschke Street monitoring station is shown during construction in Photograph 4.16. A steel cabinet was placed on the bank of the channel to house the data logger, batteries and automatic water sampler. The completed weir is shown in Photograph 4.17, although at this stage the instrumentation cage had not been placed in the correct position in the centre of the drain.

Photograph 4.16 Henschke Street monitoring station during construction.

Photograph 4.17 The completed Henschke Street weir (drain is dewatered). The steel mesh cage is not shown in the correct position.

Monitoring at this station was unsuccessful due to site constraints and malfunctioning equipment. The Henschke street catchment however, is only 10 percent of the North Arm East catchment by area and thus the contribution of pollutant load to Pond 4 was small (~10 %).

4.3.3 South Road Connector Outflow Station

The South Road Connector Outflow station was also constructed in 1997 to complete the monitoring of Pond 4. The monitoring station was placed at the outlet of Pond 4, before the water passes under the Salisbury Highway/South Road Connector and into the Northern Ephemeral Area.

An option of constructing a large concrete weir across the outflow section was considered as a method of flow measurement but rejected due to cost constraints. Instead, flow velocity metres were placed in the culverts under Salisbury Highway/South Road Connector through which the flow passes (see Section 5.2.2 for more detail). The pond level was maintained by a rock filled wire mesh gabion sealed with a geofabric membrane, and a sharp crested weir which controlled the pond level. The gabion spanned 32 metres across and was divided in the middle by the 2 metre wide sharp crested weir.

An instrumentation cage was placed immediately behind the sharp crested weir to monitor water quality as it flowed out of the pond. Instrumentation cables were housed in steel and PVC conduits connecting them to the logger and sampler cabinet on the bank of the pond.

5

The Monitoring Network

Monitoring of the entire Barker Inlet Wetland system was undertaken with a network consisting of seven streamflow monitoring stations, five additional flow monitoring stations, and eight rainfall stations. Over the two year duration of this project these stations were maintained with the help of other team members. Below is a brief discussion of the entire monitoring network, however only those stations directly related to this project will be discussed in further detail. The inflow and outflow monitoring sites are as follows:

- North Arm East Inflow Drain
- Hindmarsh Enfield Prospect Inflow Drain
- Dunstan Road Inflow Drain
- South Road North Arm West Inflow Drain
- Henschke Street Inflow Drain
- South Road Connector Outflow Station
- Bund Outflow Station

Streamflow monitoring stations were positioned in each of the five inflow drains and the two outflow locations. These stations consisted of continuous water quality monitoring probes as well as pressure transducers and velocity meters for flow measurement. In addition, each station was fitted with a GAMET automatic water sampler to collect event based water samples. All monitoring equipment was linked to a data logger and controller installed at each site to record the information for periodic downloading to a lap-top computer. The complicated nature of flow monitoring at the outflow locations required the installation of separate flow-velocity meters in three locations at South Road Connector and two locations at the Bund. These were downloaded separately and later combined with streamflow data (see Section 5.5)

The location of each of the monitoring sites were shown on Figure 4.6. The sites referred to hereafter are highlighted on the diagram and are printed in **bold-type**. Rainfall gauges were installed in each of the catchments; these are discussed in more detail in Section 5.4.

5.1 Streamflow Quality Monitoring

Five water quality parameters, along with depth and velocity, were at some stage monitored at the North Arm East and South Road Connector sites. The North Arm East station was constructed in 1994 and was the main input to the system as mentioned in the previous chapter. The equipment utilised at this site was slightly different to those at Henschke Street and South Road Connector, which were constructed in 1997 (completed in June). Despite this, the stations contained similar equipment and functioned in the same way. The water quality and flow monitoring probes used on site are summarised in Table 5.1.

Table 5.1 Details of instrumentation

Probe	Brand(s)/Model(s)	Scale/Units
Turbidity	Tain Greenspan TS100	0 – 2000 TTU 0 – 2000 NTU
Electrical Conductivity	Greenspan EC200	0 – 2000 µS/cm
pH	Greenspan pH100	0 – 14
Temperature	Combined with EC probe	0 – 50 °C
Dissolved Oxygen	Greenspan DO100	0 – 20 ppm
Stage Height	Greenspan PS200	0 – 2.5 m
Water Velocity	Mace HVQ Unidata Starflow 6526B	(-4) – 4 m/s 21 – 4500 mm/s

The instruments were housed in a vandal-proof cage mounted in the centre of each channel (or outlet weir). The cage at North Arm East was constructed differently from the others as it experienced a large debris load with each storm and therefore needed to shield the instruments to prevent clogging. The North Arm East cage, shown in Photograph 5.1, was three metres long, 500 millimetres wide and 1.2 metres long. It had a sloped debris guard on the leading edge and galvanised steel mesh sides. Inside the cage was a float and counterweight apparatus that allowed the boat, to which the probes were attached, to rise with rising stage height.

The cages at South Road Connector and Henschke Street, shown in Photograph 5.2, were slightly less resistant to debris as it was not such a problem at these locations. The cage was a simplified version of the North Arm East cage but was of the same construction, being made from galvanised steel mesh to guard against vandalism. The boat inside was also simplified and had no float counterweight arrangement, but rather relied on a balanced wooden boat constrained by wire cables to keep the probes in position.

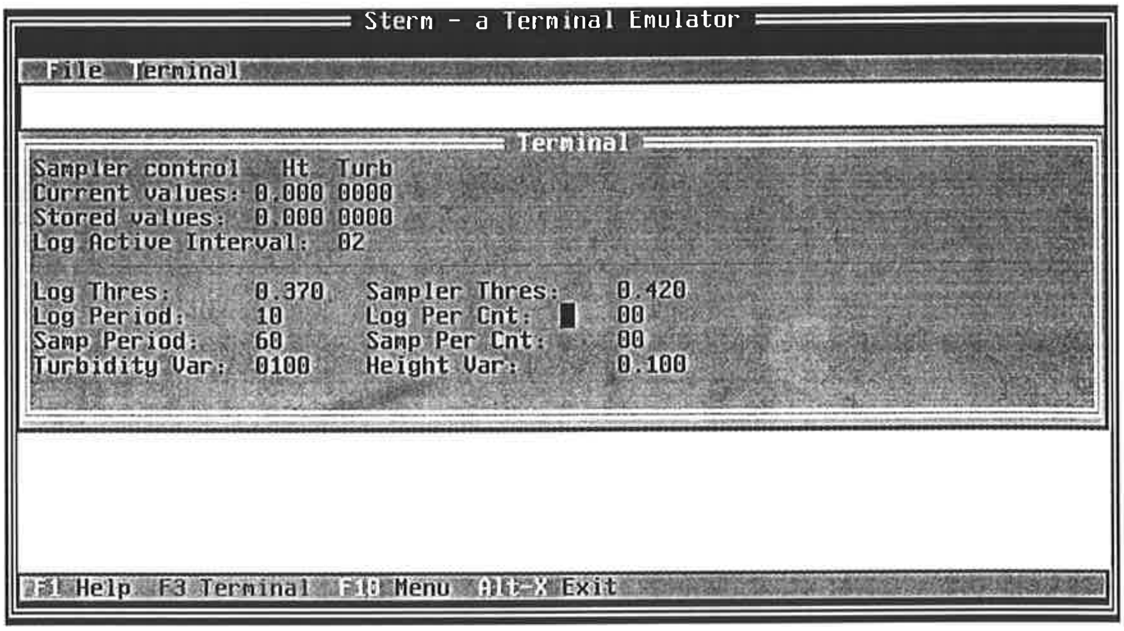
Conduits were installed during the construction of the weirs at North Arm East and Henschke Street to carry the cables from the probes in the channel to a cabinet on the bank of the channel. The steel instrumentation cabinet housed the data logger, controller, batteries, and automatic water sampler as shown in Photograph 5.3.

Photograph 5.1 Instrumentation cage at North Arm East containing water quality probes.

Photograph 5.2 Instrumentation cage at the Bund containing water quality probes. This is the same as the arrangement at the South Road Connector station.

Photograph 5.3 Instrumentation cabinet at North Arm East containing data logger and controller, battery power supply, and automatic water sampler on the right.

A UNIDATA™ Macro 7000B electronic logger was housed in the cabinet at each of the stations that recorded information from the probes at pre-determined time intervals. The logger was connected to an electronic controller designed in the Department of Civil and Environmental Engineering at the University of Adelaide which determined when instrument values were logged as well as when samples were taken by the automatic water sampler. The frequency of logging or the threshold of sampling could be controlled by a number of factors and was able to be adjusted by the user via the computer program STERM, also developed in the Department of Civil and Environmental Engineering. Figure 5.1 shows a typical STERM screen with the parameters defined for a particular station.



```
Sampler control   Ht    Turb
Current values: 0.000 0000
Stored values:  0.000 0000
Log Active Interval:  02

Log Thres:      0.370   Sampler Thres:    0.420
Log Period:     10      Log Per Cnt:      00
Samp Period:    60      Samp Per Cnt:     00
Turbidity Var:  0100    Height Var:       0.100
```

Figure 5.1 Screen shot of a STERM parameter screen

Table 5.2 gives a brief definition of each of the terms used by STERM.

Table 5.2 Definition of terms used by STERM

Parameter	Definition
Log Active Interval	The frequency (minutes) at which the logger will record values when the stage height (m) is greater than the 'log threshold' (i.e. storm sampling)
Log Threshold	The stage height (m) which, when exceeded, causes the logger begin recording at intervals defined by the 'log active interval'.
Sample Threshold	The stage height (m) at which the automatic water sampler triggers.
Log Period	The frequency (minutes) at which the logger will record values when the stage height is less than the 'log threshold' (i.e. baseflow sampling)
Sample Period	The maximum permissible time interval between water samples without activation by the 'height variance' or 'turbidity variance'.
Turbidity Variance	The turbidity increment which, when exceeded after initial activation by 'sample threshold', will induce another sample.
Height Variance	The stage height increment which, when exceeded after initial activation by 'sample threshold', will induce another sample.
Height Differential	Used at SRC and HENS only. The differential between the upstream and downstream stage heights which, if greater than the current values, stops sampling.

Put simply, in the absence of significant flow, the logger recorded values at intervals defined by the 'log period', normally set to ten minutes. Once flow began and the stage height reached the 'log threshold' the logger began to record at the 'log active interval', usually set to two minutes. If the stage height then reached the 'sample threshold' and remained there for two consecutive log periods (two minutes) a water sample was taken. While the height remained above the 'log active interval' the controller continued to check for when the next sample should be taken. If there was a rise in stage height greater than the 'height variance' within four minutes (two log active intervals) a sample was taken, similarly if there was a rise in turbidity greater than the 'turbidity variance' within four minutes a sample was taken. If neither of these conditions were satisfied while the logger was still active and the time between samples defined by the 'sample period' (normally 30 or 60 minutes) elapsed, a sample was taken. In the absence of the influence of the controller, the UNIDATA™ logger was programmed to record readings each hour, on the hour.

Figure 5.2 shows a schematic diagram of the instrumentation at the three stations. The logger and controller were powered by an external 12 V lead acid battery that was replaced periodically as required. A smaller 12 V rechargeable battery powered the automatic water sampler.

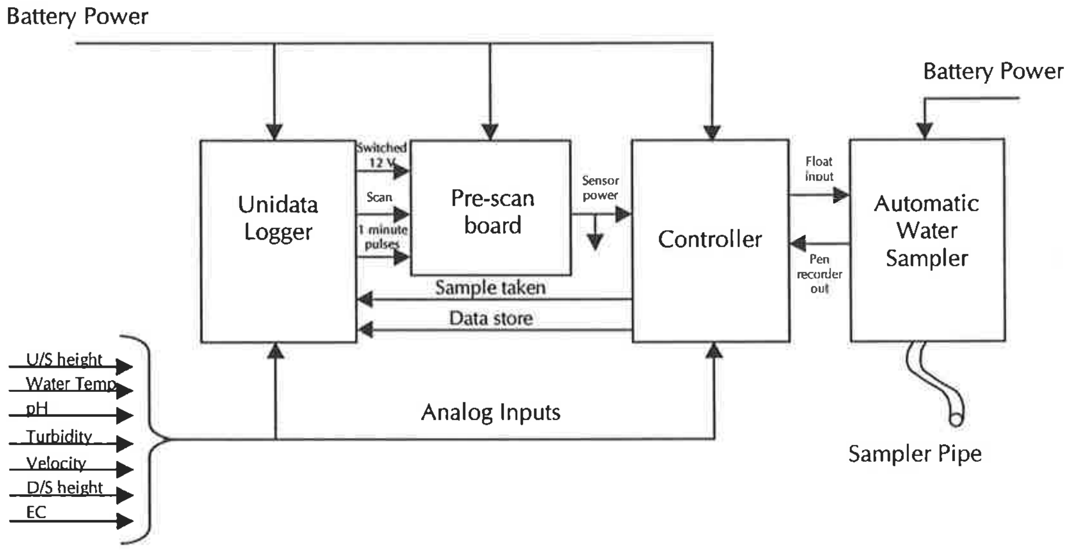


Figure 5.2 Schematic diagram of instrumentation

(Source: Williams, 1997)

5.1.1 Continuous Water Quality Monitoring

Ideally a monitoring program would consist entirely of continuously recording water quality probes. However, such a system is neither readily available nor affordable with a total water quality monitoring system valued at around $30000 plus. Therefore, for this study a selection of readily available water quality probes were utilised which were considered important parameters or indicators of other processes. For example, spikes in the pH readings can indicate alkaline discharges to the stormwater system (industrial wash water), temperature can be important for ecological processes, and EC is important when considering re-use of the water for industrial or irrigation purposes. The probes were mounted on a floating boat inside the protective instrument cage in the centre of the channel, or the outlet weir in the case of South Road Connector. Photograph 5.4 shows the boat at North Arm East during construction with the probes protruding from the bottom. The boats a South Road Connector and Henschke Street were different, Photograph 5.5 shows the completed boat out of the water at the Bund station. Regardless of the construction, the boats were designed to rise and fall with water level to keep the probes constantly submerged. The slightly more sophisticated construction of the North Arm East boat maintained the probe position at a depth of approximately 60 % of the total depth. The probes at each station varied slightly, Table 5.3 lists the probes installed at each of the stations.

Table 5.3 Probes installed at each of the streamflow gauging stations

Station	Probes
North Arm East Inflow Drain	turbidity, EC, pH (removed 1997), temperature, DO (removed 1997)
Henschke Street Inflow Drain	turbidity, EC, temperature
South Road Connector Outflow	turbidity, EC, pH, temperature

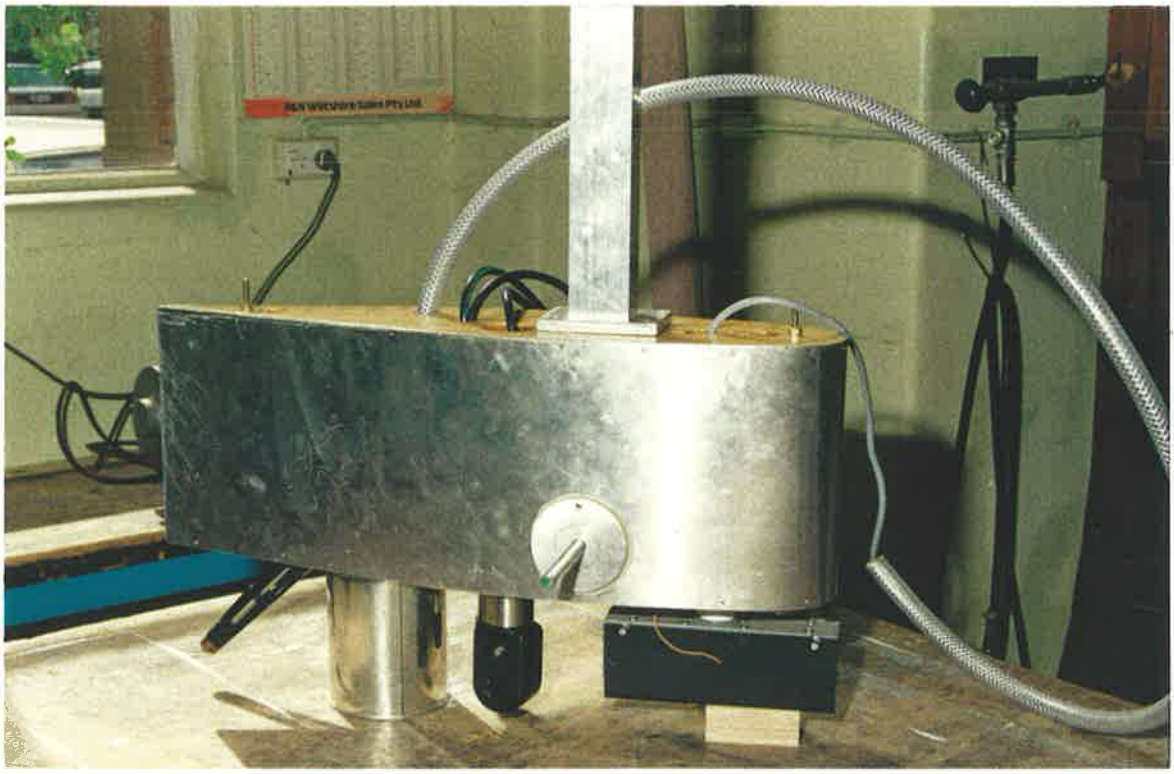
The dissolved oxygen probe at North Arm East was removed in January 1997 when it failed. It was not replaced, as the high flows at NAE tended to naturally aerate the water meaning DO depletion was not a concern at this site. The pH meter failed in early 1997 and was not replaced. The probes were cleaned as necessary, usually following each storm. The North Arm East site usually required cleaning more frequently than South Road Connector.

5.2 Flow Measurement

Flow measurement was different at each of the stations due to their different hydraulic controls. The flow measurement technique employed at each of the three stations is outlined below.

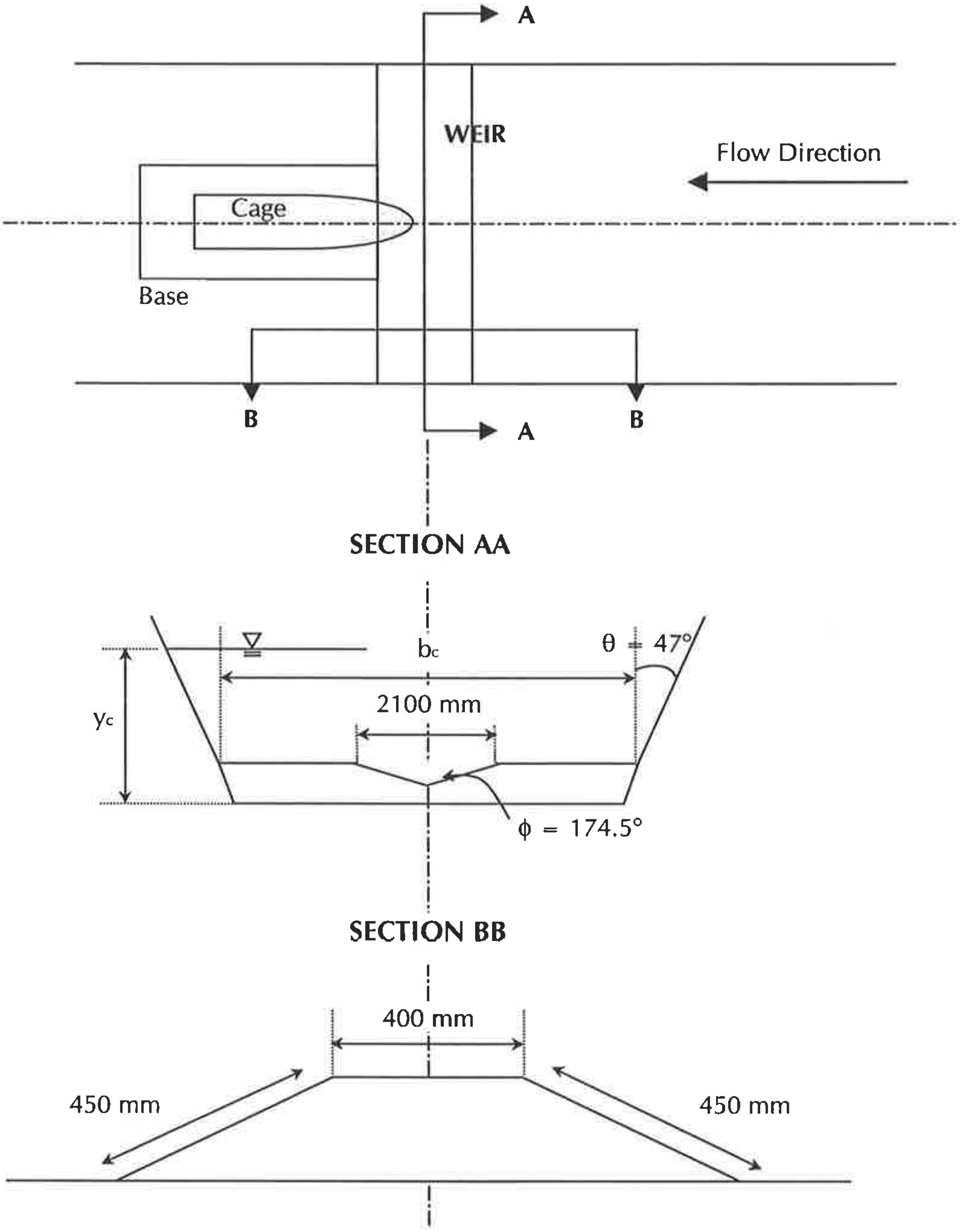
5.2.1 North Arm East Station

North Arm East was perhaps the easiest of the three stations in terms of flow measurement. The open channel, which is trapezoidal in cross section, contained a broad crested weir with a cross-section as shown in Figure 5.3. A shallow v-notch was constructed in the centre of the weir to channel low flows through a 100 mm deep trough in which the water quality probes were positioned. This station contained an upstream and downstream pressure transducer to measure stage height and together with the broad crested weir were used in a stage/discharge relationship to measure flow. In addition to this, a MACE ultrasonic doppler velocity meter provided direct velocity measurement for weir calibration. Ultrasonic doppler velocity meters are discussed in more detail in Section 5.2.2.1. The pressure transducers and velocity meter were mounted in steel casings and bolted to the floor of the channel.



Photograph 5.4 Continuous monitoring probes mounted on the boat for North Arm East – during construction.

Photograph 5.5 Continuous monitoring probes mounted on the boat at the Bund - lifted for cleaning. This is the same as the arrangement at South Road Connector.



DRAWING NOT TO SCALE

Figure 5.3 Plan and cross-sectional diagram of flow measurement structure used in the North Arm East Drain

A significant amount of work on flow monitoring at this site was conducted by Williams (1997) during 1995 and 1996. A rating curve was developed for the weir using a combination of a v-notch weir equation and a broad crested weir equation. The broad crested v-notch equation; Equation 5.1 was used for the first 50mm of flow as water was only flowing through the shallow v-notch shown in Figure 5.3 (Section AA). Once the flow exceeded this level, the broad crested weir equation for a trapezoidal cross section, Equation 5.2, was used.

$$Q = C_d C_v \frac{16}{25}\left(\frac{2}{5}g\right)^{0.5} \tan\left(\frac{\phi}{2}\right) h_1^{2.5} \qquad \textbf{Equation 5.1}$$

$$Q = C_d\left(b_c y_c + z_c y_c^{2}\right)\left(H_1 - y_c\right)^{0.5} \qquad \textbf{Equation 5.2}$$

where: Q = discharge (m^3/s)
C_d = discharge coefficient
C_v = approach velocity coefficient
ϕ = angle at bottom of v-notch
h_1 = upstream depth - cease to flow height
b_c = breadth of weir (see Figure 5.3)
z_c = $\tan(\theta/2)$ (see Figure 5.3)

and the ratio y_c/H_1 in equation 5.2 varies with z_c and H_1/b_c as per Table A.1 (Appendix A)

This equation was calibrated using the velocity data obtained from the velocity probe and it was found that the coefficients C_d and C_v could be approximated to 1. The initial stage-discharge relationship was developed in 1995 prior to the installation of trashracks 50 metres downstream of the monitoring station in January 1996. Although trashracks are a popular method of removing the portion of gross pollution from stormwater, they can create backwater problems upstream.

The trashracks, when full of litter as shown in Photograph 5.6, tend to drown out measurement structures such as weirs upstream and significantly reduce flow velocitites. Williams (1997) found that on occasions they reduced the flow rate in the channel by around 30 to 40 % depending on the amount of litter collected. Photograph 5.7 and Photograph 5.8 show the difference the trashracks make to flow in the channel. In the backwatered condition, there is no head drop over the weir.

A second rating curve was developed by Williams, using velocity data for backwatered events after the installation of the trashracks. This curve, along with the non-backwatered rating curve is given in Table A.2 and Figure A.1, Appendix A.

Photograph 5.6 Trashracks downstream of the North Arm East monitoring station full of litter during an event

Photograph 5.7 North Arm East weir during an event prior to the installation of the trashracks. Note the head drop over the weir.

Photograph 5.8 North Arm East weir during an event after the installation of the trashracks. Note there is NO drop over the weir, which is completely drowned out.

Investigation of individual events showed that often the rising limb, or some portion of it, was un-backwatered, whereas the falling limb was backwatered. Figure 5.4 shows an example of one such event in 1998. The green line shows the theoretical non-backwater curve, and the pink line is the backwater curve. The blue scatter points represent instantaneous flow calculated from the velocity meter recordings, and as can be seen some of these lie on the green line and some on the pink line. Between the two curves a 'transition curve' was used which represents the flow as the trashracks were accumulating litter.

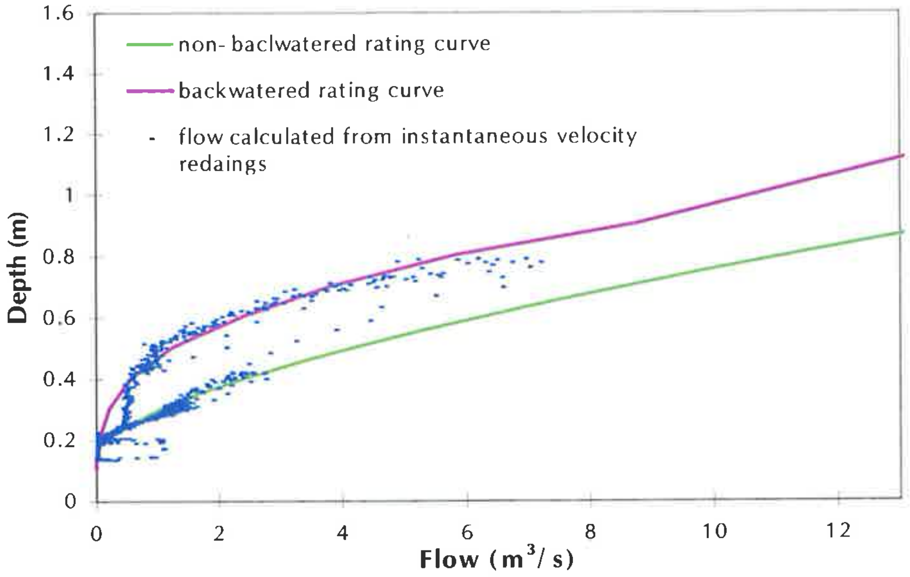


Figure 5.4 Instantaneous flow calculated from velocity compared to two dynamic rating curves developed by Williams (1997) for event on 11/06/98.

Figure 5.5 shows a comparison between the flow hydrograph assuming two different conditions. The blue line is the hydrograph assuming the fully non-backwatered condition while the black line assumes that backwatering occurred at some point and uses the three-pronged rating curve approach. The theoretical non-backwater curve gives quite a large over-estimation of flow: 180 ML compared to 99 ML.

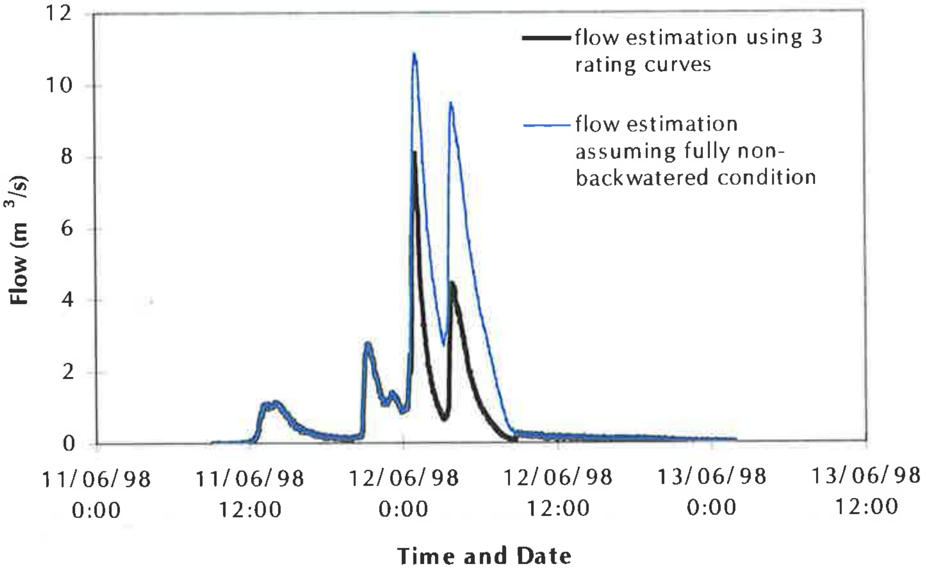


Figure 5.5 Comparison of flow hydrographs obtained when assuming fully non-backwatered condition (blue line) or using the three-pronged rating curve approach

Events analysed in this study supported the findings of Williams, with the exception of an extremely large event on October 30 1997. This event and its associated problems are dealt with in Chapter 6. Each event in this study was analysed separately to determine the point at which the channel became backwatered, if at all. Where possible velocity data was converted to flow. However, when a close match could be made between velocity flow and rating table flow, the flow calculated using the three rating tables was favoured as it tended to give a smoother hydrograph.

5.2.2 South Road Connector

The construction of the control structure at the outlet of Pond 4 presented a very hydraulically challenging situation. No flow measurement studies had been undertaken at the South Road Connector outflow site previously so calibration of the weir was necessary.

The outlet control can be seen in Photograph 5.9 (which was taken facing roughly east). The main control is a sharp crested weir up to a depth of 50mm over the weir when the rock filled wire mesh gabions become the controlling structure. The gabions span approximately 32.5 metres, broken in the middle by the sharp crested weir consisting of two, one metre wide metal plates. Downstream of the gabions is a shallow tailwater pond adjacent the culverts taking the water under South Road Connector/Salisbury Highway (see Photograph 5.10). This pond receives a small amount of overflow from the adjacent Pond 3 (see Figure 4.6) during high flows. The amount of flow however has not been quantified.

Flow measurement at this site was a two-step process.

1. Calculating flow through the culverts using ultrasonic doppler velocity meters placed in the culverts through which flow passes (see Photo)
2. Relating this discharge to upstream pond depth to calibrate a stage/discharge rating curve for back calibration of events for which there was no velocity data.

Photograph 5.9 Outlet structure at the South Road Connector station which the shows sharp crested weir and rock gabions either side.

Photograph 5.10 Culverts taking water from the outlet of Pond 4 underneath South Road Connector/Salisbury Highway. (Photograph taken facing North)

5.2.2.1 Flow Measurement Using Ultrasonic Velocity Probes

The South Road Connector station, because of the nature of the outflow structure, required a method of flow monitoring to allow calibration of a stage/discharge relationship. This was achieved by the use of three Unidata STARFLOW™ ultrasonic doppler velocity meters placed in three of the eight culverts that carried the water under South Road Connector/Salisbury Highway. The velocity meters were placed in the culverts shown in Figure 5.6.

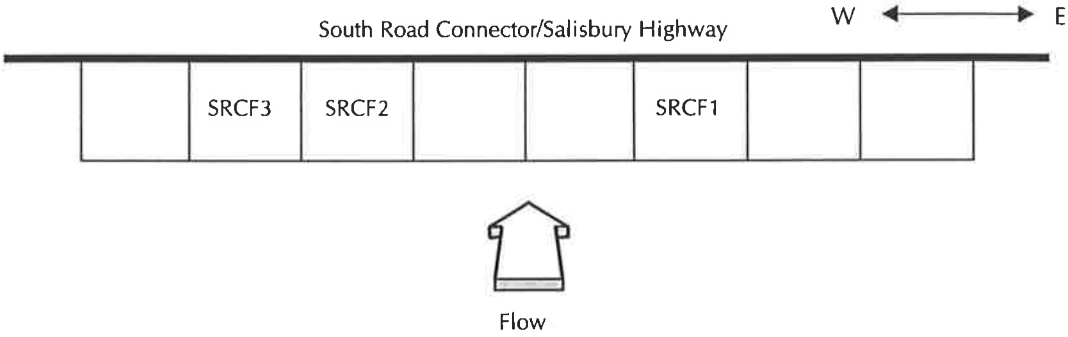


Figure 5.6 Location of the STARFLOW™ velocity meters in the South Road Connector Outflow culverts.

The probes calculated water velocity by utilising the Doppler principle, which is based on the principle that when sound is reflected from a moving object the frequency of the sound is altered by the velocity of the target. The fixed velocity probe emits a continuous acoustic signal at a fixed (ultrasonic) frequency, called the carrier frequency, which is reflected back off particles entrained in the moving water (see Figure 5.7). A receiver in the probe listens for the returning signals and detects any frequency changes. A processing system then accumulates and analyses the frequency changes and calculates a representative doppler shift from the range received (Unidata, 1996).

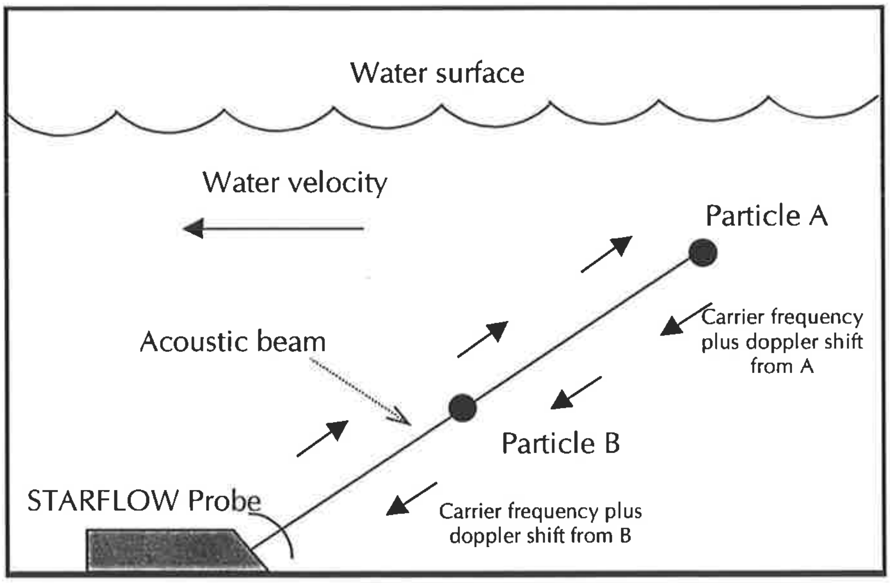


Figure 5.7 How STARFLOW measures velocity

(Source: Unidata, 1996)

Each doppler shift is directly related to the water velocity component along the beam via a physical relationship. Therefore by knowing the speed of sound in water, the velocity of the reflector, and hence the water, can be calculated by adjusting the velocity by the cosine of the angle of the beam. The velocity of water is calculated by Equation 5.3.

$$V_{water} = f_{shift}\frac{c}{2}f_c \cos\theta$$ **Equation 5.3**

where: V_{water} = velocity of water (mm/s)
f_{shift} = doppler shift
f_c = carrier frequency
c = speed of sound in water
θ = beam angle
= 30°

The speed of sound in water varies significantly with pressure, temperature, salinity and sediment. Temperature has the most significant effect, and the variation is shown in Table 5.4 for fresh water.

Table 5.4 Velocity of sound in fresh water at atmospheric pressure

Temperature (°C)	Velocity of sound (m/s) (1)
0	1402
5	1426
10	1447
15	1466
20	1482
25	1497
30	1509
35	1520

(1) Unidata, 1996

The STARFLOW instrument also measures temperature (along with depth and velocity) and applies a correction factor of 0.00138 mm/s/Hz/°C. This correction is the best fit for temperatures between 0 and 30°. The STARFLOW™ measures velocity to an accuracy of ±2% of measured velocity and depth to ±0.25% of calibrated range (Unidata, 1996).

Generally at the South Road Connector site it was found that velocities measured by SRCF2 and SRCF3 (see Figure 5.6) were very similar while SRCF1 was sometimes slightly different. Therefore, when all in operation, flow was calculated by averaging the velocities measured by SRCF2 and SRCF3 for the four western culverts and using the velocity recorded by SRCF1 for the four eastern culverts, and multiplying by cross sectional area. This strategy is represented by Equation 5.4.

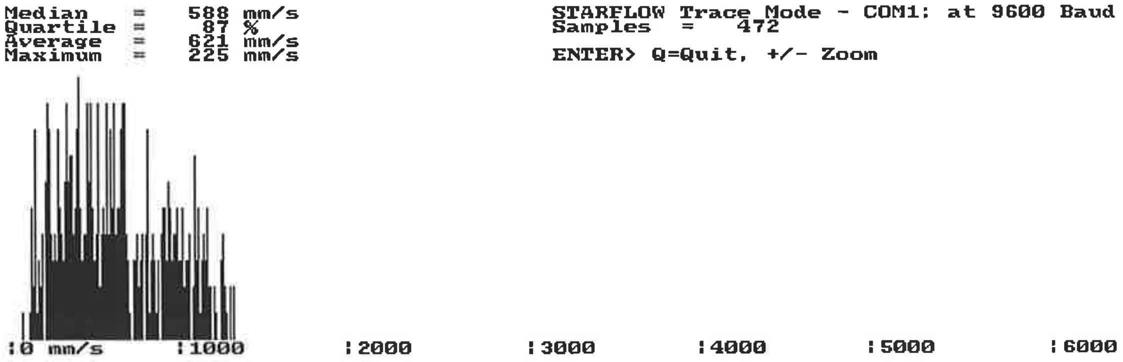
$$Q = W d_{av}\left[4\left(\frac{V_2 + V_3}{2}\right) + 4V_1\right]$$ **Equation 5.4**

where: Q = flow (m^3/s)
W = width of culvert
= 3.3 m
d_{av} = average depth recorded by the three starflows.
V_x = velocity recorded by SRCFX

The depth used in the calculations was the average of all three sensors. Generally there was very close agreement between the three measurements.

One of the limitations of this method of flow calculation was provided by the STARFLOW™ probes themselves. Flume testing of the instruments before installation revealed that for shallow depths of flow the instrument recorded very noisy data, particularly when combined with water surface perturbations caused by wind for instance. Figure 5.8 shows a velocity trace obtained for a shallow depth and high velocity with surface waves. Ideally the trace should roughly approximate a normal distribution, as seen in Figure 5.9, with the median of the distribution being the velocity that is actually recorded by the logger.

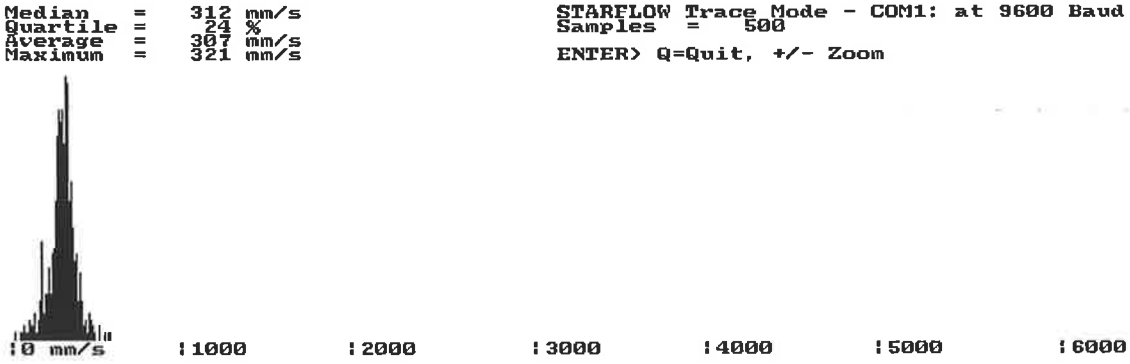
Even when velocity data was smoothed (using a moving average for example) often the data was too noisy to produce a smooth hydrograph. This also proved a problem when trying to develop a rating curve for the outlet weir.



```
Median    =   588 mm/s
Quartile  =    87 %
Average   =   621 mm/s
Maximum   =   225 mm/s

STARFLOW Trace Mode - COM1: at 9600 Baud
Samples   =    472
ENTER> Q=Quit, +/- Zoom
```

Figure 5.8 Noisy velocity trace recorded by STARFLOW™ logger for condition of high velocity, shallow depth and surface waves



```
Median    =   312 mm/s
Quartile  =    24 %
Average   =   307 mm/s
Maximum   =   321 mm/s

STARFLOW Trace Mode - COM1: at 9600 Baud
Samples   =    500
ENTER> Q=Quit, +/- Zoom
```

Figure 5.9 Ideal velocity trace recorded by STARFLOW™ logger

Figure 5.10 shows an instance for which the velocity meters provided satisfactory data producing a reasonable hydrograph. The sudden gap in the hydrograph is a result of backwatering effects from the Northern Ephemeral pond downstream. Tidal movements caused the Bund gates to close, thus backwatering the tailwater pond of Pond 4 although Pond 4 itself was not affected. This was another reason why the rating curve needed to be developed – to eliminate the need to consider complicated backwatering effects.

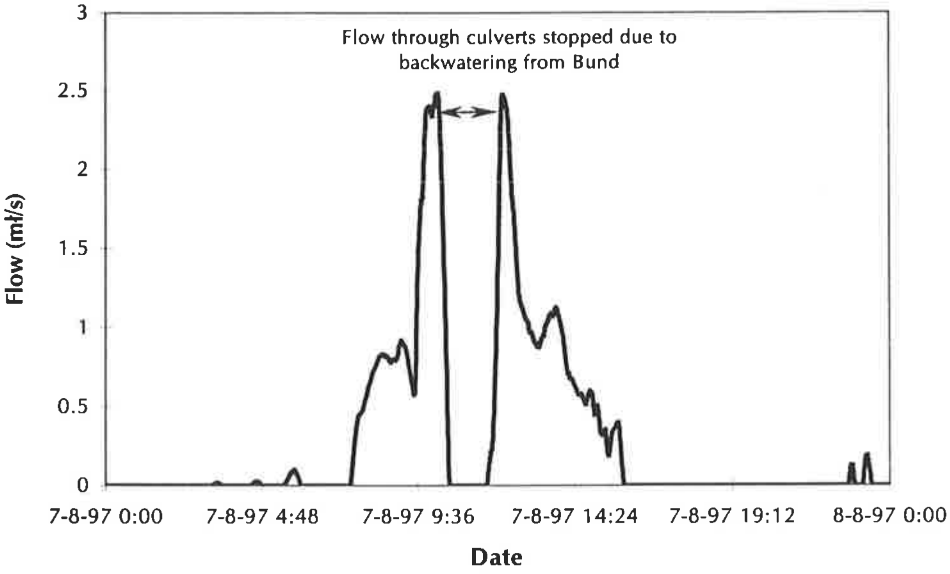


Figure 5.10 Outflow hydrograph calculated based on velocity data showing backwatering effects from the Bund (indicated by a sudden stop in the flow)

5.2.2.2 Development of a Pond-Height/Discharge Rating Curve

A first approximation of a stage discharge relationship was made using a combination of sharp crested and broad crested weir equations. The first 50 mm of flow is controlled by the sharp crested weir with a rectangular control section, and therefore Equation 5.5 from Bos (1989) was applied.

$$Q = C_e \frac{2}{3} (2g)^{0.5} b_c h_1^{1.5}$$ **Equation 5.5**

where: Q = discharge (m^3/s)
C_e = effective discharge coefficient = 0.587
b_c = breadth of weir = 2m
h_1 = head over weir (m)
(see Figure A.2, Appendix A)

The value of C_e was determined from Table A.3 in Appendix A, taken from Bos (1989).

Once the depth over the weir exceeded 50mm, water began flowing over the rock gabions (as shown in Photograph 5.11). Therefore the discharge over the broad crested weir had to be added to the flow passing through the sharp crested weir control.

Photograph 5.11 Flow over rock gabions which are beginning to act as a broad crested weir

The broad crested weir equation (Equation 5.2) was used to evaluate the discharge over the rock gabions. The value of C_d was determined from Equation 5.6.

$$C_d = 0.93 + 0.1\frac{H_1}{L} \qquad \textbf{Equation 5.6}$$

where: H_1 = upstream sill-referenced energy head
L = length of weir crest in direction of flow

The value was approximated to 0.93 as the ratio $0.1H_1/L$ was sufficiently small to make little difference to the value of C_d. The value of C_v in Equation 5.2 was determined to be 1 from Figure A.3 in Appendix A as the area ratio C_dA^*/A was also sufficiently small.

The combination of sharp and broad-crested weir equations provided a preliminary rating curve to be calibrated using velocity data from the culverts. At this point another obstacle was realised in the tailwater pond. During small storms this acted as a second ponding basin which further reduced flow peaks and hence velocities through the culverts. The movement of the peak through the culverts was also delayed and backwatering effects from the Bund sometimes halted flow through the culverts altogether (shown in Figure 5.10).

An alternative of developing a rating curve using downstream depth in the tailwater pond was investigated but discarded as the input from Pond 3 was unknown. Although flow occurs over a weir similar to that of Pond 4, height data was not available for flow quantification.

It therefore became a matter of finding an event large enough such that routing effects through the tailwater pond were insignificant, and velocity data was of adequate quality. An event on April19 1998 provided the first calibration of the preliminary rating curve as shown in Figure 5.11.

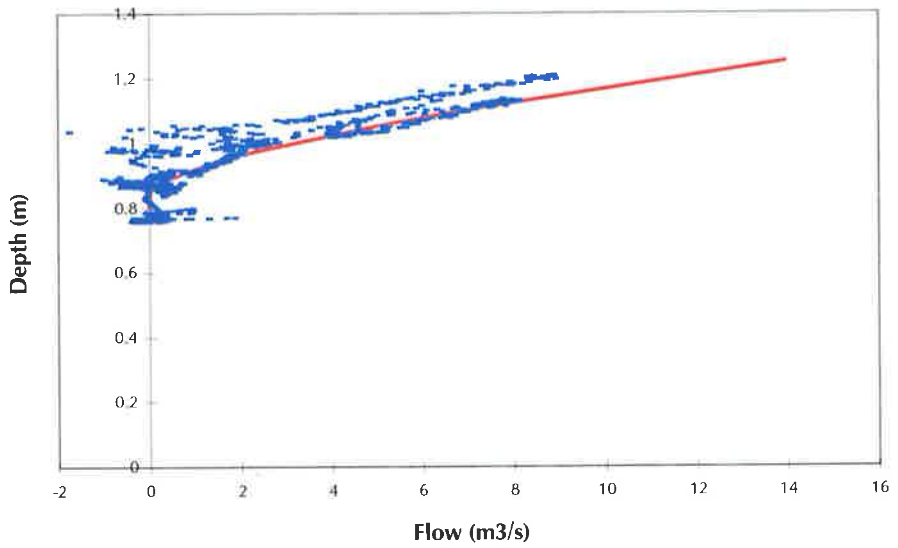


Figure 5.11 Instantaneous flow calculated from velocity for event on April 19 1998 compared to preliminary rating table.

The preliminary curve gave a surprisingly good estimation of flow considering the uneven nature of the rock gabions acting as a broad crested weir. Inserting data from other events (see Figure 5.12) in fact showed that a majority of events for which there was sensible velocity data fitted at the lower end of the rating curve and formed a reasonable distribution band.

The pond had a significant effect on reducing flow peaks such that high flows were rarely experienced. The rating curve was adjusted to fit the available data and its calibration was checked with each individual event analysed. Adjustments were made to the calibration when necessary. Figure 5.13 shows the preliminary and final calibrated rating curves. The rating curve is defined in Table A.4, Appendix A.

Outflows calculated were compared to the corresponding flows at the inlet (North Arm East Drain) for each event to ensure no gross errors were made in volume calculations.

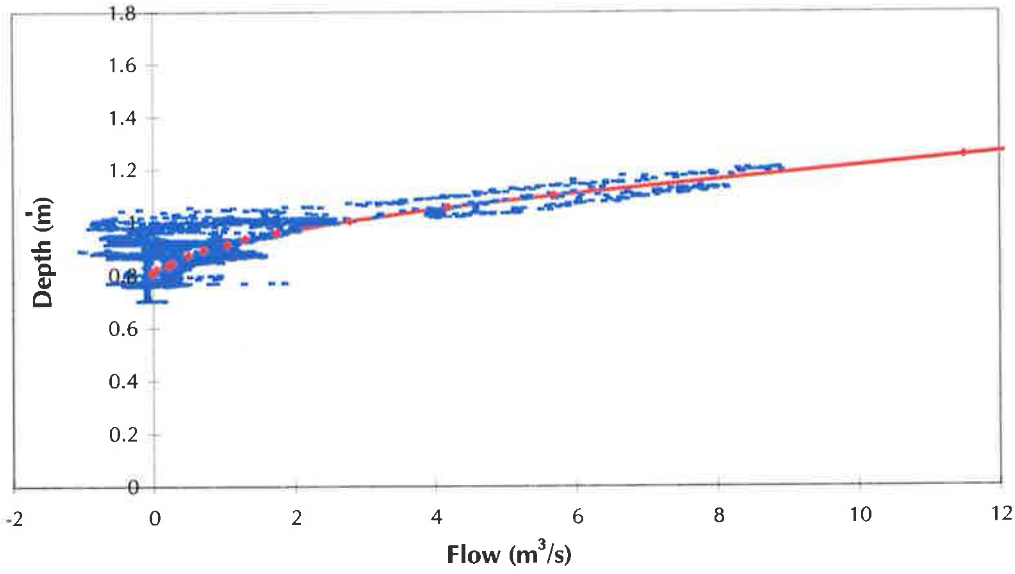


Figure 5.12 Measured flow data compared to rating curve

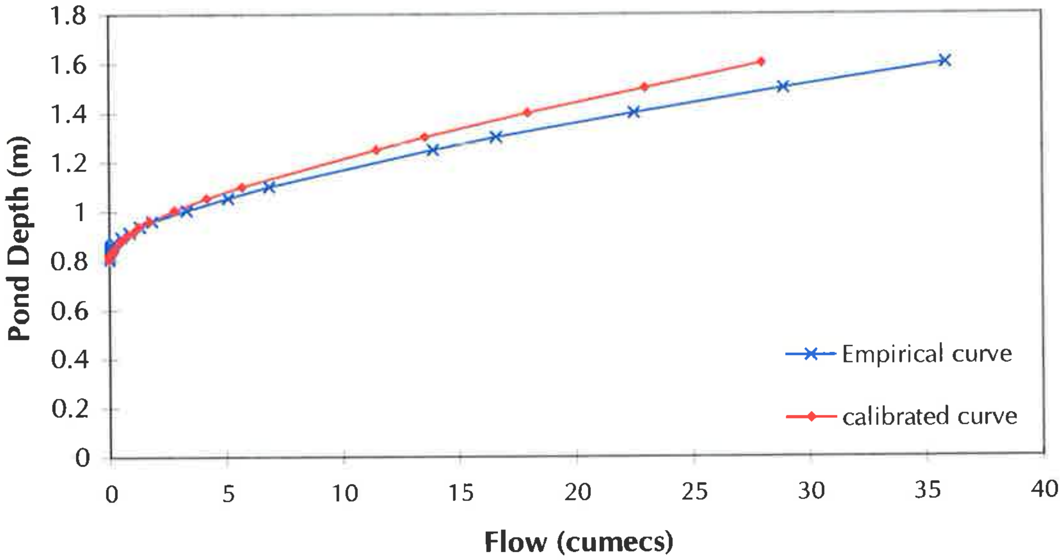


Figure 5.13 Empirical and calibrated rating curves for South Road Connector outflow station

5.2.3 Henschke Street

A triangular profile flat v-weir, or Crump-weir, was constructed at the Henschke Street site as discussed in Chapter 4. The head discharge equation for a short crested flat v-weir with vertical side walls taken from Bos (1989) is given by Equation 5.7. The Henschke street weir is a 1-to-2/1-to-5 weir.

$$Q = C_d C_v \frac{4}{15}\sqrt{2g}\,\frac{B_c}{H_b}\left[h_e^{2.5} - (h_e - H_b)^{2.5}\right]$$ **Equation 5.7**

where: Q = discharge (m^3/s)

C_d = discharge coefficient

C_v = approach velocity coefficient

h_e = the effective head over the weir crest

= $h_1 - K_h$

and: K_h = an empirical quantity representing the combined effect of several phenomena attributed to viscosity and surface tension.

Figure 5.14 defines the other terms used in Equation 5.7. For the 1-to-2/1-to-5 weir an average C_d value of 0.66 was used (Bos, 1989) and the C_v value determined from Figure A.4 in Appendix A. The K_h value used was 0.0005 for a 1-to-2/1-to-5 weir with a 1-to-20 cross slope (see Table A.5).

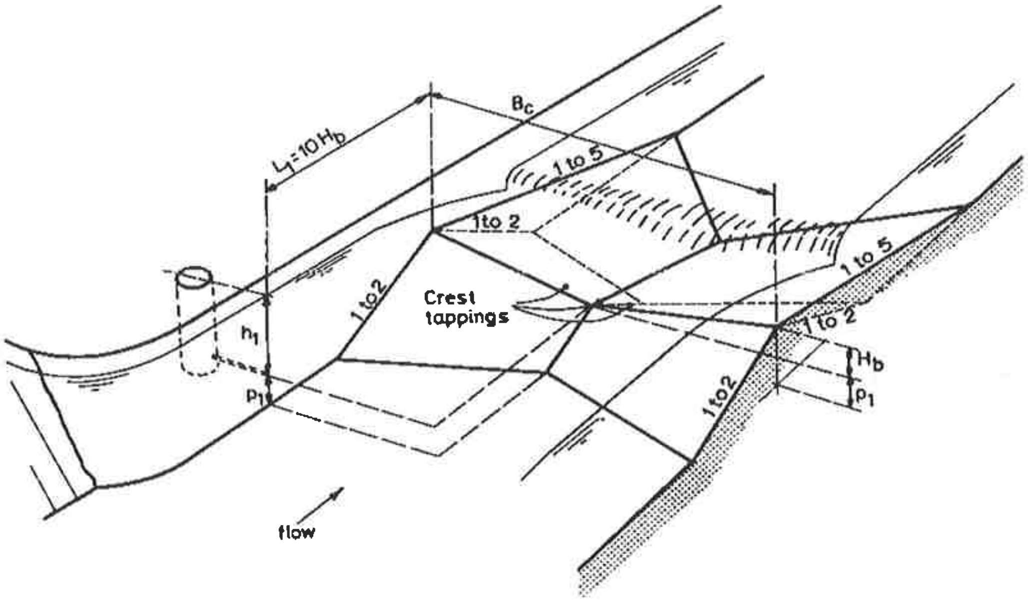


Figure 5.14 Dimensions of a short crested flat v-weir with vertical side walls
(Source: Bos, 1989)

A theoretical rating curve was developed from Equation 5.7, which is shown in Figure 5.15. The cease to flow level of the weir is 0.4 m.

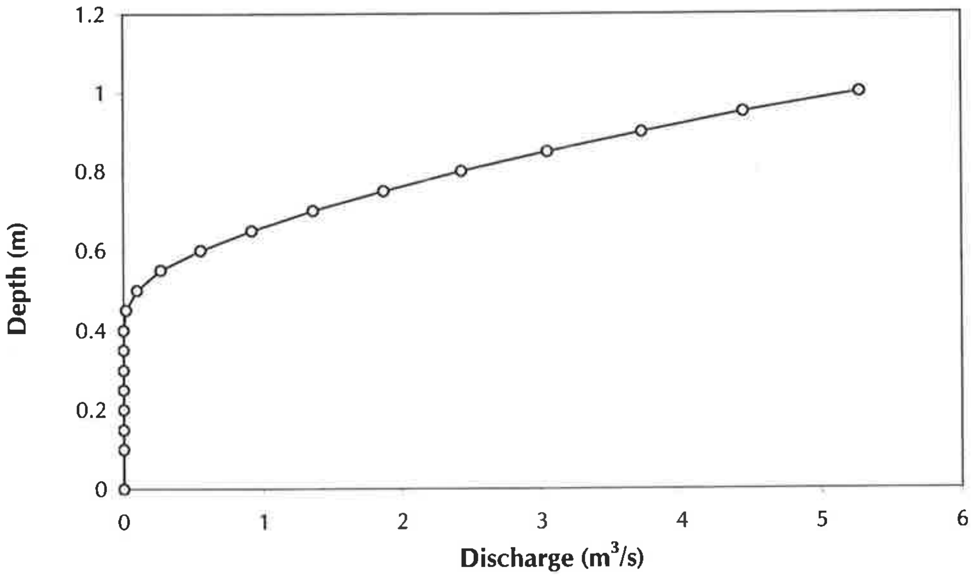


Figure 5.15 Theoretical stage discharge relationship for the Henschke Street Weir

The first few flow events following the completion of the weir indicated severe backwatering of the drain caused by rapid inflow from North Arm East drain and subsequent raising of the water level in Pond 4. The level of the weir crest in the Henschke Street drain is 0.7 m AHD, and the operating level of Pond 4 is 0.6 m AHD. It is common for the pond level to rise more than 100 mm during a moderate event and, as the Henschke Street catchment is small compared to North Arm East, the small amount of inflow from the Henschke Street catchment can not counteract the rising level in the pond from the NAE inflow. Figure 5.16, which is a graph of water depth upstream and downstream of the weir during an event, clearly shows that the downstream height is rising faster than the upstream height which can only be a result of water flowing up the drain from the downstream Pond 4. Photograph 5.12 shows the drain in the backwatered condition. There is no sign of the weir in front of the cage as it is completely drowned out.

As the weir rating curve could not be used, and the velocity meter installed at this site did not provide any useful data, the only other method of flow calculation contemplated was the use of an equation such as Manning's formula. Investigations into using this method proved the data set was not reliable enough and hence attempts at flow calculation were abandoned. The small size of the catchment compared the North Arm East (~10% of NAE) means that any inflow from this drain was likely to be insignificant in terms of total volumes and associated loads.

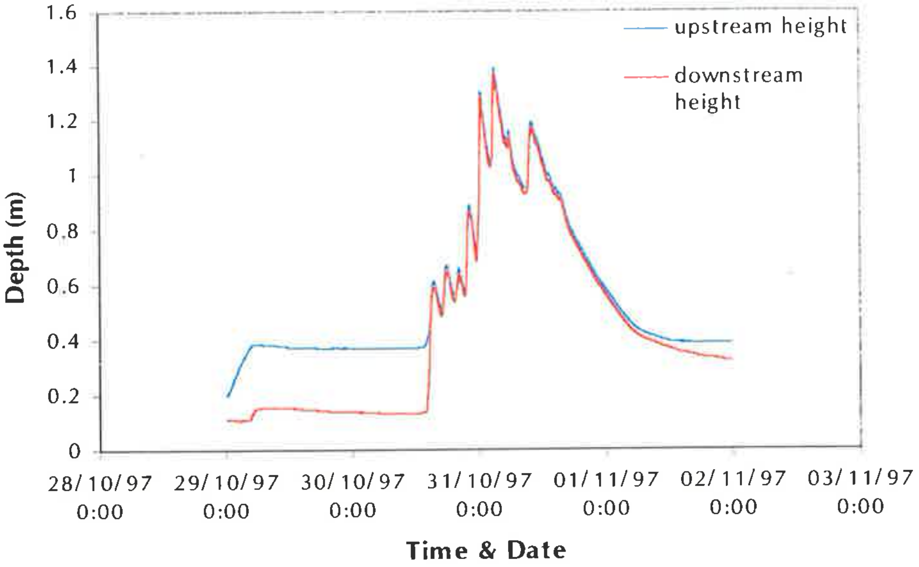


Figure 5.16 Upstream and downstream height in the Henschke Street Drain during an event in which the drain was backwatered by Pond 4

Photograph 5.12 The Henschke Street Drain during an event. There is no sign of the drowned out weir.

5.3 Event Based Water Quality Monitoring

The monitoring of water quality during events consisted of a combination of continuous monitoring by probes, as already discussed, and testing of samples taken by the automatic water sampler. The samples collected were analysed both in-house as well as being sent to the Australian Water Quality Centre for more comprehensive testing.

Samples were collected from site as soon as possible after the event, usually within 24 hours. Composite samples were either made up on site and taken directly to the Australian Water Quality Centre, or were taken back to the University and placed in cold storage for composite sample preparation and in-house testing.

5.3.1 In House Laboratory Testing

Each individual sample collected during an event was tested in the ESSO Environmental Laboratory in the Department of Civil and Environmental Engineering at the University of Adelaide. Samples were tested for total suspended solids, turbidity, total phosphorus, total nitrogen, and for some events ammonia-N, nitrate-N and total dissolved solids.

5.3.1.1 Total Suspended Solids

The TSS concentration of samples was determined using a vacuum filtration technique. Each sample was thoroughly mixed and a known volume passed through a pre-weighed 47 μm glass fibre filter paper using a Nalgene filtration apparatus. Filter papers were then placed on petri dishes and dried in an oven at 80 °C overnight. The dried samples were weighed and the TSS concentration calculated using Equation 5.3.

$$TSS = \frac{W_d - W_{FP}}{V}$$

Equation 5.3

where: TSS = total suspended solids concentration (mg/L)
W_d = dry weight of filter paper plus sediment (mg)
W_{FP} = filter paper weight (mg)
V = sample volume (L)

5.3.1.2 Turbidity

The turbidity of each sample was determined using the HACH model 2100P portable turbidimeter. This had a range of 0 – 1000 NTU and an accuracy of ± 0.5 NTU. Each mixed sample was poured into a clean turbidity vial to the marked line and the sample vial wiped with a soft cloth to remove any water or marks. A direct reading in NTU was then obtained from the turbidimeter using the 'auto range' and 'signal average' modes of operation.

5.3.1.3 Nutrients

The nutrients selected for in-house laboratory testing were total phosphorus, total nitrogen and, when time permitted, nitrate and ammonia. The HACH Test 'N Tube™ procedures were used which provided a safe and efficient method of testing with minimal waste generation. TP was tested using the PhosVer 3 and Acid Persulfate Digestion method, and TN was tested using the TNT Persulfate Digestion method. Both were digested in HACH COD reactors according to the standard procedures. A HACH DR/2000 Spectrophotometer was used for providing all nutrient concentrations. This enabled batch testing with a slight modification to the standard procedures.

With the large number of samples collected during wet periods, ammonia and nitrate were only tested sometimes, with the aim in mind of determining a rough nitrogen speciation. Nitrate was tested using the HACH Test 'N Tube™ Chromotropic Acid method while ammonia was tested using the low range Salicylate method. All parameters appeared to return satisfactory results with the exception of the TN test, which had a maximum resolution of 1 mg/L and tended to give inconsistent results on occasions.

5.3.1.4 Total Dissolved Solids

Total dissolved solids (TDS) were tested on sequential samples towards the end of the sample period to test the accuracy of the continuous EC probes. The samples were tested in the laboratory using a hand held Activon TDScan probe.

5.3.2 Composite Sample Preparation

As mentioned previously, up to 24 samples could be taken during an event at each station. As testing can be quite time consuming, as well as costly, samples collected during each event were combined into a flow weighted composite sample which was then analysed for a variety of water quality parameters.

Composite samples were prepared using a computer program called H_2OSAMP developed in the Department of Civil and Environmental Engineering at the University of Adelaide. The program weighted each sequential sample taken during an event according to the flow that occurred over the time period represented by the sample. Figure 5.17 summarises the procedure by which H_2OSAMP weighted each sample.

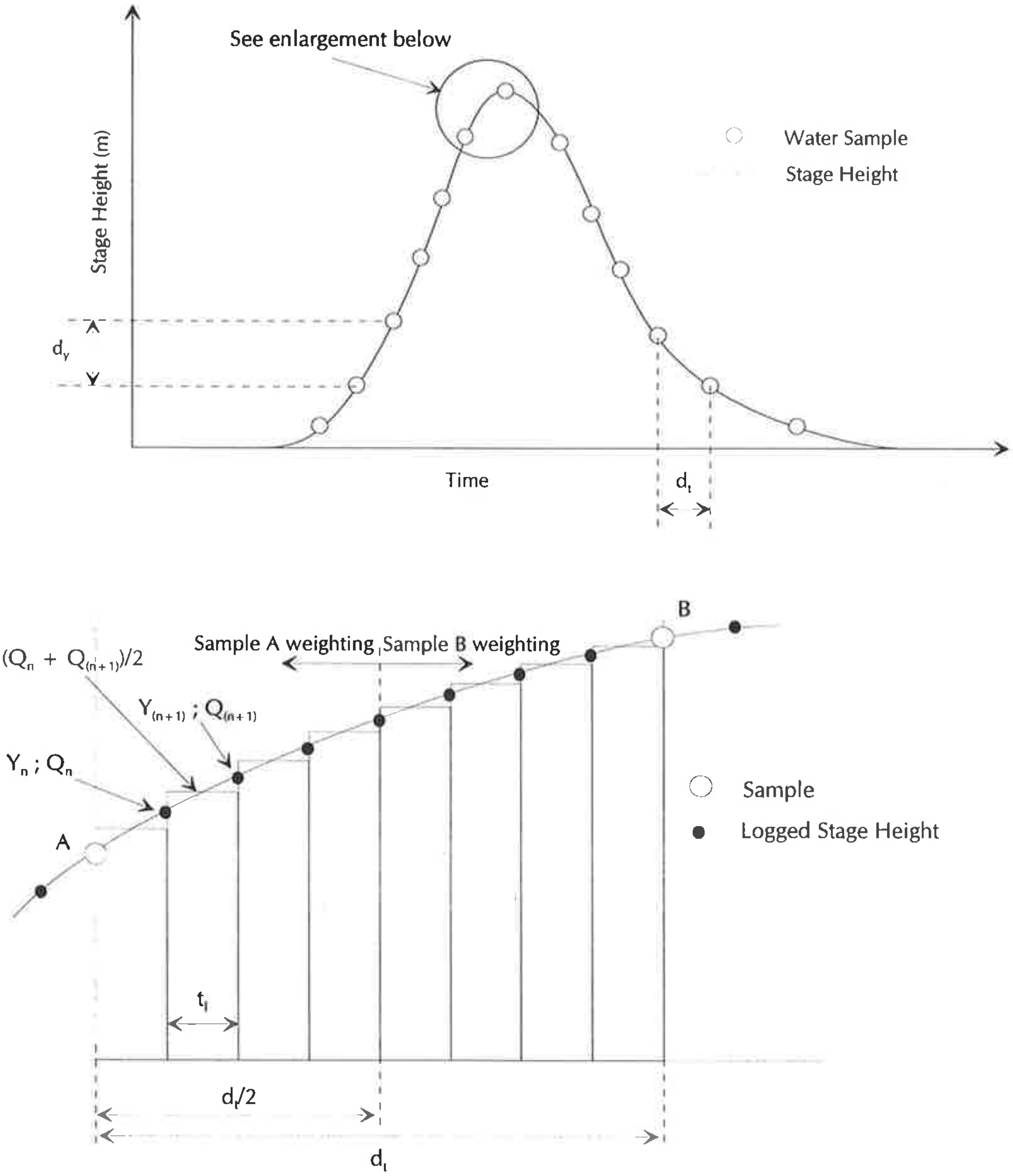


Figure 5.17 Method by which H_2OSAMP distributes weighting to each sample

Considering samples A and B (Figure 5.17 top) separated by a time d_t, H_2OSAMP distributed the first $d_t/2$ fraction of runoff to sample A and the second $d_t/2$ fraction to sample B. H_2OSAMP used rating tables to convert stage heights, Y_n, in the streamflow files to flows, Q_n, and then averaged the flow points Q_n and Q_{n+1} to determine the average flow rate for each logged time increment t_l.

The average flow rate $\frac{(Q_n + Q_{n+1})}{2}$ was multiplied by the time increment t_l to determine the volume of flow for t_l. The volumes of all increments were summed within the $d_t/2$ interval and this volume plus the volume for the $d_t/2$ interval before the sample, divided by the total event volume determined the weighting for sample A. This calculation was repeated for each sample providing a composite sample "recipe" over a specified period of time. The user was prompted to enter the required composite sample volume, for which the amount of each sample to be added to the composite sample was returned along with the total event volume in kilolitres.

5.3.3 External Testing

Each composite sample collected was sent to the Australian Water Quality Centre at Bolivar, South Australia, for analysis. The results returned gave event mean concentrations (EMCs) for 16 water quality parameters; a breakdown of five nutrient species and eleven metals.

The following lists are the nutrient species and metals tested at the Australian Water Quality Centre:

Nutrients: Ammonia –as N;
Nitrate + Nitrite –as N;
TKN;
Total phosphorus; and
Filterable reactive phosphorus –as P

Metals: Aluminium (total);
Arsenic (inorganic);
Cadmium (total);
Chromium (total);
Copper (total);
Iron (total);
Lead (total);
Manganese (total);
Mercury (total);
Nickel (total); and
Zinc (total).

Occasionally samples were analysed for Poly-cyclic aromatic hydrocarbons (PAHs), however, more often than not each parameter was below the level of detection. The following were the PAHs tested:

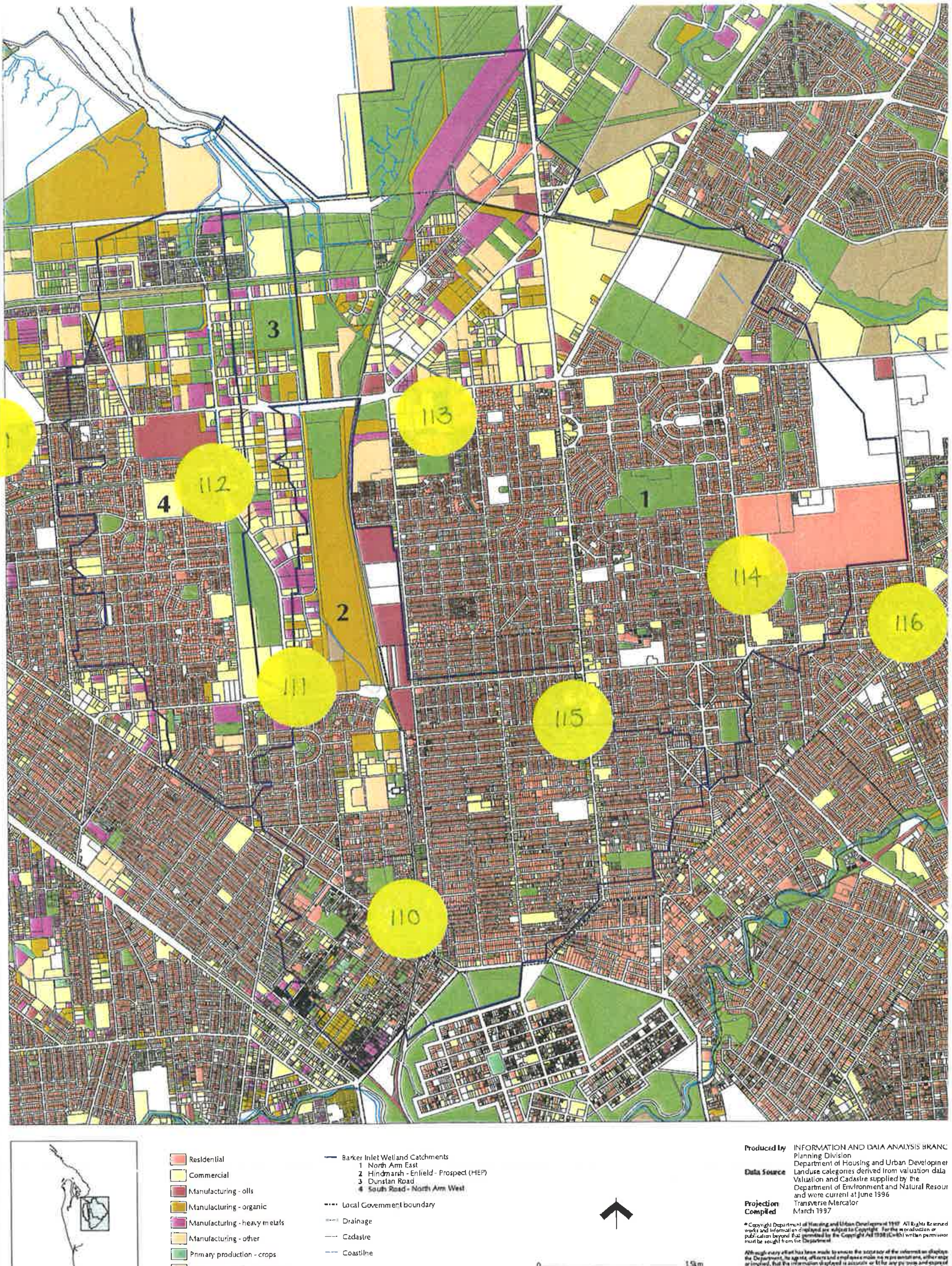
PAH: Acenaphthene;
Acenaphthylene;
Anthracene;
Benzo(a)anthracene;
Benzo(a)pyrene;
Benzo(*b*)fluoranthene;
Benzo(*g,h,I*)perylene;
Benzo(*k*)fluoranthene;
Chrysene;
Di-benzo(*a,h*)anthracene;
Fluoranthene;
Fluorene;
Indeno (*1,2,3–c,d*)pyrene;
Naphthalene;
Phenanthrene; and
Pyrene.

5.4 Rainfall Monitoring

Pluviometers were placed in each of the catchments of the Barker Inlet Wetland System between 1994 and 1997 to gain an understanding of the spatial and temporal variation in rainfall that can be experienced both within and between catchments. Locations were chosen to cover the catchment area while also providing security for the instrumentation.

Figure 5.18 shows the location of the seven rain gauges in the Barker Inlet Catchment. Five of these were Dataflow automatic rain gauges connected to DS93 8 channel 128k loggers. The other two were tipping bucket rain gauges. The North Arm East catchment, as Figure 5.18 shows, was covered by the rain gauges at the Enfield Council Depot (AU504113) and the Hampstead Centre (AU504114). The Prospect Council Depot gauge was on the boundary between the NAE and HEP catchments. Photograph 5.1 shows the pluviometer at the Hampstead centre in the North Arm East catchment.

Photograph 5.13 Pluviometer at the Hampstead Centre in the NAE catchment



Produced by INFORMATION AND DATA ANALYSIS BRANC
Planning Division
Department of Housing and Urban Developmer
Data Source Landuse categories derived from valuation data
Valuation and Cadastre supplied by the
Department of Environment and Natural Resour
and were current at June 1996
Projection Transverse Mercator
Compiled March 1997

Figure 5.18 Location of rain gauges in the Barker Inlet Wetland Catchment

5.5 Data Archiving

All data recorded by the loggers, both for flow and raingauges, downloaded to a lap top computer, was entered into HYDSYS, a hydrometric archiving and retrieval system upon return to The University of Adelaide. Each station was assigned a HYDSYS identification number as shown in Table 5.5 and Table 5.6, these are used as the archive filenames for each site.

Table 5.5 Streamflow station HYDSYS identification numbers

Station	HYDSYS Identification
North Arm East Drain	AU504101
Hindmarsh Enfield Prospect Drain	AU504102
Dunstan Road Drain	AU504103
South Road North Arm West Drain	AU504104
Henschke Street Drain	AU504105
South Road Connector A Outflow Station	AU504106
BUND Outflow Station	AU504107
Eastern Parade Drain	AU504201

Table 5.6 Rainfall station HYDSYS identification numbers

Station	HYDSYS Identification
Ovingham DRT Depot (REMOVED)	AU504110
Regency Park STA Depot (now Coopers Distribution Centre)	AU504111
Angle Park ETSA Depot	AU504112
Enfield Council Depot, Kilburn	AU504113
Hampstead Centre , Clearview	AU504114
Prospect Council Depot	AU504115
Greenacres Council Depot	AU504116
Enfield Council Depot Tipping Bucket	Not Archived
Rosewater Bowling Club	AU504211
Woodville High School (REMOVED)	AU504212

Data from North Arm East was archived and then checked for irregularities and gaps. Gaps were filled in and spikes, particularly in the velocity record, smoothed using the Data Managers Workbench. Figure 5.19 shows a section of the velocity record before and after smoothing.

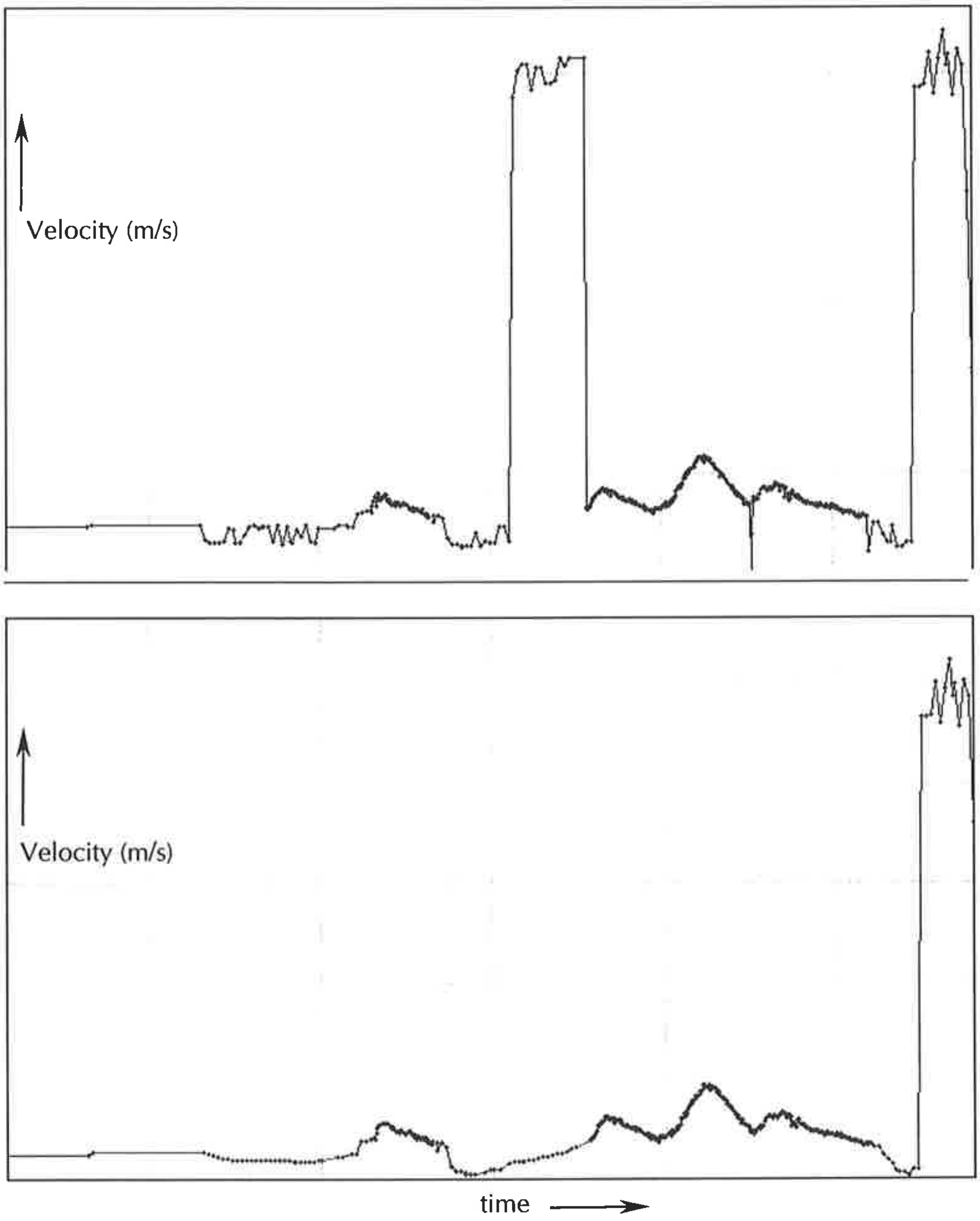


Figure 5.19 Section of velocity record in HYDSYS before and after smoothing (the spike on the right has been left purely to maintain the same scale in each shot)

Stage height data was calibrated using readings taken from a staff gauge on site during each visit. When compared to the logger reading at the same instant, a calibration factor could be applied.

Before South Road Connector data could be archived, the streamflow files had to be combined with the three separate flow files. This was achieved using a merge program and batch file. The C program compared the times and dates of the streamflow file with the SRCF1 file firstly and matched them. Because the program allowed a tolerance of ±1 minute, it didn't matter if the loggers were started on an odd or even numbered minutes. Both loggers recorded at two minute intervals during flows, however the streamflow logger only recorded at ten minute intervals during baseflow while the starflow logger recorded at two minute intervals all the time. This meant that some superfluous flow data was discarded during baseflow conditions. A batch file was used to repeat the execution of the C program three times, each time adding on the next flow file.

Once combined, the South Road Connector data was archived in the same manner as North Arm East. Due to the three sets of velocity data, a large amount of data smoothing was sometimes necessary.

5.6 Data Collection Problems

As with any data collection program, it can be expected that there will be some problems encountered along the way. Due to the nature of this data collection program, over the two year period a number of problems were encountered; some due to instrument malfunctions, some due to human error, and some that were, and still are, largely unexplainable.

At the South Road Connector site the main problems experienced initially were with the velocity meters. The three probes failed continuously, having to be replaced several times each before the problem was identified. It was found that the failure was occurring when the outer casings cracked and moisture penetrated the instruments. The casings were more than likely cracked by people stepping on them while "exploring" the culverts. The casings were modified using a more robust material and the same problem was not encountered again. Despite finally getting all the probes up and running, they have often recorded extremely noisy data even during flow conditions as shown in Figure 5.20.

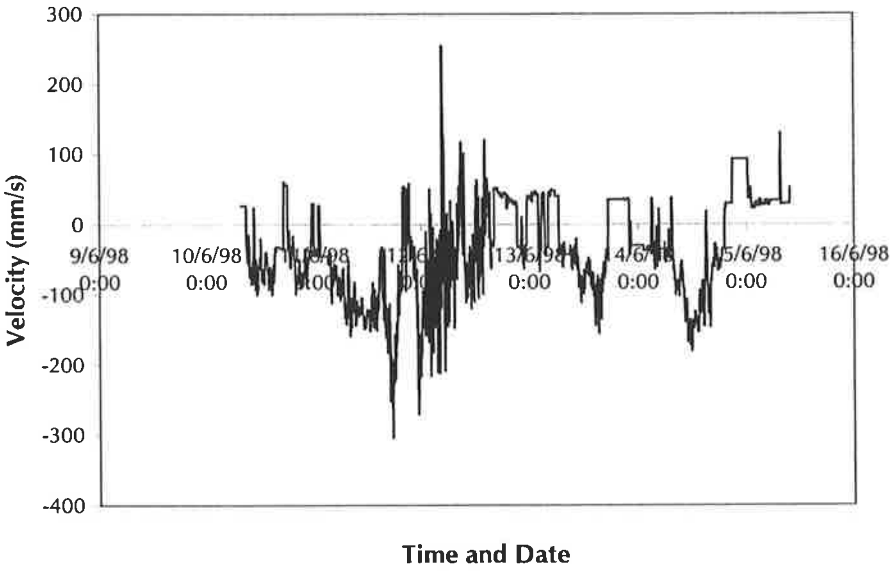


Figure 5.20 **Section of the velocity record at South Road Connector flow station 1. This was during an event , however the probe recorded nothing but noise.**

This graph shows a section of velocity data recorded during an event. The data makes no sense as it is not only noisy but consists almost entirely of negative velocities. This was a common problem with each of the STARFLOW™ meters at the South Road site, however not with the same instruments at the Bund. The most likely cause is the placement of the meters rather than a problem with the instruments themselves. The culverts under South Road Connector are much larger than at the Bund and create a "wind tunnel" effect with quite large waves on the water surface resulting. These waves were probably the cause of the noisy data.

Pressure transducer failure was also a common problem encountered; the upstream NAE transducer was replaced once and the upstream SRC transducer replaced twice. Figure 5.21 shows a section of the upstream height data at South Road when the first transducer failed, producing useless spikey data. Figure 5.22 shows another section of the same record once a new transducer had been installed. This showed strange daily fluctuations similar to the behaviour that temperature or DO would display. This problem could not be rectified so the transducer was replaced again.

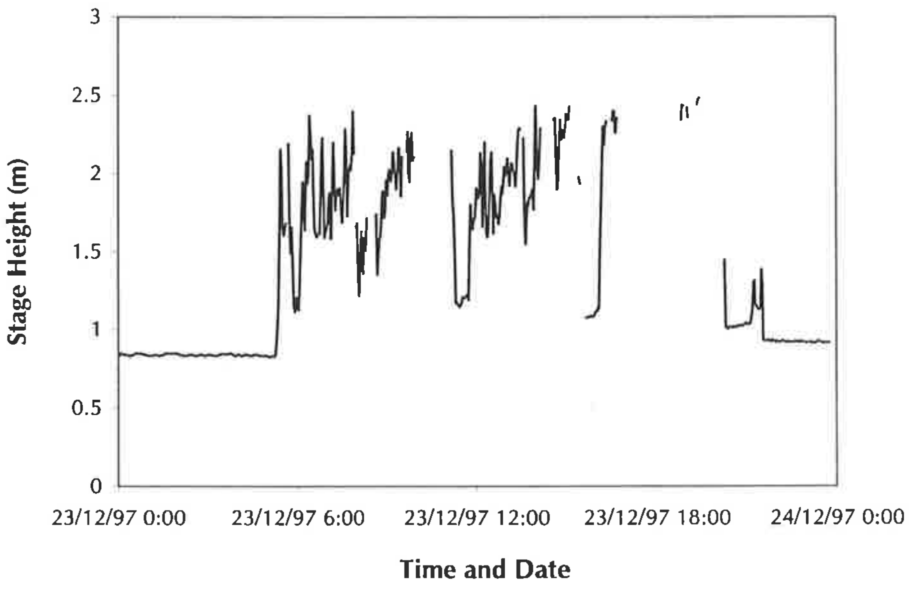


Figure 5.21 Upstream height data at SRC when the first pressure transducer failed.

The original inflow sites used Tain turbidity sensors which had problems with lenses becoming fouled up during storms, particularly at North Arm East. There were other problems associated with them which are discussed in more detail in Williams (1997). The new stations used Greenspan turbidity probes which were not susceptible to fouling, and required infrequent cleaning. The first probe installed at South Road Connector recorded unrealistic high readings, and the probe at the Bund showed daily fluctuations and high readings as shown in Figure 5.23.

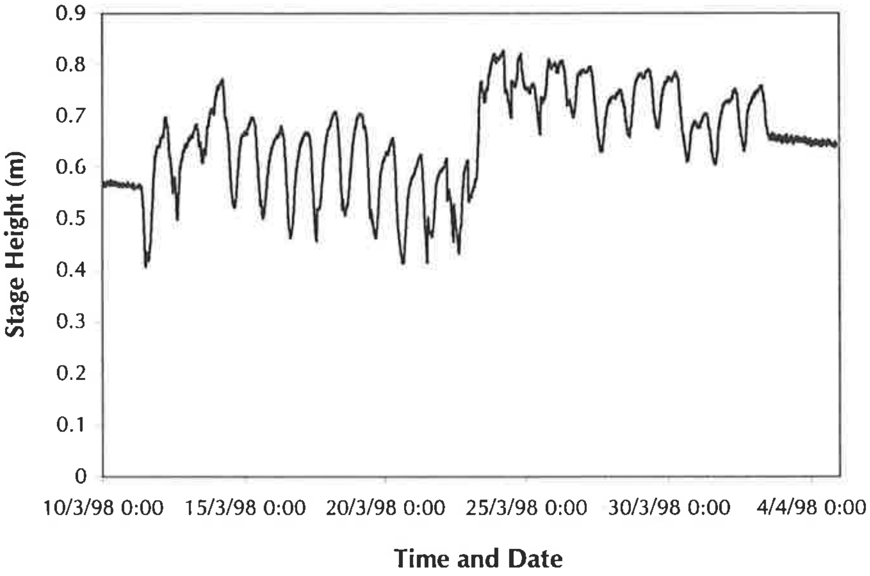


Figure 5.22 **Upstream height data at South Road after replacement of the first pressure transducer – this transducer was subsequently removed and replaced!**

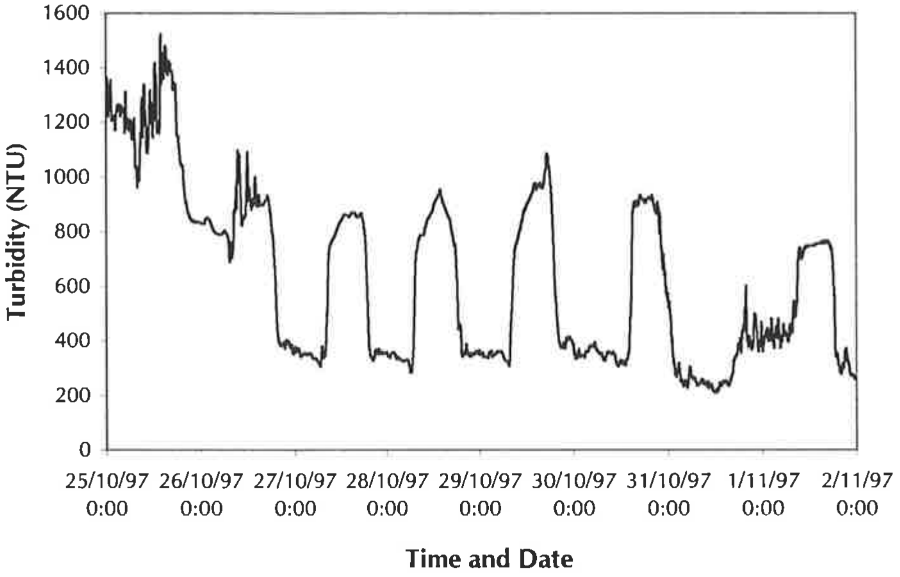


Figure 5.23 **A section of the Turbidity record at the Bund which shows daily fluctuations and unrealistically high values.**

Aside from the many instrumentation problems encountered, there were other problems such as vandalism. The South Road Connector and Henschke Street sites were vandalised in August 1997 which resulted in a significant storm being missed and the stations being off line for around one week (longer for Henschke Street). The steel cabinets were broken into and all electronic equipment stripped and thrown in the water. Although most of it was recovered, it was damaged by water and had to be replaced. Around $15, 000 worth of damage was done in the two vandal attacks. A similar break-in at North Arm West occurred in early 1998, after which the station was abandoned.

Human error occasionally resulted in missing data or storms. Sources of these errors were flat batteries, full logger buffers (someone forgot to re-load the logger!), forgetting to switch on the sampler, or loading loggers with the wrong scheme. Problems have also been experienced with samplers not triggering, loggers malfunctioning, hoses falling off samplers and many more. These "glitches" were usually not identified and rarely happened again. They were simply put down to "Gremlins" which helped to make life interesting and keep the team on their toes.

6

Water Quality Results

There are a number of different ways in which to interpret the data available, and to use it to explain the performance of Pond 4. Firstly the data can be examined on an event by event basis. This provides an understanding of how the pollutant concentrations may change throughout the duration of an event, provides evidence of phenomena such as 'first flush', and may lead to an indication as to which is the critical portion of an event to capture and treat. Event analysis can also lead to the development of retention curves based on residence time such as those developed by Lawrence (1986), which will describe how the pond can be expected to perform under different flow conditions.

A second, and perhaps more important, way of analysing the data is to look at it over a longer time frame such as a year. This can put into perspective the longer term performance of the pond that may be achieved since ultimately long term pollutant removal is what is required.

In this chapter, water quality results are dealt with both on an event basis as well as examining the corresponding annual figures. An example of a small, an intermediate, and a very large event are given for comparison. The ultimate aim of the project, the determination of the pond performance, is discussed in some detail.

6.1 Event Analysis

Event monitoring was carried out over the period from January 1997 to August 1998. Over this period a total of 54 storms were recorded and monitored at the North Arm East inflow station; 23 of these in 1997, and 31 during the period 1st January to 31st July 1998. The South Road Connector record is not as comprehensive as the station was not completed until July 1997. Events at this station were monitored over the period of 1st July 1997 to 31st July 1998. Emphasis has been placed on 51 inflow events in total, completing a 12-month monitoring period from 1st August 1997 to 31st July 1998. During this period about 6 storms were missed, or not monitored entirely, due to instrumentation malfunctions, achieving a success rate of about 88 %.

The monitored storms range in size from very small runoff events of 5 ML to the extremely large event of October 30 1997 at 760 ML. Figure 6.1 displays the inflow event volumes monitored over the 12-month period. The graph shows that the majority of events are less than 50 ML in volume, while only three were greater than 100 ML over the study period. The vertical scale on Figure 6.1 has been shortened to give greater clarity of the smaller events, and hence the actual size of the largest event (October 30th, 1997) is not shown.

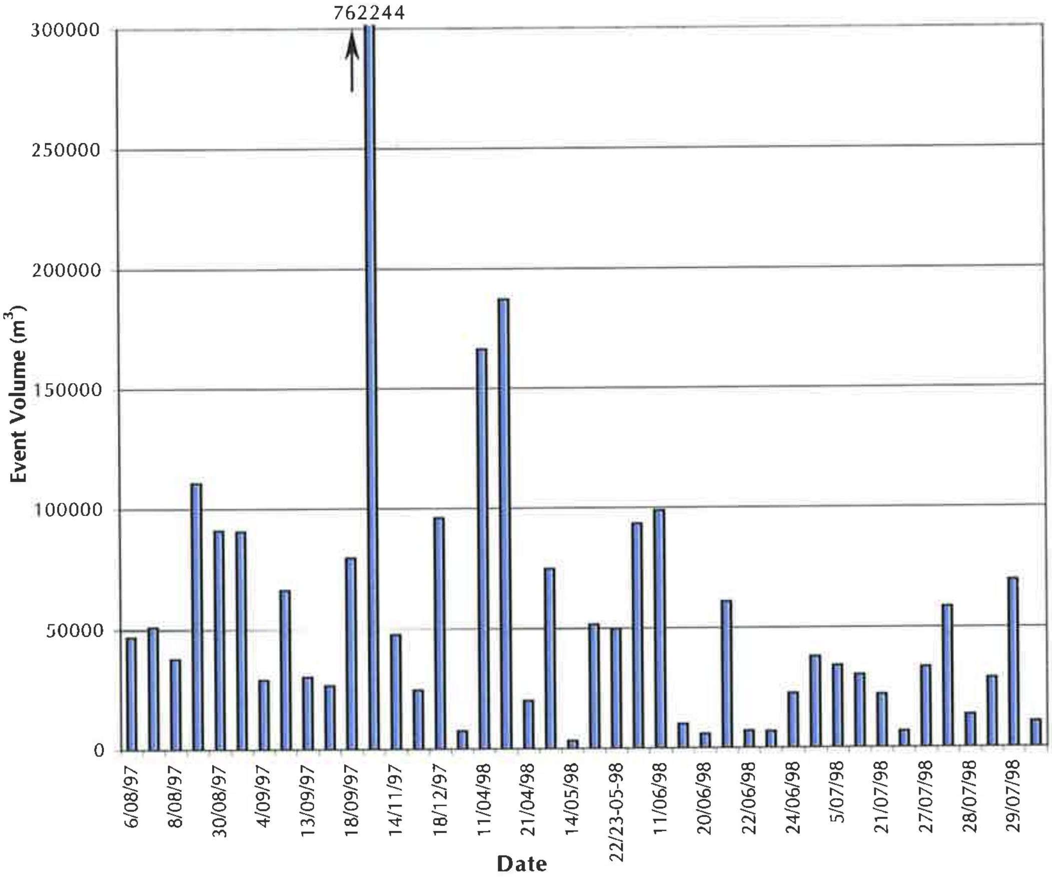


Figure 6.1 **Inflow event volumes monitored at the North Arm East station over the period 1st August 1997 – 31st July 1998**

Not all events recorded at the inflow station were recorded at the outflow of Pond 4. Small events following dry periods, particularly those in summer when evaporation rates were typically high, on occasion did not produce any outflow but rather replenished the pond storage depleted by evaporation.

The concentrations quoted in the following sections are the event mean concentrations (EMCs) determined for each individual event, except graphs, which show the instantaneous concentration throughout an event. The event mean concentration is effectively a flow weighted concentration which is representative of the average concentration over the entire storm event. Results returned from the Australian Water Quality Centre were already in EMC format as the sample delivered was a flow weighted composite (see Section 5.3.2). The event mean concentration (EMC) was determined using sequential samples by Equation 6.1.

$$EMC = \frac{\sum C_i V_i}{\sum V_i}$$ **Equation 6.1**

where: C_i = Concentration of sample i (mg/L)
V_i = flow volume since last sample taken (m^3)

The TSS EMCs quoted throughout the text have been derived from sequential samples analysed in-house and reduced to an event mean concentration by the application of Equation 6.1.

Over the 18 month monitoring period a total of 103 composite samples were prepared and taken to the Australian Water Quality Centre for analysis, 69 of these were from the North Arm East station and 34 from the South Road Connector station. In addition, 1074 sequential samples were individually tested for TSS and turbidity in the University of Adelaide Environmental Research Laboratory (Department of Civil and Environmental Engineering). A large number of these were also tested for total phosphorus and total nitrogen.

Table 6.1 and Table 6.3 provide the summary statistics obtained over the study period for the North Arm East and South Road Connector stations respectively. The number of samples analysed in each case is given to provide an indication of the size of the data set. The results obtained from the previous monitoring period, over 1995 and 1996 are given for comparison in Table 6.2. A similarity between the statistics of both data sets is evident.

The coefficient of variation, C_v, shown in the far right column of Table 6.1 and Table 6.3, provide a measure of the variation in each of the pollutants data sets. Generally, a C_v of less than 0.6 indicates a parameter of low variability which does not need to be monitored as intensively. Tables 6.1 and 6.2 suggest that the metals are the parameters which exhibit the smallest variation, particularly at the South Road Connector Outflow Station.

The entire set of event mean concentrations and event loads for all parameters monitored are given in Appendix B. Table B.1 and B.2 show the EMC data and loads (respectively) for the North Arm East Station while Table B.3 and B.4 are the equivalent tables for the South Road Connector station.

Table 6.1 Summary statistics for North Arm East event mean concentrations 1997-98

Parameter	Number	Maximum (mg/L)	Minimum (mg/L)	Mean (x) (mg/L)	Median (mg/L)	Standard deviation (σ)	Coef. Variation $C_v = \sigma/x$
Ammonia (as N)	62	1.52	0.016	0.208	0.12	0.253	1.22
TKN (as N)	61	8.05	0.68	1.86	1.77	1.24	0.66
Nitrate + Nitrite (as N)	62	0.583	< 0.05 *	0.322	0.328	0.115	0.355
Total Nitrogen	61	8.15	0.15	2.17	1.77	1.21	0.560
Filt. Reactive Phosphorus	60	0.81	0.033	0.133	0.082	0.142	1.07
Particulate Phosphorus	60	1.60	0.084	0.307	0.266	0.238	0.777
Total Phosphorus	61	1.94	0.131	0.435	0.360	0.313	0.718
Aluminium	63	12.6	0.827	3.26	2.99	2.26	0.695
Arsenic (inorganic)	63	0.05	0.001	0.005	0.003	0.006	1.29
Cadmium	58	0.0032	< 0.0002 *	0.001	0.0008	0.001	0.582
Chromium	55	0.055	< 0.005 *	0.015	0.012	0.010	0.705
Copper	63	0.141	0.01	0.044	0.039	0.027	0.599
Iron	65	12.4	0.607	3.00	2.47	2.14	0.715
Lead	65	0.636	0.02	0.137	0.117	0.112	0.821
Manganese	63	0.341	0.02	0.088	0.072	0.061	0.695
Mercury	36	0.0005	< 0.0001 *	-	-	-	-
Nickel	62	0.017	< 0.001 *	0.006	0.005	0.003	0.495
Zinc	65	1.26	0.081	0.473	0.427	0.225	0.476
Total suspended solids	59 **	828.1	33.3	191.5	147.2	170.4	0.890

* *Indicates concentration is below the limits of detection for analysis method.*

** *606 sequential samples tested to produce 59 EMC values*

Table 6.2 Summary statistics for North Arm East event mean concentrations 1995-96

Parameter	Number	Maximum (mg/L)	Minimum (mg/L)	Mean (x) (mg/L)	Median (mg/L)	Standard deviation (σ)	Coef. Variation $C_v = \sigma/x$
Ammonia (as N)	29	1.1	0.033	0.204	0.15	0.22	1.09
TKN (as N)	29	4.96	0.44	1.82	1.42	1.18	0.65
Nitrate + Nitrite (as N)	29	0.592	0.01	0.339	0.33	0.14	0.40
Total Nitrogen	29	5.5	0.69	2.16	1.82	1.20	0.56
Filt. Reactive Phosphorus	29	0.278	0.021	0.079	0.072	0.05	0.63
Particulate Phosphorus	29	0.745	0.065	0.274	0.216	0.17	0.62
Total Phosphorus	29	0.8	0.116	0.352	0.314	0.17	0.48
Aluminium	29	5.53	0.417	2.55	2.36	1.27	0.50
Arsenic (inorganic)	29	0.005	0.001	0.003	0.003	0.001	0.40
Cadmium	29	0.0019	0.0002	0.0008	0.0006	0.000	0.59
Chromium	29	0.03	0.005	0.012	0.009	0.007	0.59
Copper	27	0.082	0.016	0.039	0.032	0.02	0.48
Iron	29	6.36	0.469	2.51	2.21	1.38	0.55
Lead	29	0.36	0.042	0.150	0.112	0.10	0.65
Manganese	29	0.201	0.029	0.086	0.065	0.047	0.55
Mercury	29	0.003	0.0001	0.0003	0.0002	0.001	1.78
Nickel	29	0.033	0.003	0.009	0.007	0.006	0.71
Zinc	29	1.01	0.258	0.507	0.438	0.21	0.41
Total suspended solids	22	529.4	27.1	147.6	116.6	111.62	0.76

* *Indicates concentration is below the limits of detection for analysis method.*

** *606 sequential samples tested to produce 59 EMC values*

Table 6.3 Summary statistics for South Road Connector event mean concentrations 1997-98

Parameter	Number	Maximum (mg/L)	Minimum (mg/L)	Mean (x) (mg/L)	Median (mg/L)	Standard deviation (σ)	Coef. Variation $C_v = \sigma/x$
Ammonia (as N)	33	0.50	0.028	0.193	0.2	0.114	0.592
TKN (as N)	34	21.4	0.81	2.158	1.13	3.84	1.78
Nitrate + Nitrite (as N)	30	0.371	< 0.005 *	0.208	0.246	0.123	0.591
Total Nitrogen	29	21.4	0.938	2.34	1.41	3.79	1.62
Filt. Reactive Phosphorus	33	0.345	0.005	0.098	0.045	0.107	1.10
Particulate Phosphorus	34	0.421	0.06	0.141	0.126	0.077	0.548
Total Phosphorus	34	0.485	0.097	0.235	0.212	0.112	0.476
Aluminium	34	2.62	0.047	0.816	0.622	0.667	0.818
Arsenic (inorganic)	34	0.01	0.001	0.004	0.004	0.002	0.405
Cadmium	14	0.0064	< 0.0002 *	0.0007	0.0003	0.0016	2.254
Chromium	18	0.019	< 0.005 *	0.010	0.008	0.005	0.491
Copper	30	0.047	< 0.005 *	0.016	0.012	0.010	0.630
Iron	35	2.06	0.106	0.921	0.7	0.549	0.596
Lead	26	0.047	< 0.001 *	0.018	0.015	0.013	0.741
Manganese	34	0.213	0.012	0.072	0.069	0.046	0.635
Mercury	11	0.0003	< 0.0001 *	-	-	-	-
Nickel	32	0.012	< 0.001 *	0.004	0.003	0.003	0.645
Zinc	35	0.239	0.05	0.128	0.130	0.048	0.375
Total suspended solids	31 **	99.0	7.45	33.4	24.0	24.0	0.719

* *Indicates concentration is below the limits of detection for analysis method.*

** *468 sequential samples tested to produce 31 EMC values*

Each event was analysed by exporting data from the HYDSYS database and combining them with in-house testing results from sequential samples. This allowed plotting of flow hydrographs and parameter "pollutographs" which showed the variation in pollutant concentration throughout the events. The spreadsheets and plots were generated in Microsoft® EXCEL 97 and are included on the CD enclosed with this thesis.

Three events are presented in detail in the following sections. The events were chosen as being representative of a small, an intermediate, and a large event and were accompanied by a good set of water quality results. An estimation of pond efficiency on an individual event basis has been given for each event. During small events, where the inflow occurring during the event did not reach the outlet until perhaps the following event, calculating the pond efficiency becomes slightly complicated. This was noted by US EPA (1987), "*...the effluent displaced during a particular event represents, in fact, a volume contributed to by the runoff of some antecedent event.*" Displacement of water previously deposited in the pond (ie. from previous event(s)) therefore needs to be considered when comparing inlet and outlet water quality, which was accomplished in this study by assuming that the flow within the pond was plug flow, a concept illustrated in Figure 6.2. In this case, the water quality monitored at the pond outlet during event *i* would in fact be the water deposited by event *i-6* and, depending on the inflow volume, possibly some of events *i-5*, *i-4* etc.

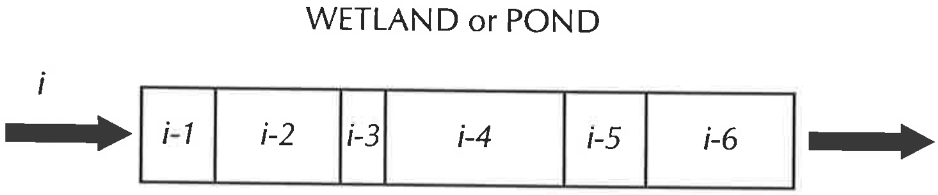


Figure 6.2 **Events passing through pond as pure plug flow. In this case the i^{th} flow is displacing flows deposited by earlier events**

6.1.1 Extreme Event – October 30 1997

The event that occurred on October 30 1997 has been mentioned previously as being a unique event due to its magnitude. Rain gauges located in the Barker Inlet Wetland catchment indicated that close to 100 millimeters of rain fell over a period of approximately 24 hours. Appendix C contains the IFD information for Enfield used to estimate the ARI of each event, this rainfall event was estimated as a 1 in 50 year event. While none of the monitoring equipment malfunctioned as such, the sheer size of the event created its own set of problems, mainly with the flow measurement techniques normally implemented.

The method by which flow was normally calculated at the North Arm East site was discussed in Chapter 5. For this event, as will be shown below, a different approach had to be taken.

The blue scatter points in Figure 6.3 indicate the instantaneous flow during the event at the North Arm East site, calculated by converting velocity data to flow (by multiplying by the cross sectional area of the channel). The backwatered (pink) and non-backwatered (green) rating curves are shown on the graph for comparison, as well as the flow as estimated by the Manning equation (purple x-line). The graph shows that for a depth up to approximately 0.8 metres, the two rating curves, as defined in Chapter 5, are valid. The non-backwatered curve is applicable in the early stages of the event, and the backwater curve in the latter stage. However, past this depth the velocity data becomes very scattered and makes little sense, an indication that the velocity meter ceased to function accurately.

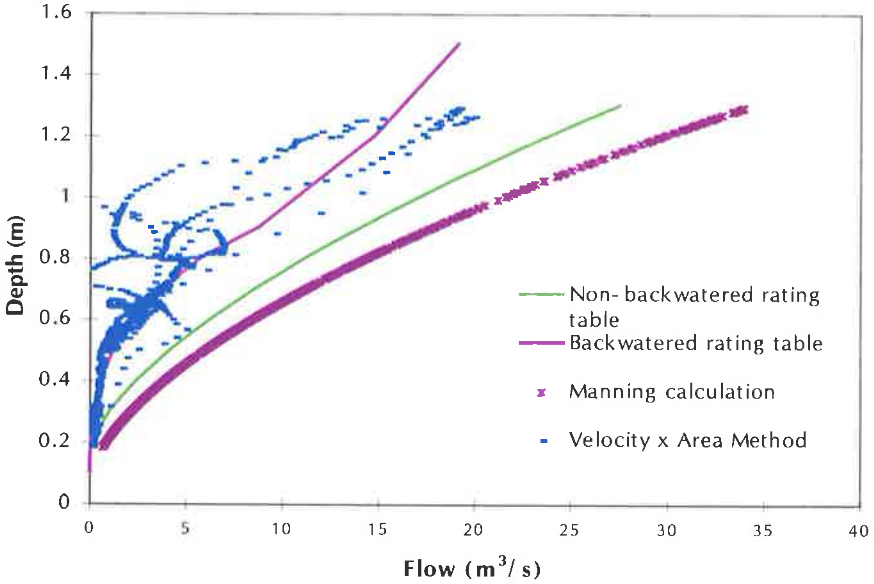


Figure 6.3 **Flow distribution at North Arm East during event on October 30th 1997 compared to backwater rating curve (pink line), non-backwatered rating table (green line) and Mannings Equation calculation's (x-line)**

A flow estimation was made using the rating tables developed by Williams (1997) shown in Figure 6.3. The rating tables are included in Appendix A. The non-backwatered curve was used for the very early stage of the event, and the backwatered curve for the remainder. The calculation yielded an extremely low runoff coefficient for the magnitude of the event and was thus considered incorrect. This was not a surprising result as the rating tables were never calibrated beyond a depth of 0.8 metres, because an event of this size had not been monitored previously.

A second reason for the invalidity of the backwatered rating table for high depths is demonstrated by Photograph 6.1. This shows the trash racks downstream of the North Arm East monitoring station during an event in which they filled to capacity with litter. This was a common occurrence, which led to the development of the backwatered rating curve (discussed in Chapter 5). Photograph 6.1 also shows that under these circumstances, the full trashracks tended to act in a similar manner to a broad crested weir with some flow occurring through the litter, but the majority over the top. The broad crested weir equation (Equation 5.2) was applied to the trashracks to obtain an estimation of the flow over them during the event.

Photograph 6.1 Trashracks downstream of the North Arm East station full of litter during an event. Note that they are acting roughly as a broad crested weir.

Calculating the outflow of this event also differed from the normal calculations. Section 5.2.2 described the process by which outflow was normally calculated using a rating curve developed for the sharp crested weir and rock gabions. The rating curve however, did not take into account backwatering effects from the tailwater pond, which only occurred during the October 30 event. This backwater effect was caused by the flow restriction at the outflow of the Northern Ephemeral Area (NEA). The NEA is separated from the Marine Intertidal Area by a bund and outflow occurs through four culverts under the bund. At the exit of each culvert is a gate, which pivots on a pin and opens and closes with hydraulic pressure. Outflow can only occur through the gates at low tide, when the pressure head on the upstream side of the gate is greater than the downstream side. At high tide during the large event, the gates closed and caused water to accumulate in the NEA, which eventually backed up to the tailwater pond of Pond 4. Although this was not an uncommon occurrence (discussed in Section 5.2.2.2), in this case the tailwater pond filled to capacity and caused a backwatering effect in Pond 4.

The velocity measurements recorded in the three culverts under Salisbury Highway/South Road Connector were examined and proved to be satisfactory with only a small amount of noise as shown in Figure 6.4. The flow was therefore calculated using the velocity data (Equation 5.4) rather than the rating table.

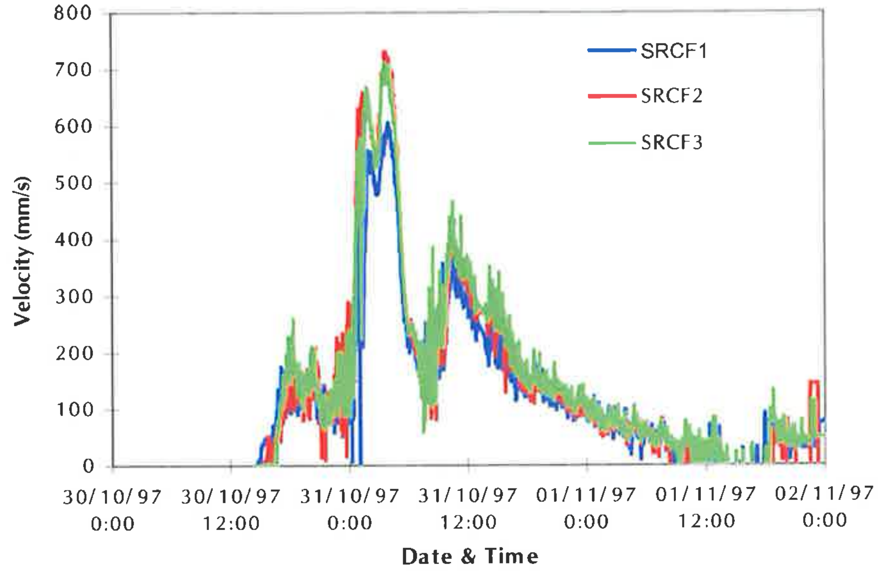


Figure 6.4 **Velocity (mm/s) recorded by the three velocity meters at the outflow of Pond 4 during October 30th event.**

The raw velocity data were smoothed using a moving average before being used to calculate flow. This was the most appropriate data smoothing technique provided by HYDSYS. The resulting inflow and outflow hydrographs and cumulative rainfall are shown in Figure 6.5.

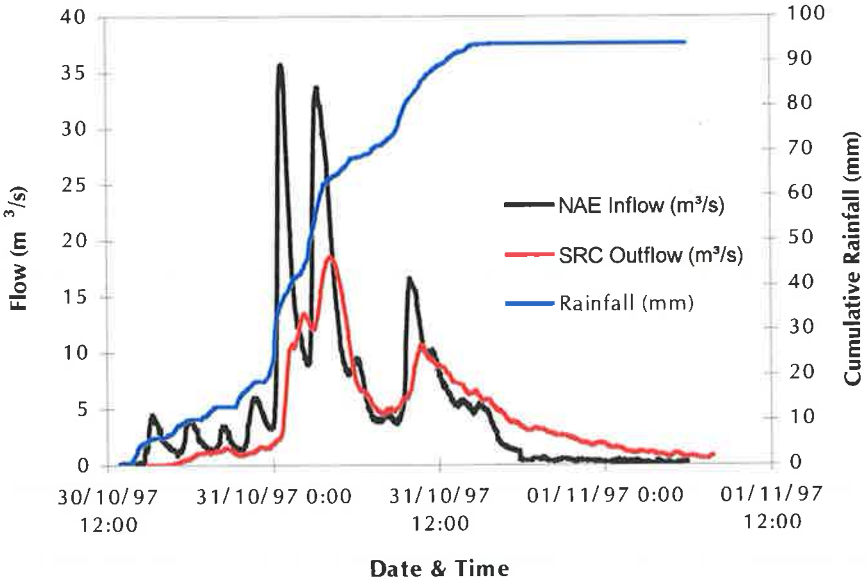


Figure 6.5 **Inflow and outflow hydrograph and cumulative rainfall for 30th - 31st October**

The peak outflow of almost 20 m³/s was the highest recorded during the study period but was still significantly less than the peak inflow rate of 35 m³/s which was also the highest recorded. The total inflow volume for the event was 760 ML.

The measured inflow and outflow turbidity and TSS concentrations for the event are illustrated in Figure 6.6.

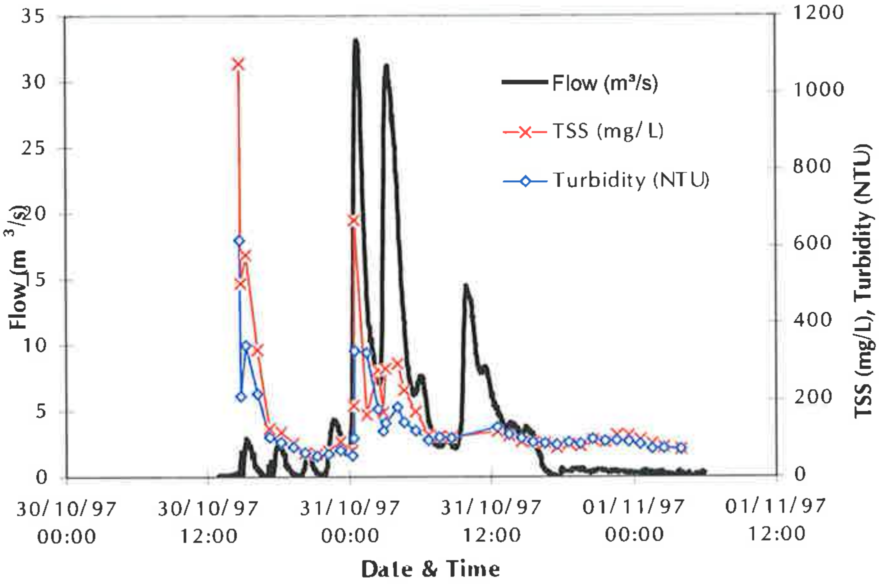


Figure 6.6 **Variation in inflow turbidity and inflow TSS concentration throughout event on 30th October 1997.**

The high initial TSS concentrations recorded at the onset of flow may be indicative of the first flush phenomenon. Duncan (1995a) defines the occurrence of first flush as "*.....when the incremental load exceeds the incremental flow at the start of a runoff event.....*" He goes on to say that first flush produces high concentrations early in the event and a concentration peak which precedes the flow peak. This early concentration peak was demonstrated both in Figure 6.6 and Figure 6.7 for TSS and TP concentrations respectively.

In addition to the early concentration peak, a concentration peak at the tail end of the event was also observed. This was noted on a number of occasions, however the reason for this is not known.

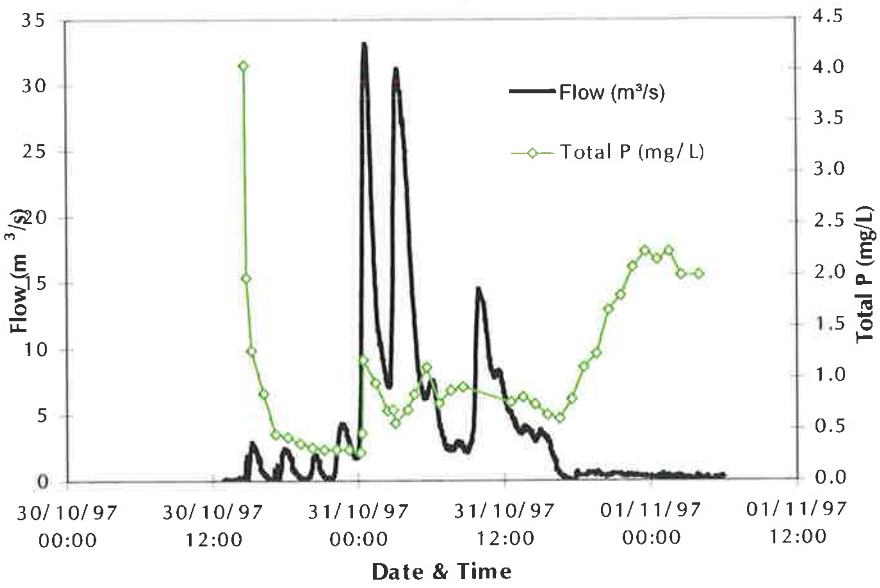


Figure 6.7 **Variation in inflow total phosphorus (TP) concentration throughout event on October 30th 1997.**

The variation in outflow TSS and turbidity concentrations for this large event, shown in Figure 6.6, were quite different to those normally observed, as will be demonstrated later.

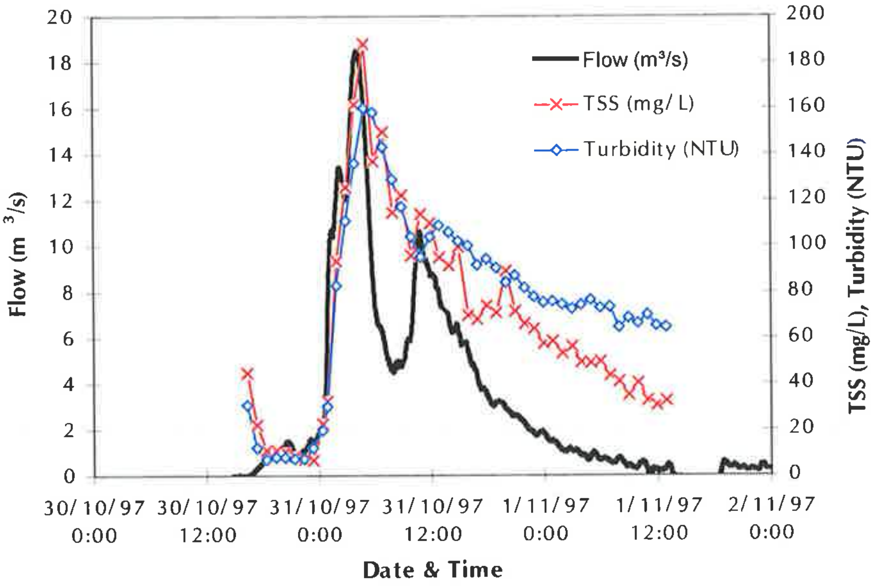


Figure 6.8 **Variation in outflow turbidity and TSS concentration throughout event on October 30th 1997.**

It would be expected that, as a result of displacement of water in the pond during inflow events, outflow hydrograph peaks would not correspond to outflow pollutograph peaks. In this instance however, the water passed through the pond so quickly that the peaks in the flow and pollutant concentration did correspond, not unlike an inflow event. This demonstrates the very short treatment time that was achieved.

Variation in total phosphorus was not as large, except for the high value for the first sample taken, which was possibly due to sample contamination.

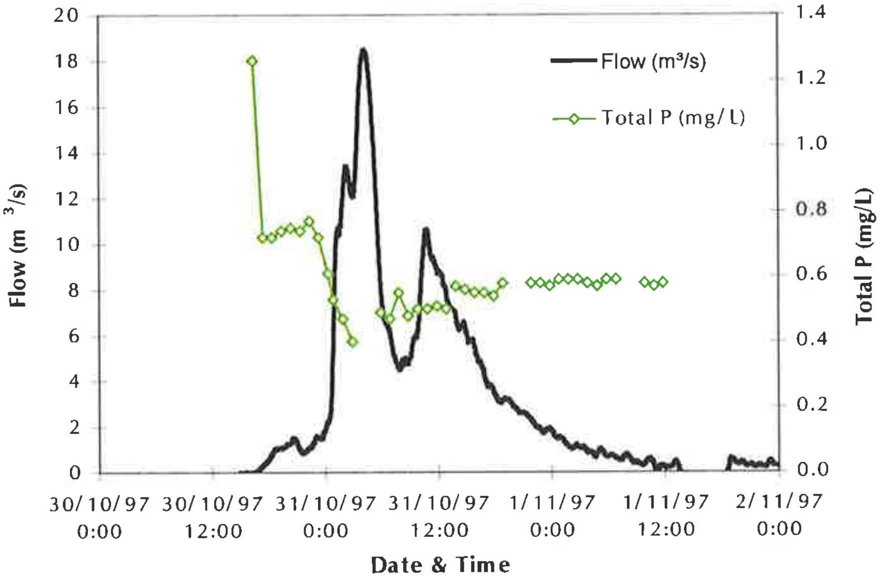


Figure 6.9 **Variation in outflow total phosphorus (TP) concentration throughout event on October 30th 1997.**

The effects of water displacement were not taken into account for this large event as the inflow volume was so large compared to the volume of the pond (more than 10 times its volume). Any water displaced from the pond was insignificant compared to the total event.

The event mean concentration (EMC) data along with an estimate of the removal efficiency obtained for the event are given in Table 6.4.

Reasonably consistent removals were experienced across the range of parameters with the exception of a couple, such as arsenic and mercury which are present only in minute quantities.

Despite the large event, significant removals were still achieved. This is at odds with authors who recommend that large events be diverted to prevent detrimental effects. Somes and Wong (1998), for example, suggest that extreme flows should be diverted, or by-pass the system, in order to prevent scouring and remobilisation of previously deposited sediments. In addition to this not always being possible due to space constraints, these results have shown that it is not always necessary given a sufficiently sized and well designed wetland.

Table 6.4 Inflow and outflow event mean concentrations (EMCs) reduction in concentration for 30th October 1997

Parameter	Inflow EMC (mg/L)	Outflow EMC (mg/L)	Reduction (%)
Ammonia (as N)	0.426	0.248	42
TKN	2.09	1.38	34
Nitrate + Nitrite (as N)	0.244	0.132	46
Total Nitrogen	2.33	1.51	35
Filt. Reactive Phosphorus	0.277	0.276	0.4
Total Phosphorus	0.557	0.415	25
Particulate Phosphorus	0.280	0.139	50
Aluminium	3.91	2.62	59
Arsenic (inorganic)	0.004	0.005	- 25
Cadmium	0.0008	0.0003	63
Chromium	0.018	0.011	39
Copper	0.058	0.026	55
Iron	3.31	1.77	47
Lead	0.132	0.042	68
Manganese	0.078	0.046	41
Mercury	0.0001	0.0001	0
Nickel	0.006	0.003	50
Zinc	0.404	0.164	59
Total Suspended Solids	185.4	99	47

6.1.2 Intermediate Event – 11th June 1998

A storm that occurred on the 11th June 1998 was chosen to represent an event of intermediate size. Rainfall of 21 millimetres produced a runoff volume of 98 megalitres, with a volumetric runoff coefficient of about 22 percent. The rainfall occurred over a period of approximately 24 hours and was estimated as a less than 1-year ARI rainfall event (see Appendix C).

The inflow and outflow hydrographs for the event are shown in Figure 6.10. The inflow hydrograph (black line) consisted of four peaks which were each sampled individually and then combined to give an event mean concentration for the whole event. The peak outflow rate (red line) was significantly reduced by the wetland and a small amount of outflow continued for more than a day after the rainfall occurred.

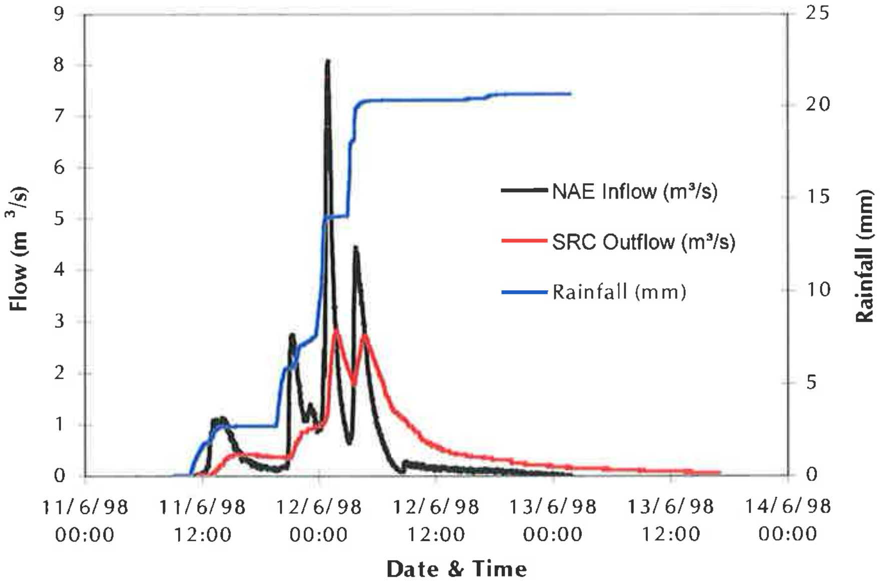


Figure 6.10 Inflow and outflow hydrographs and cumulative rainfall for 11th - 13th June 1998.

Figure 6.11 shows the variation in inflow turbidity, and TSS and TDS concentrations throughout the event.

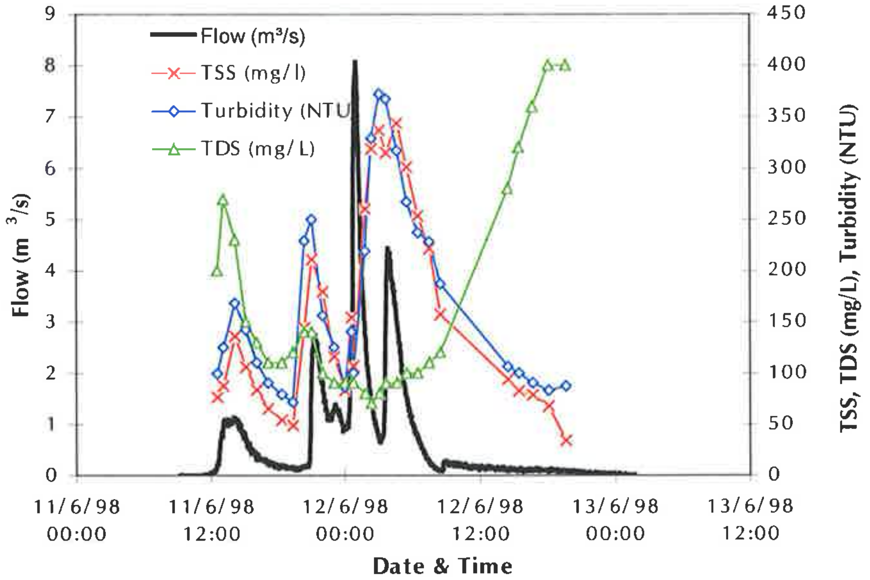


Figure 6.11 Variation in inflow turbidity and inflow total suspended solids (TSS) and total dissolved solids (TDS) concentrations throughout event on 11th June 1998.

The behaviour displayed in the above graph was reasonably typical for an event of this size. The turbidity and TSS concentrations were initially low, and once again increased significantly with the first peak in the hydrograph. The rise and fall in the pollutographs roughly followed the peaks in the inflow hydrograph as would be expected due to the process of washoff. Washoff is the process by which previously deposited material is removed from catchment surfaces by rainfall and runoff, and is incorporated into the flow (Duncan, 1995a). Therefore it follows that as the runoff rate increases and has more energy, the amount of material removed from surfaces will increase as will the flows ability to maintain particles in suspension. A close relationship between TSS and turbidity is evident from the graph, which is investigated further in Section 6.4.2.

The TDS graph displays the inverse behaviour to TSS. The salinity was commonly high preceding an event and dropped significantly following the first storm peak due to the flushing effect by fresh rain water with a typically low salinity (<100 mg/L). The rapid increase in salinity following the event was also a common phenomenon, although the reason for this is not clear.

The variation in inflow total phosphorus concentration is shown in Figure 6.12 determined by in-house testing of sequential samples. Total phosphorus displayed similar behaviour to that of TSS, although the highest concentrations were recorded at the beginning and the end of the event. The initial peak may be due to a first flush type behaviour, however the peak at the end of the event, which also occurred in the large event, is not so easily explained.

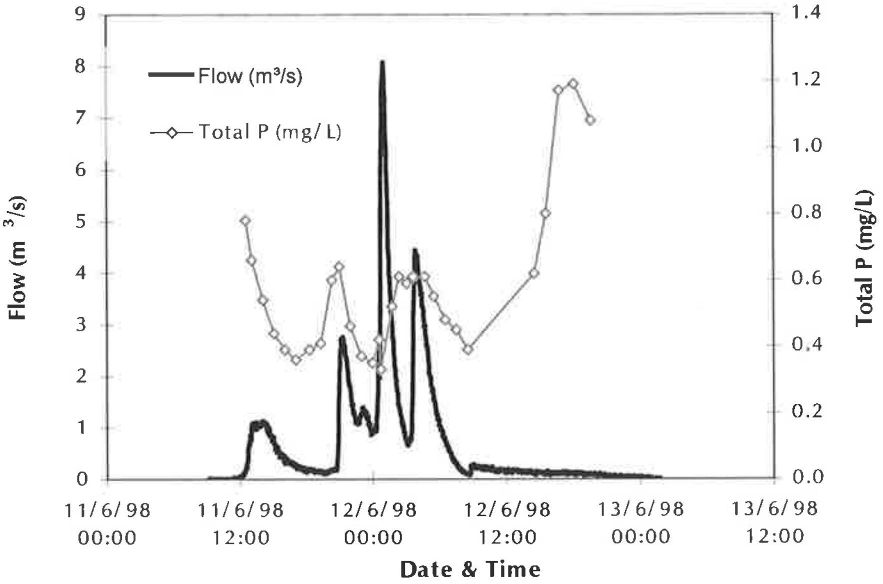


Figure 6.12 Variation in inflow total phosphorus (TP) concentration throughout event on 11th June 1998.

The outflow turbidity and TSS concentrations for the same event are shown in Figure 6.13 along with the outflow hydrograph.

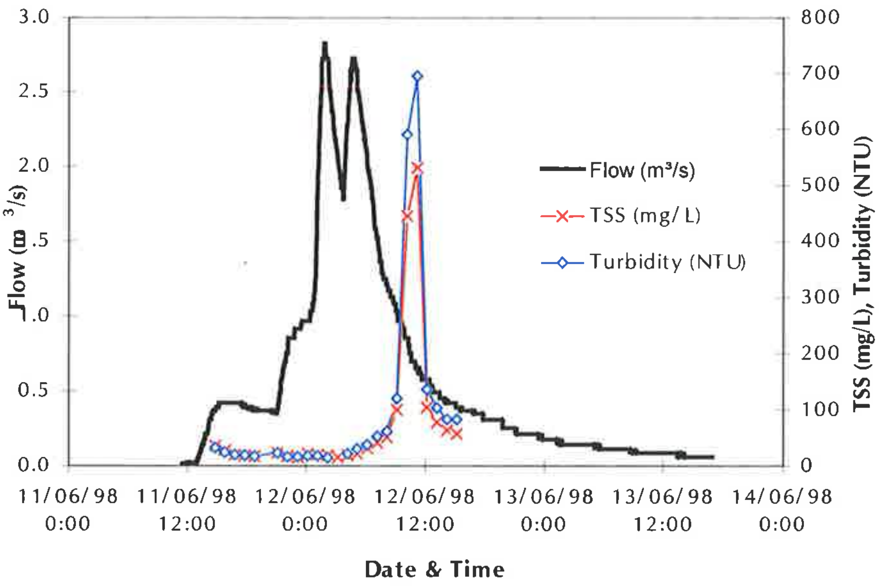


Figure 6.13 Variation in outflow turbidity and outflow total suspended solids (TSS) concentration throughout event on 11th June 1998.

The same close correlation between TSS and turbidity is evident from this graph as was displayed in Figure 6.11. The graph also shows that the first portion of outflow, which is the pond water deposited by the preceding event, is very low in both turbidity and suspended solids. Following the peak in the outflow hydrograph, a single spike in concentration moved through the pond, with a peak significantly higher than the peak inflow concentration. This increase in suspended sediment concentration may have been a result of resuspension or scouring near the outlet. After the spike passed, the turbidity and TSS returned to low concentration values again. A similar trend was observed in the outflow total phosphorus concentrations given in Figure 6.14.

This behaviour resembles the findings of Shatwell and Cordery (1998) in that part of the pollutant load seemed to move through the pond as a discrete parcel or plug. This may indicate that during this particular event the flow regime was characterised by limited mixing or near plug flow. Following the spike, the TP concentration returned to its background concentration of about 0.25 mg/L (c.f. Figure 6.20, total phosphorus concentration during small event).

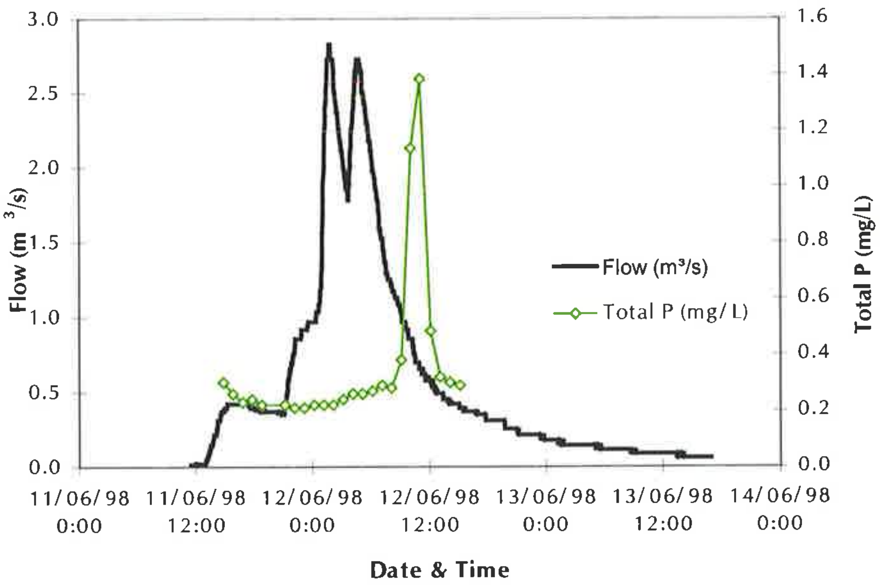


Figure 6.14 Variation in outflow total phosphorus (TP) concentration throughout event on 11th June 1998.

Figure 6.10 showed that this inflow event consisted of four distinct peaks, each of which were sampled with a separate composite sample. Figure 6.15 shows how the inflow pollutant concentrations varied across each peak. The left hand side of Figure 6.15 plots the variation in nutrient species, which all had higher concentrations for the first and last storm peak. Metal concentrations, shown on the right hand side, displayed the opposite behaviour and increased with increasing flow, as did the TSS concentration. Although this may be nothing more than coincidence, it is more likely that the graphs are demonstrating the common sources or transport mechanisms of pollutants in stormwater. Nitrogen and phosphorus follow the same trend suggesting that they may have a common source while the metal species display the same behaviour as TSS. This suggests that a large portion of the metal loads are transported attached to particulate matter.

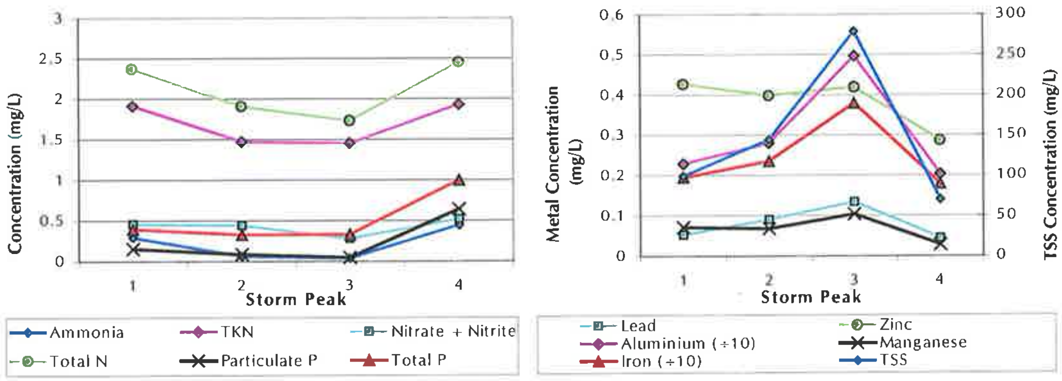


Figure 6.15 Variation in inflow nutrient concentration (left), and metal and TSS concentration (right) with each storm peak on 11th June 1998.

The single set of event mean concentration values for each of the parameters tested are given in Table 6.5 along with an estimation of the reduction in pollutant concentration that occurred during the event.

Table 6.5 Inflow and outflow event mean concentrations (EMCs) and concentration reduction for 11th June 1998

Parameter	Inflow EMC (mg/L)	Outflow EMC (mg/L)	Reduction (%)
Ammonia (as N)	0.097	0.259	- 167
TKN (as N)	1.54	1.14	26
Nitrate + Nitrite (as N)	0.345	0.287	17
Organic Nitrogen	1.44	0.881	39
Total Nitrogen	1.89	1.43	24
Filt. Reactive Phosphorus	0.097	0.054	44
Total Phosphorus	0.369	0.186	50
Particulate Phosphorus	0.299	0.132	56
Aluminium	4.06	0.959	76
Arsenic (inorganic)	0.005	0.005	0
Cadmium	0.0007	< 0.0002 *	> 71
Chromium	0.008	0.005	38
Copper	0.019	0.011	42
Iron	3.17	1.14	64
Lead	**0.110**	**0.027**	75
Manganese	0.089	0.074	17
Mercury	< 0.0001 *	0.0001	-
Nickel	0.005	0.003	40
Zinc	0.410	0.134	67
Total Suspended Solids	**255.6**	**60.5**	76

* *Indicates concentration is below the limits of detection for analysis method*

The removals calculated for this event (Table 6.5, far right column) support the statement of Duncan (1995a) that "Nitrogen exhibits by far the most complex behaviour in storage, since it is by no means a conservative parameter." Although ammonia has been found to decrease in storage (Martin, 1988), it more commonly increases as found by Randall *et al.* (1982) and Holler (1989). During the June event, the ammonia increased by 167 percent, which may have been due to a number of factors, including ammonification, or mineralisation, which is the conversion of organic-N to ammonia (see Chapter 3). This is supported by the fact that the organic nitrogen EMC decreased by close to 40 percent.

Nitrate + nitrite removal was fairly low as would be expected as they are highly soluble and have very little affinity for sorbing to particulate surfaces (Randall *et al.*, 1982) reinforcing the importance of sedimentation in pollutant removal. Total phosphorus was reduced by 50 percent, with a slightly larger proportion of particulate phosphorus removed than the dissolved fraction.

While nutrient removal was variable, metal and TSS removal was more encouraging. Laboratory tests conducted by Randall *et al.* (1982) to determine the efficacy of sedimentation in the removal of pollutants from urban stormwater found that TSS and lead are removed the most successfully by sedimentation. This indicates the strong affinity of lead for particulates, which is also reflected in Table 6.1 by the significant removals of lead and TSS (highlighted). Other metals that exhibited high removals were aluminium, cadmium, iron, and zinc. With the exception of cadmium, these (along with lead) were the metals present in the highest concentrations.

6.1.3 Small Event – 24th June 1998

A small runoff event of 22 megalitres occurred on 24th June 1998, following a 4.5 millimetre rainfall event. The event was estimated as having an Average Recurrence Interval (ARI) of less than 1 year (see Appendix C), the runoff coefficient was 23 percent. The rain produced a single peaked inflow and outflow hydrograph, shown in Figure 6.16.

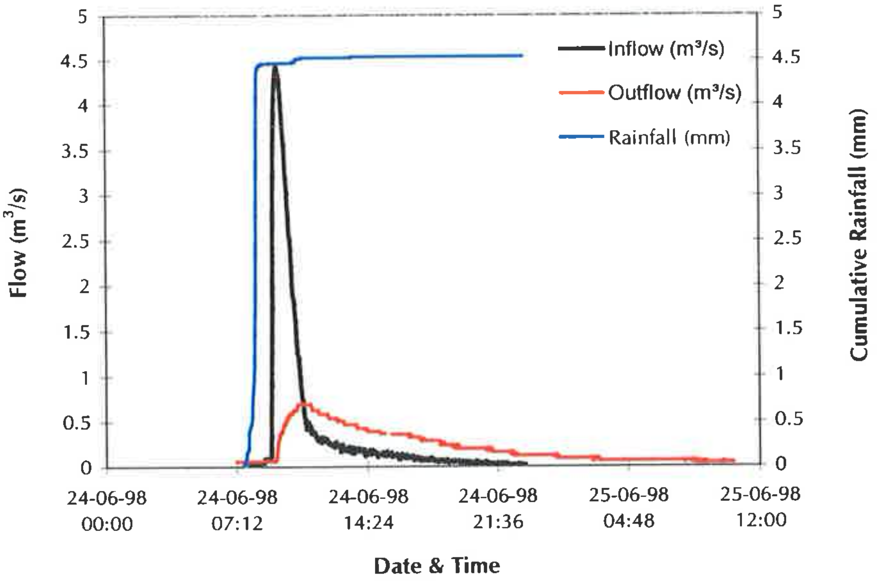


Figure 6.16 Inflow and outflow hydrographs and cumulative rainfall for 24th – 25th June 1998.

The turbidity, TSS and TDS concentrations monitored at the inlet during the event are shown in Figure 6.17.

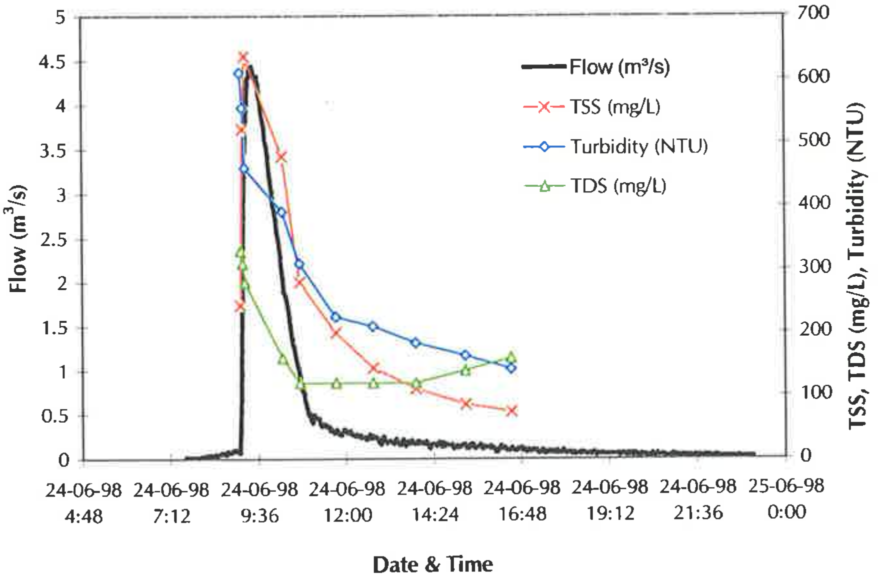


Figure 6.17 Variation in inflow turbidity and inflow total suspended solids (TSS) and total dissolved solids (TDS) concentrations throughout event on 24th June 1998.

The peaks in turbidity and TSS generally followed the peak in the hydrograph, a trend which was observed during many events throughout the study period. The TDS concentrations displayed different behaviour, beginning high and dropping off as the event came through. Total phosphorus displayed the same behaviour as turbidity and TSS as shown in Figure 6.18.

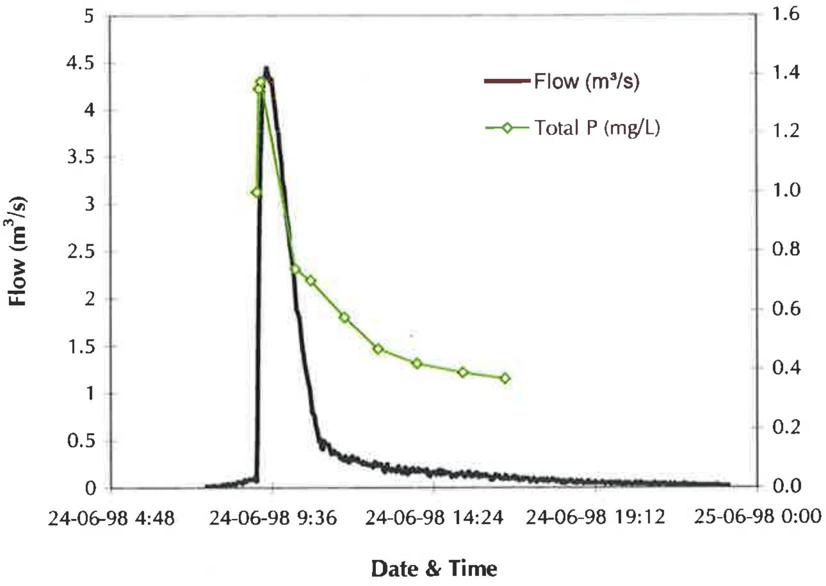


Figure 6.18 Variation in inflow total phosphorus concentration during 24th June 1998.

The outlet turbidity and TSS concentrations, shown in Figure 6.19 were much more constant than at the inlet. Concentrations were also very low, lower than any concentration recorded at the inlet, indicating a significant amount of treatment within the pond. It should be noted that the outflow concentrations on the pollutographs (Figure 6.19 and Figure 6.20) cannot be directly compared to the incoming concentrations due to water displacement as described in Section 6.1.

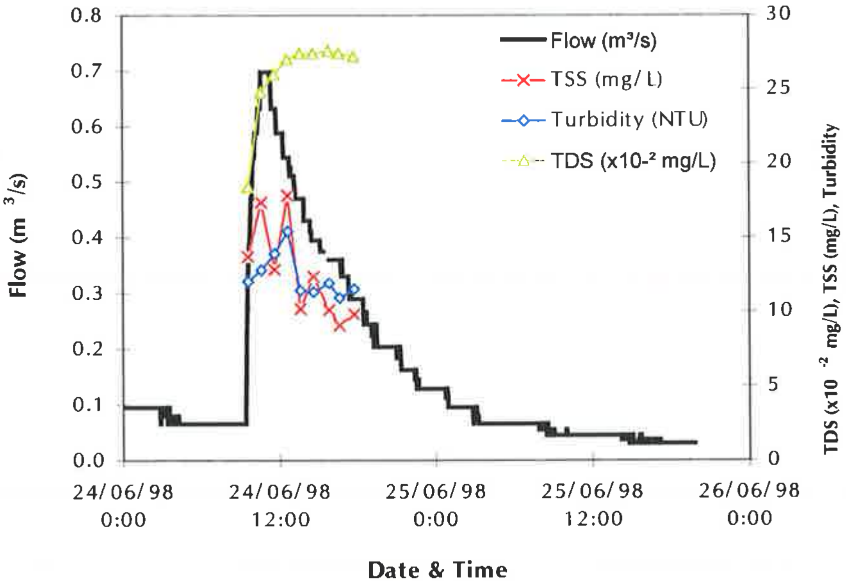


Figure 6.19 Variation in outflow turbidity and outflow TSS and TDS concentrations during 24th June 1998.

The TDS concentration was also relatively constant, however an order of magnitude higher than the incoming concentrations normally observed. This may indicate either evaporation and subsequent concentration of salts, or, more likely, groundwater interaction. The groundwater under the pond is significantly more saline than the surface water and may have been contributing to the salinity of the pond via an osmotic effect.

The total phosphorus concentration of the outflowing water, shown in Figure 6.20, remained reasonably constant. This provides an indication of the background concentration of total phosphorus within the pond water (0.2 - 0.25 mg/L) which is likely to be achieved given sufficient treatment time. Table 6.3 showed that the mean and median TP concentrations for the study period were also close to this value.

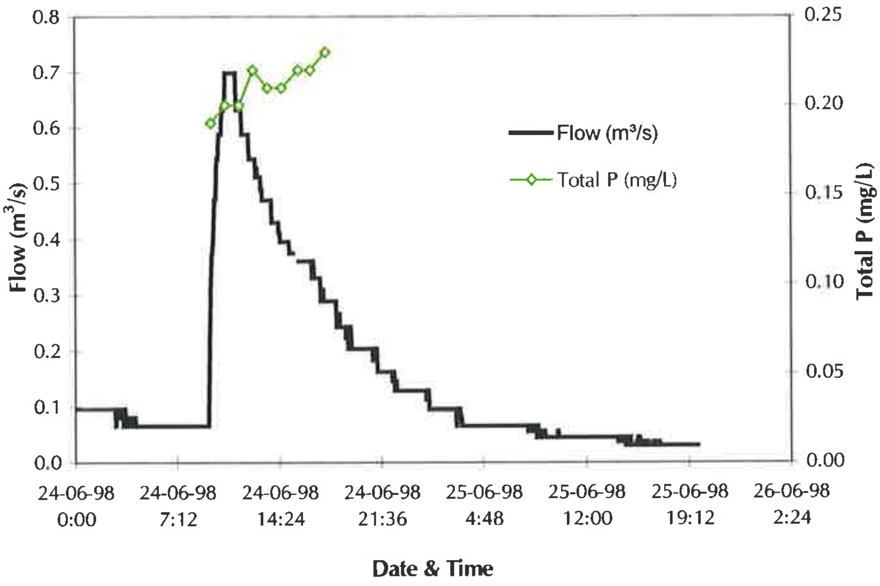


Figure 6.20 Variation in outflow total phosphorus concentration throughout event on 24th June1998.

The inflow and outflow event mean concentrations for the June 24th event are given in Table 6.6. Rather than using the outlet water quality results monitored during the 24th June, results of a successive event were used. As mentioned previously, the assumption of plug flow was made and the concept illustrated in Figure 6.2 utilised. Since the pond volume is approximately 70 ML, the next 48 ML (70 - event volume) of outflow following the event was disregarded and the outflow water quality monitored during the event following that (5th July 1998) was used to calculate the removal efficiency.

The inflow and outflow EMCs and removal efficiencies are summarised in Table 6.6.

Table 6.6 Inflow and outflow event mean concentration (EMC) data, and estimated pollutant reduction for 24th June 1998.

Parameter	Inflow EMC (mg/L)	Outflow EMC (mg/L)	Reduction %
Ammonia (as N)	0.016	0.311	-1843 (!)
TKN (as N)	2.2	1.23	44
Nitrate + Nitrite (as N)	0.312	0.323	-3.5
Organic Nitrogen	2.18	0.92	58
Total Nitrogen	2.51	1.55	38
Filt. Reactive Phosphorus	0.063	0.032	49
Particulate Phosphorus	0.427	0.092	78
Total Phosphorus	0.490	0.125	74
Aluminium	9.33	0.63	93.2
Arsenic (inorganic)	0.003	0.002	33
Cadmium	< 0.0002 *	< 0.0002 *	-
Chromium	0.016	0.009	44
Copper	0.049	0.014	71
Iron	7.94	0.675	91
Lead	**0.178**	**< 0.001 ***	**> 99**
Manganese	0.148	0.128	13.5
Mercury	0.0001	< 0.0001 *	-
Nickel	0.01	0.004	60
Zinc	0.771	0.115	85
Total Suspended Solids	**385.9**	**14.9**	**96**

* *Indicates concentration is below the limits of detection for analysis method*

Once again, the ammonia event mean concentration was higher at the outlet, this time by a significant amount. In this instance, the conversion of organic nitrogen to ammonia is even more likely, as the organic nitrogen decreased by 58 percent while the ammonia increased by more than 1000 percent.

The removal of TSS and pollutants typically particle bound, such as metals and particulate phosphorus, was very high as would be expected for a small event.

6.2 Bacteriological Water Quality

A series of water samples were taken over a short period (May 19th to May 22nd 1998) to determine the bacteriological quality of the water in Pond 4 compared to the water flowing into it. To accurately determine the bacterial counts in the water, sterile sample bottles were used and samples delivered to the laboratory within 6 hours. Sampling therefore had to be undertaken by hand, and thus only a limited number of samples were analysed.

Samples were analysed for faecal coliforms (FC) and *e-coli*. The results for the North Arm East site are shown in Figure 6.21 while the bacterial counts at the outlet to Pond 4 are shown in Figure 6.22

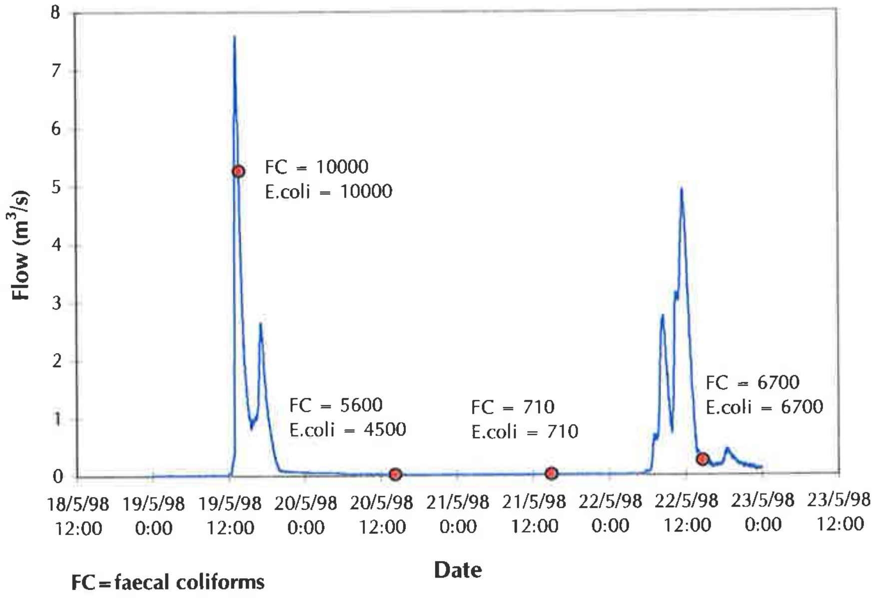


Figure 6.21 Biological water quality in North Arm East Drain during sampling period

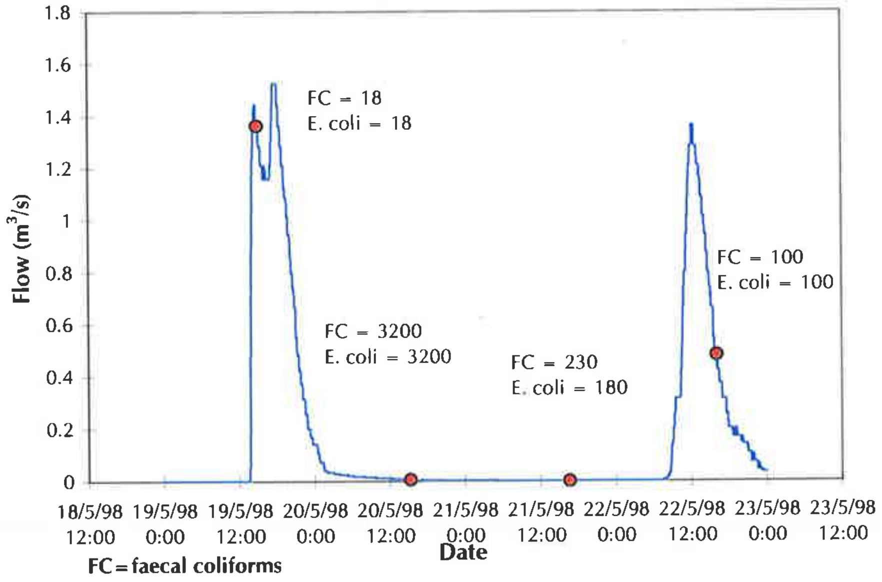


Figure 6.22 Biological water quality at South Road Connector station during sampling period.

Comparing Figure 6.22 to Figure 6.21 indicates that the number of faecal coliforms and *E-coli* are lower at the pond outlet than the inlet. The reduction in bacteria and pathogens in wetlands and ponds can be a result of natural attrition or ultra violet disinfection. Although the method by which bacteria was reduced has not been investigated in this study, it is clear that a reduction did occur.

6.3 Continuous Real-Time Monitoring

The monitoring network was described in detail in Chapter 5, and discussed the data collection network that was maintained throughout the study period. This section presents some of the results obtained from the continuous real-time monitoring probes that were installed at each of the sites. The parameters monitored at the sites included all or some of the following: stage height, flow velocity, turbidity, electrical conductivity, water temperature, and pH.

The use of the stage height and velocity data has already been shown in the calculation of event flow rates and volumes. The data sets obtained for the other parameters monitored are discussed briefly below.

The continuous real time monitoring data set for each parameter is given in Appendix D.

6.3.1 Electrical Conductivity

Electrical conductivity (EC) at the North Arm East site typically displayed behaviour as indicated by the small section of the data set shown in Figure 6.23.

The EC during base flow periods was commonly around the mid to high hundreds and dropped off markedly as an event came through and flushed the channel with fresh water. Following each event the EC climbed again. Although an increase in EC in a shallow body of water could be attributed to evaporation, this was clearly not the case at the North Arm East site as Figure 6.23 illustrates. The EC increases steeply following the event, and returns to the high EC value much too rapidly to be as a result of evaporation. The exact cause of this behaviour is not known. Although there is a possibility that the phenomenon is caused by groundwater interaction, this is unlikely as the drain is concrete lined.

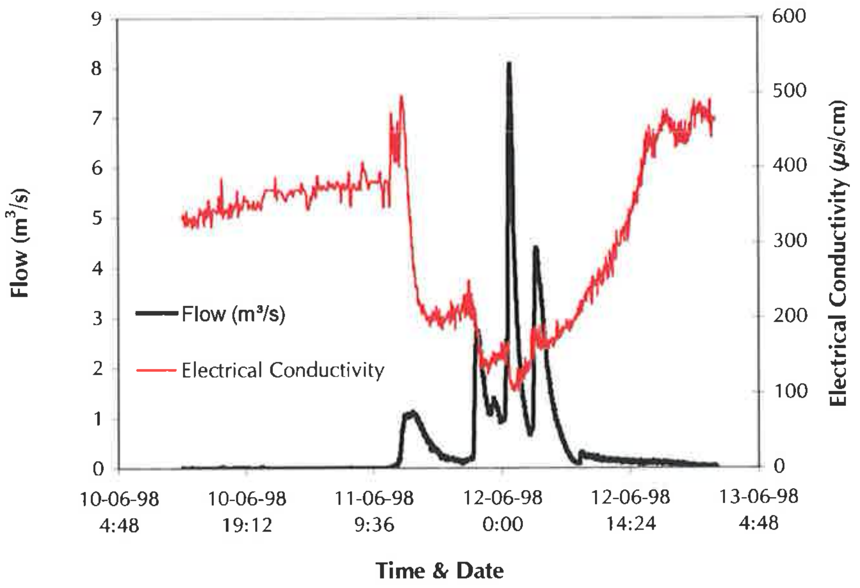


Figure 6.23 Typical electrical conductivity behaviour displayed during an event as recorded by EC probe at North Arm East station.

Electrical conductivity can be useful as a tracer, and could have been used to detect the point at which the inflow reached the outlet, indicated by the passing of a 'slug' of very fresh water. However, the EC meter at the South Road Connector station did not function correctly throughout the study period and was of little use.

6.3.2 Turbidity

Turbidity has already been examined to some extent in the event water quality results (Section 6.1). Monitoring of turbidity can be useful if a reliable relationship between turbidity and suspended solids can be obtained. This will be discussed further in Section 6.4.2, however to put it briefly, while turbidity is an easy parameter to measure on a continuous basis, TSS is not. If a reliable TSS/turbidity relationship were available, there would be no need to collect water samples and test them for TSS, turbidity could simply be monitored and converted to TSS using the appropriate relationship.

Figure 6.24 shows a small portion of the continuous turbidity data set recorded during an event. The red line also compares the turbidity probe's readings to those obtained by laboratory testing of the sequential samples taken during the event. Although the blue line and the red line in Figure 6.24 follow the same shape, the continuous turbidity probe clearly needs a calibration factor applied to it to make the readings accurate.

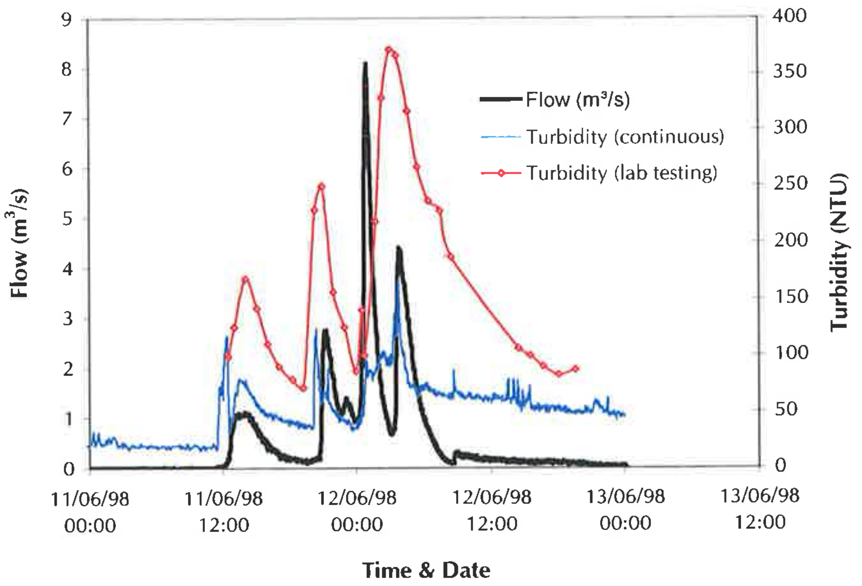


Figure 6.24 **A section of the continuous turbidity data set at the North Arm East Station recorded during an event (compared to lab results)**

By applying a calibration factor of two to the continuous readings, as shown in Figure 6.25 the two lines become much closer.

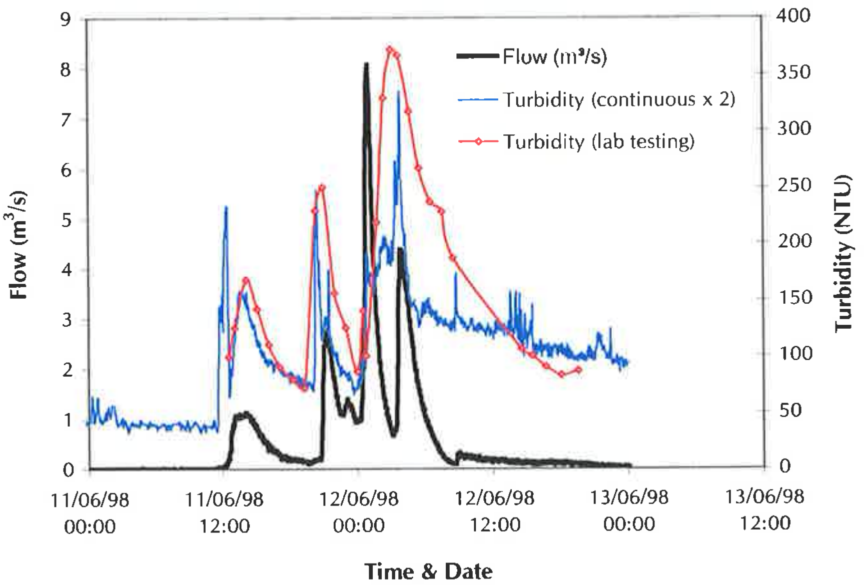


Figure 6.25 **Comparison between laboratory tested and continuously monitored turbidity with a calibration factor of two applied.**

Post-archive data smoothing and calibration can be done in the HYDSYS database using the data managers workbench application. A multiplication factor of two could be applied to the turbidity data set to bring it into line with the observed measurements.

6.3.3 pH

The pH meter at the North Arm East site ceased to work early in the study period and was removed. A small section of the record from 1995 is given in Figure 6.26. The ANZECC Australian Water Quality Guidelines (ANZECC, 1992) recommend that the pH of freshwater should not be permitted to vary beyond 6.5 to 9. This range is shown by the red lines on Figure 6.26, and as can be seen, the pH at the NAE site is consistently above the recommended levels.

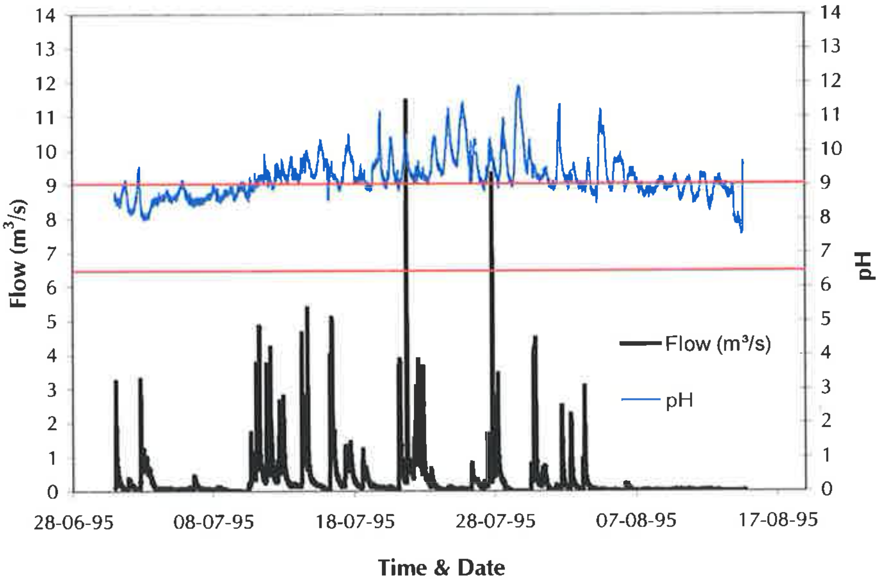


Figure 6.26 A section of the continuous pH data set recorded at the North Arm East station (ANZECC Guidelines recommended pH range indicated by red lines)

6.3.4 Temperature

A section of the temperature records at the NAE and SRC stations is given in Figure 6.27. It shows that the inflow water temperature (blue-line) undergoes larger daily fluctuations than the in pond water (red-line). This is what would be expected as the pond is a much bigger and deeper water body than the inflow channel and thus takes longer to heat up and cool down.

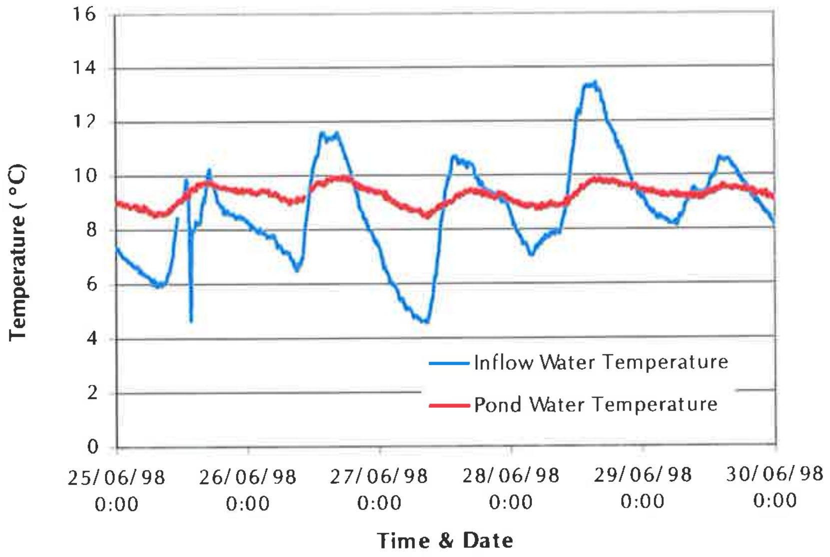


Figure 6.27 **Section of the inflow and pond water temperature data set recorded at the North Arm East and South Road Connector stations.**

6.4 Data Estimation

In order to calculate annual loads entering and exiting the wetland, it was necessary to have flow and water quality data for every event recorded over a 12-month period. A 12-month period was chosen that had the most complete data set, however some gaps still existed that needed to be filled in both at the North Arm East and South Road Connector stations.

A number of "modelling" techniques are available to estimate missing data. In this study a number of alternatives were investigated; event volume/load relationships, rainfall/load relationships, parameter correlations, and average annual concentrations. These alternatives are discussed below.

6.4.1 Data Estimation Using Volume/Load and Rainfall/Load Relationships

Parameter loads were plotted against volume for each event available at both the stations. A log-log scale was used due to the large variation in event volumes and loads.

A line of best fit, either linear or power, was fitted to each of the data sets and the r^2 determined. Figure 6.28 and Figure 6.29 are examples of the relationships found at the North Arm East Station for nitrate + nitrite and zinc respectively.

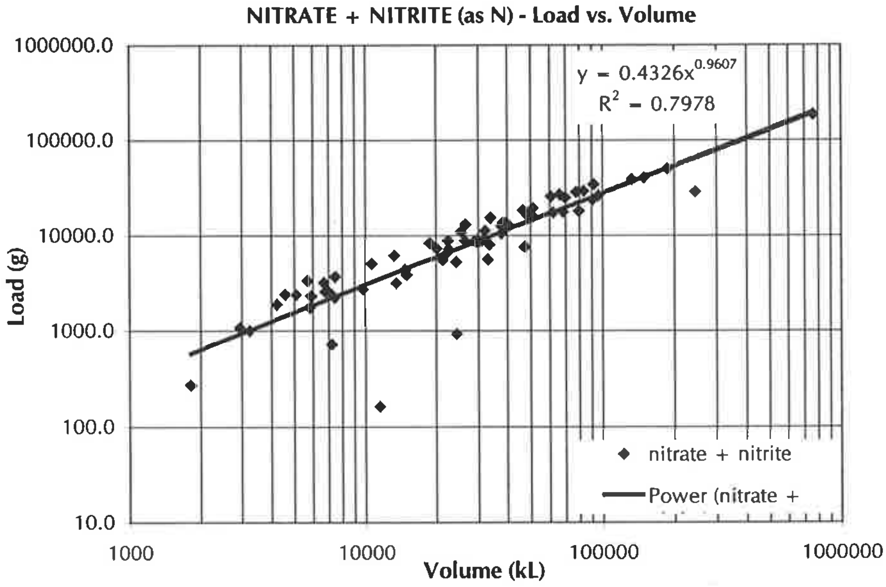


Figure 6.28 Nitrate + nitrite load vs. event volume relationship for North Arm East inflow station

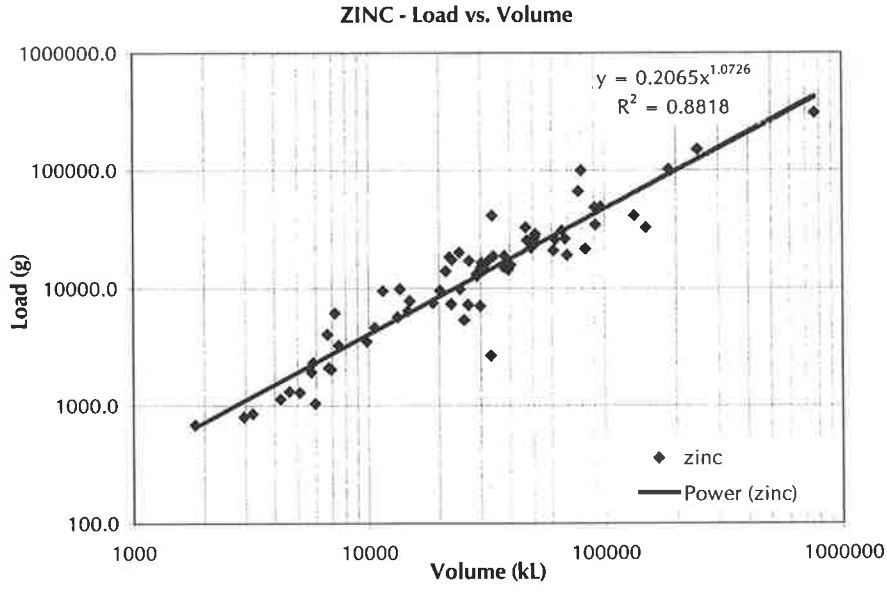


Figure 6.29 Zinc load vs. event volume relationship for North Arm East inflow station.

These were two of the best fits obtained, both modelled the large event satisfactorily. Similar relationships were developed for the South Road Connector outflow station, examples are given in Figure 6.30 and Figure 6.31 for TKN and zinc respectively.

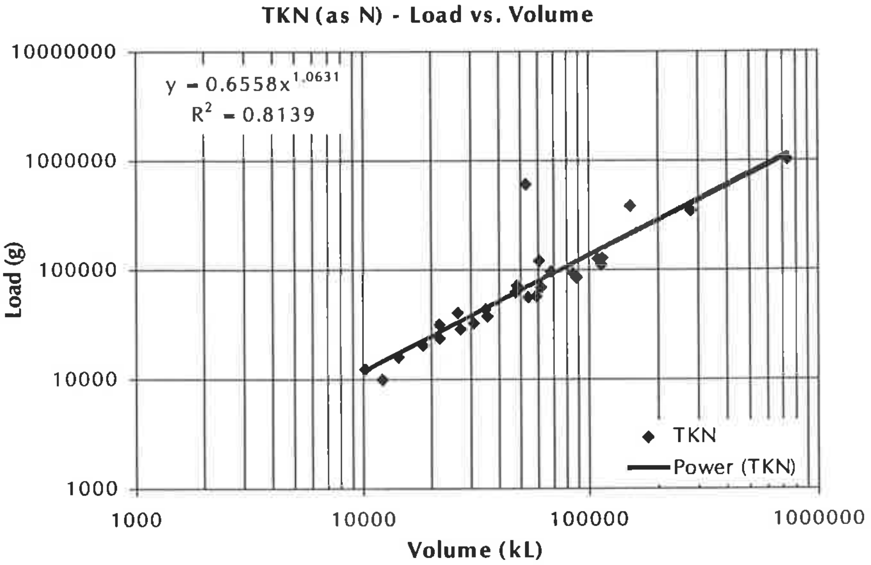


Figure 6.30 Total Kjeldahl nitrogen (TKN) load vs. event volume relationship for South Road Connector outflow station.

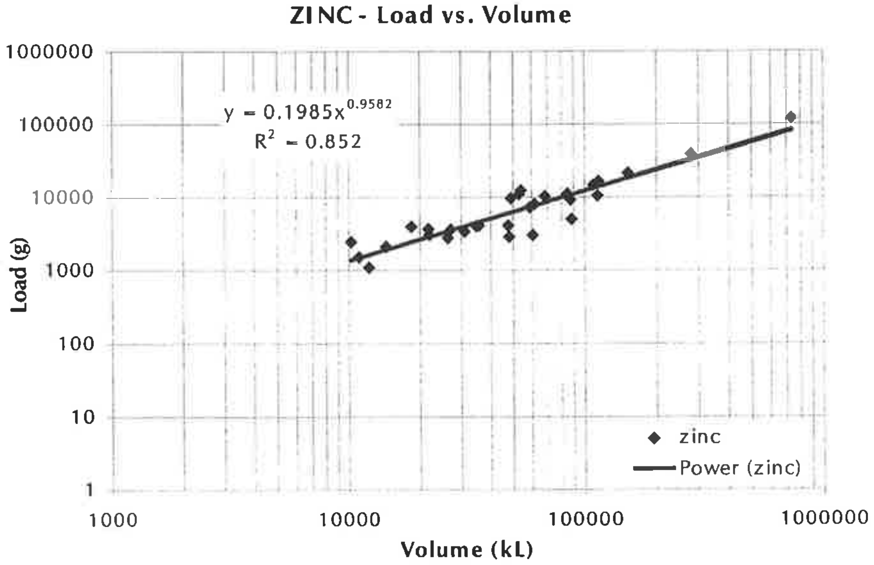


Figure 6.31 Zinc load vs. event volume relationship for South Road Connector outflow station

These two examples again show that the large event was modelled satisfactorily. The entire set of load vs. event volume plots and relationships are given in Appendix E for each station. The determined relationships with their corresponding r^2 values are summarised in Table 6.7.

Table 6.7 **Event volume/load relationships for the water quality parameters at the North Arm East Inflow and South Road Connector Outflow stations, including r^2 values.**

	North Arm East		South Road Connector	
Parameter	**Equation**	**r^2**	**Equation**	**r^2**
Ammonia (as N)	$0.3086V^{0.9079}$	0.55	$0.0211V^{1.1913}$	0.78
Nitrate + nitrite (as N)	$0.4326V^{0.9607}$	0.80	$2.7973V^{0.7445}$	0.46
TKN (as N)	$2.0035V^{0.9795}$	0.86	$0.6536V^{1.0634}$	0.85
Filt. reactive phosphorus (as P)	$1.9471V^{0.7028}$	0.64	$0.0012V^{1.3413}$	0.62
Total phosphorus	$1.0456V^{0.9053}$	0.76	$0.0125V^{1.2565}$	0.90
Aluminium (total)	$0.3686V^{1.1924}$	0.83	$0.598V^{1.0136}$	0.63
Arsenic (inorganic)	$0.0009V^{1.1384}$	0.88	$0.0008V^{1.1423}$	0.93
Cadmium (total)	$0.0005V^{1.0488}$	0.86	$0.0004V^{0.9884}$	0.64
Chromium (total)	$0.0033V^{1.1306}$	0.85	$0.0395V^{0.8586}$	0.85
Copper (total)	$0.0081V^{1.1513}$	0.87	$0.0745V^{0.8482}$	0.74
Iron (total)	$0.445V^{1.1683}$	0.84	$1.2375V^{0.9632}$	0.72
Lead (total)	$0.0062V^{1.2787}$	0.83	$0.0122V^{1.0163}$	0.61
Manganese (total)	$0.0434V^{1.0518}$	0.82	$0.4031V^{0.8329}$	0.68
Mercury (total)	$0.0004V^{0.9106}$	0.80	$0.0012V^{0.7984}$	0.93
Nickel (total)	$0.0029V^{1.0621}$	0.89	$0.0017V^{1.0611}$	0.83
Zinc (total)	$0.2076V^{1.0721}$	0.88	$0.1985V^{0.9582}$	0.85
Total suspended solids	$18.437V^{1.2031}$	0.81	$7.0356V^{1.121}$	0.71

Although this modelling technique appears to offer an accurate method of predicting pollutant loads based on event volume, some caution needs to be exercised as noted by Millar (1996). The mass of pollutant exported during an event is equal to the product of runoff volume and event mean concentration. Millar stated that the concern with the mass export-runoff volume modelling approach is that the runoff volume appears in both the dependent and independent variables of the regression analysis (x and y axes of Figure 6.28 - Figure 6.31) and therefore spurious correlation must be considered. Spurious correlation was defined by Reed (1921) and Rowe (1963) as follows:

"Though no correlation exists between any two of a set of variables there will still exist correlation between any two functions of these variables whenever these functions have any of the variables in common. The correlation existing under these conditions will be called spurious correlation."

Millar goes on to say that relations that result from spurious correlation are not wrong *per se*. The main problem is that they usually instil a false sense of confidence. A high coefficient of determination, or r^2 value, is usually an indication of a reliable model or good model calibration. However, when the r^2 has been the result of spurious correlation, the confidence is not justified.

Because pollutant mass is simply the product of concentration and runoff volume, when mass is plotted against runoff volume, the same regression equation is obtained as if concentration had been plotted against runoff except the regression exponent is increased by one. The effect however, is to significantly improve the r^2 value. The effect is demonstrated by Figure 6.32. If this is compared to Figure 6.31, it can be seen that the regression equation is the same in both cases except for the exponent. The plot of concentration versus volume has an associated r^2 value of just 1.8 percent.

Despite the spurious correlation involved in using a mass export-runoff volume modelling approach, it does offer a relatively accurate method of predicting pollutant mass export given only event runoff volume. This is demonstrated in the following section which validates the modelling approach.

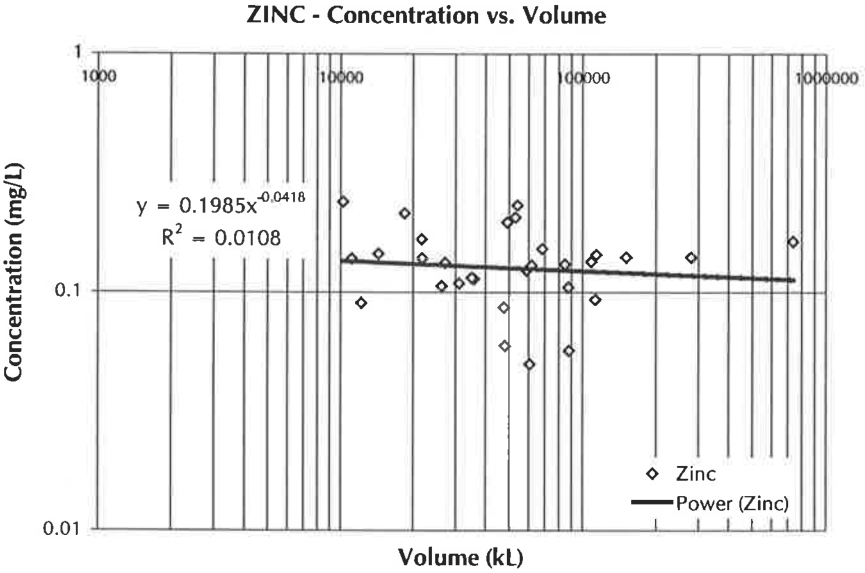


Figure 6.32 Zinc concentration vs. event volume relationship for South Road Connector outflow station

Also investigated was a relationship between rainfall volume and event load. The relationships for each of the parameters at the North Arm East site are given in Table 6.8. Relationships are not given for South Road Connector as it was considered that the outflow load would not be related to the event rainfall. Duncan (1997) tested rainfall as a variable for explaining basin performance (which is a function of outflow concentration) but found it significant in only a few cases.

Although this modelling technique eliminates the spurious correlation, the previous method was the preferred modelling technique as it offered relationships for both stations. This type of relationship may however, be useful for predicting loads with only pluviometer data in the absence of streamflow monitoring. Caution should be exercised when using a relationship based on pluviometer data as storms can be isolated, particularly in a large catchment such as that of the North Arm East drain.

Table 6.8 Event rainfall volume / event load relationships for North Arm East, including r^2 value. R = rainfall volume (m^3)

	North Arm East	
Water quality parameter	**Equation**	**r^2**
Ammonia (as N)	$0.0109R^{1.0745}$	0.65
Nitrate + nitrite (as N)	$0.0897R^{0.9674}$	0.58
TKN (as N)	$0.2215R^{1.0405}$	0.87
Filt. reactive phosphorus (as P)	$0.1178\ R^{0.8463}$	0.67
Total phosphorus	$0.0654R^{1.0213}$	0.80
Aluminium (total)	$0.0791R^{1.1729}$	0.75
Arsenic (inorganic)	$4E{-}05R^{1.2451}$	0.86
Cadmium (total)	$3E{-}05R^{1.1416}$	0.78
Chromium (total)	$0.0002R^{1.2349}$	0.85
Copper (total)	$0.0007R^{1.214}$	0.87
Iron (total)	$0.1089R^{1.1407}$	0.76
Lead (total)	$0.0012R^{1.2573}$	0.77
Manganese (total)	$0.0047R^{1.107}$	0.82
Mercury (total)	$2E{-}05R^{1.0165}$	0.71
Nickel (total)	$0.0008R^{1.0409}$	0.85
Zinc (total)	$0.0498R^{1.0591}$	0.87
Total suspended solids	$0.7164R^{1.3212}$	0.80

6.4.1.1 Validation

Data collected by Williams (1997) at the North Arm east site during 1995 and 1996 was used to validate the modelling technique discussed above. The load/volume relationships were applied to the runoff data available, and these modelled values were compared to the actual monitored values.

Figure 6.33 through to Figure 6.36 show the comparison between monitored and modelled pollutant mass inflow data for the North Arm East station. Figure 6.33 and Figure 6.35 have used the 1995-1996 data collected by Williams (1997), which was not used in the calibration. The slightly better fits demonstrated by Figure 6.34 and Figure 6.36 are a result of bias as much of the data was used in the model calibration.

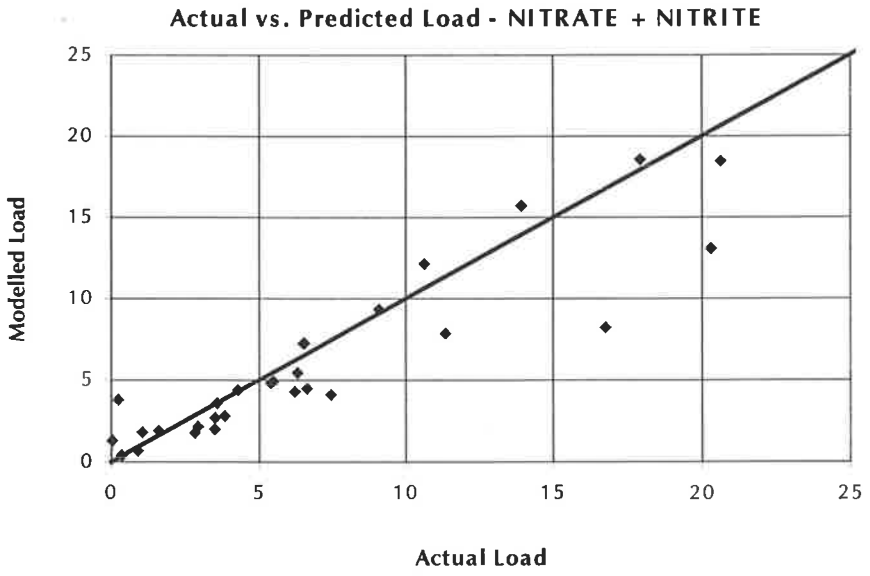


Figure 6.33 Actual vs. predicted loads of inflow nitrate + nitrite determined by applying load/volume relationship to 1995/96 data from North Arm East inflow station.

Aside from a few of the larger events, the relationship appears to model the pollutant load reasonably well as shown in Figure 6.33. The scatter in the data may be a result of poorer quality data obtained in 1995 and 1996 as noted by French (1999). Velocity meters did not function consistently during this time and a heavier reliance was placed on the rating curves. This may have reduced the accuracy of the flow calculations as there was no velocity data to use as a check.

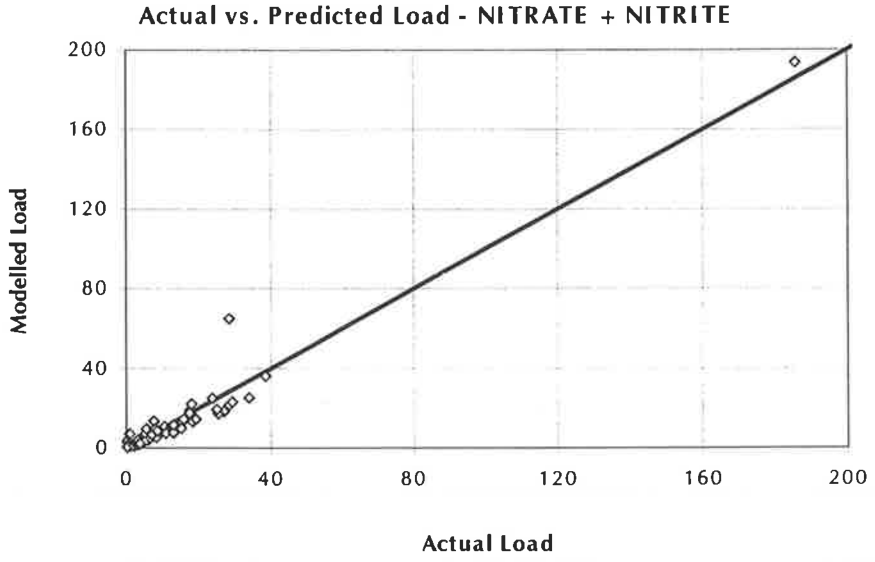


Figure 6.34 Actual vs. predicted loads of inflow nitrate + nitrite determined by applying load/volume relationship to 1997/98 data from North Arm East inflow station.

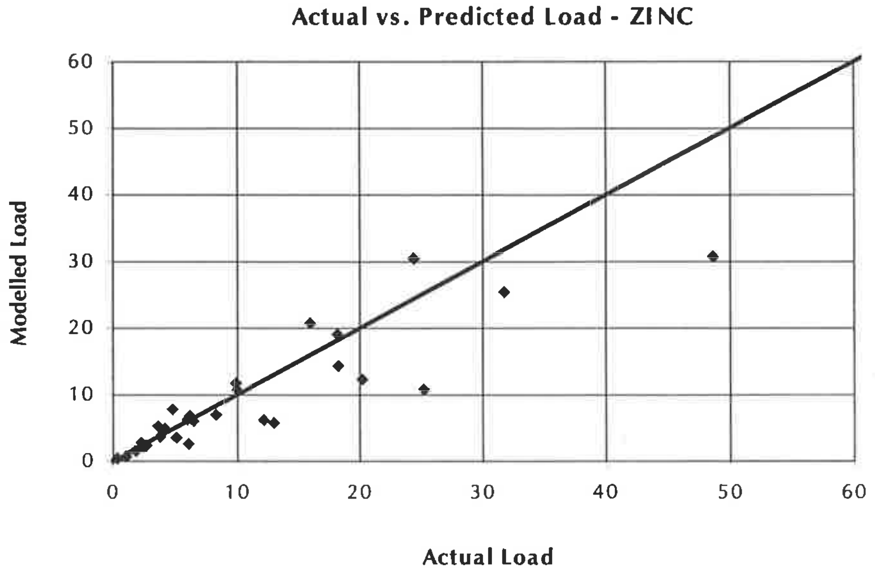


Figure 6.35 Actual vs. predicted loads of zinc determined by applying load/volume relationship to 1995/96 data from North Arm East inflow station.

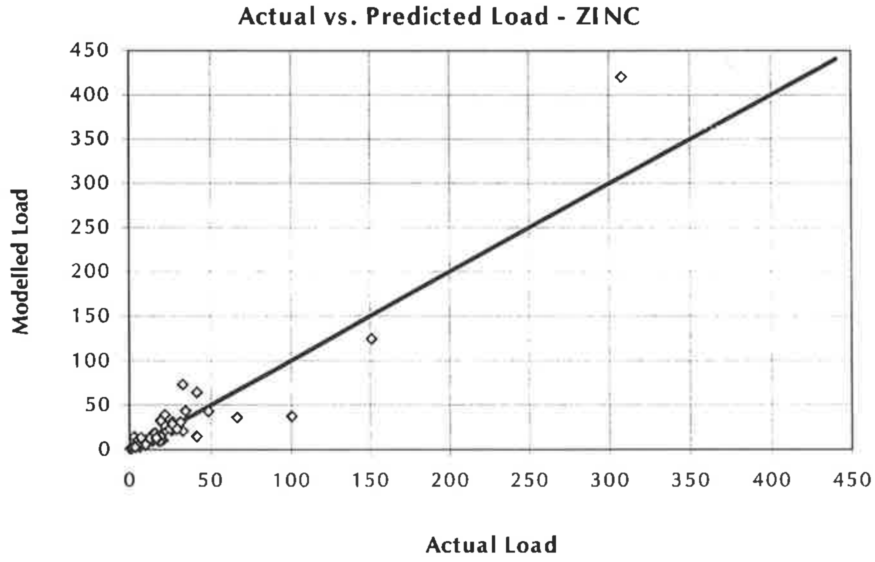


Figure 6.36 Actual vs. predicted loads of zinc determined by applying load/volume relationship to 1997/98 data from North Arm East inflow station.

Aside from a few of the larger events, Figure 6.36 also shows that the loads are modelled reasonably well. The same scatter as in Figure 6.33 can be noted.

The following graphs illustrate the monitored and modelled annual loads. The monitored loads show 15% error bars to demonstrate whether or not the modelled loads fall within 15 percent of the 'actual' loads. Some of the parameters have been multiplied by a factor of 10, 100, or 1000 simply so all the parameters could be displayed on the same graph.

Figure 6.37 compares the monitored and modelled loads for the 1995 an 1996 data collected by Williams (1997) which was not used in the model calibration.

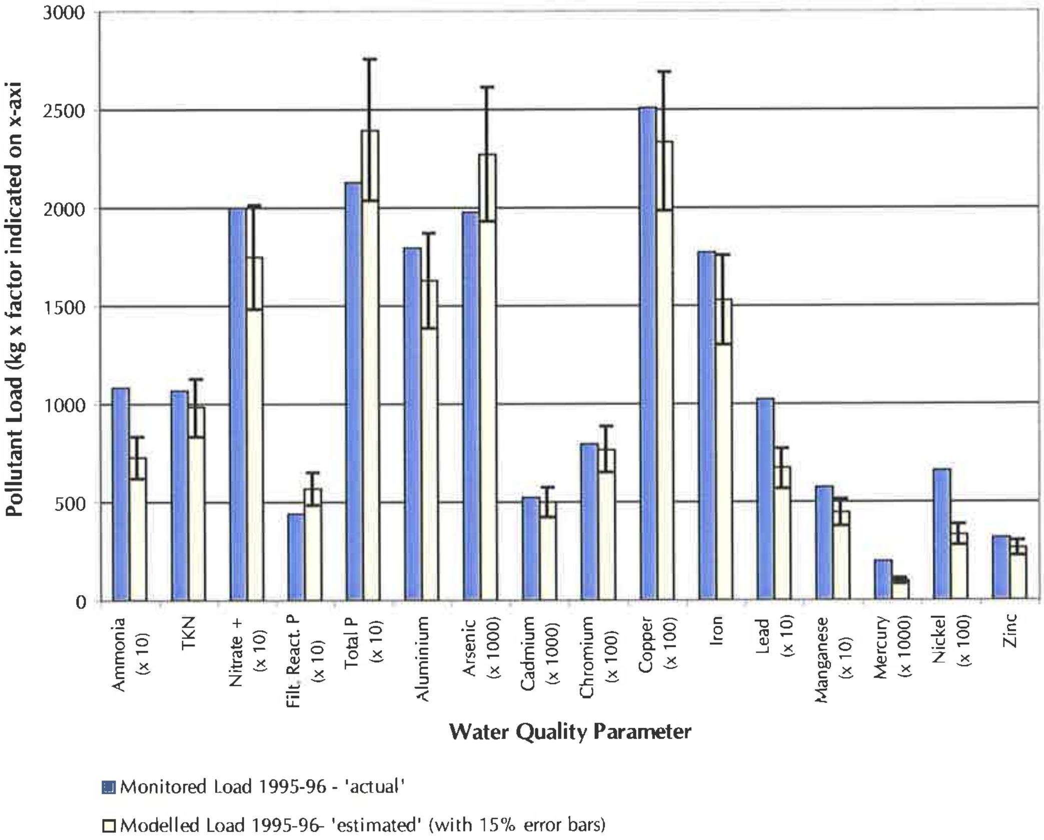


Figure 6.37 Comparison of monitored and modelled loads at North Arm East station using 1995/96 data

As can be seen, only six of the sixteen parameters modelled fall outside the 15 percent error bars. These parameters were ammonia, filterable reactive (or dissolved) phosphorus, lead, manganese, mercury and nickel.

Figure 6.38 is the same graph for the 1997 and 1998 data. As has already been mentioned, this data was used in the calibration and the better results are a result of this bias. With the exception of only two pollutants this time, all other parameters were modelled within 15 percent of the monitored values. The two exceptions were ammonia and filterable reactive phosphorus, which were not surprisingly the parameters with the highest coefficient of variation and subject to the largest fluctuations in concentration from event to event (see Table 6.1)

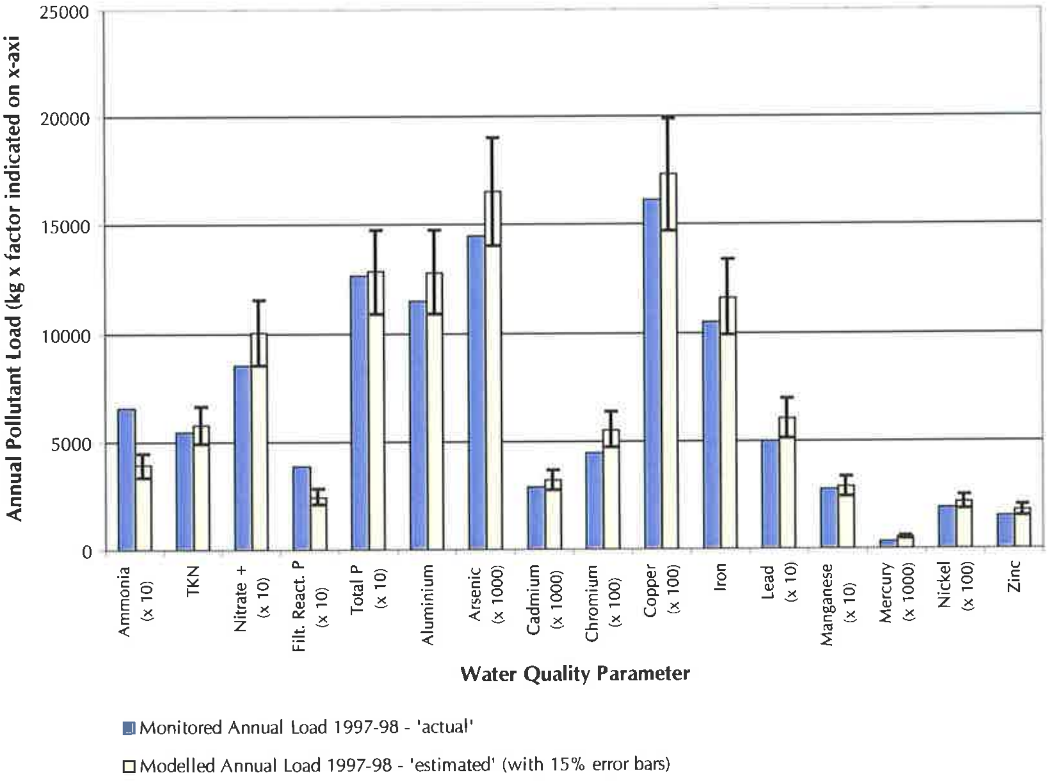


Figure 6.38 **Comparison of monitored and modelled annual loads at North Arm East station using 1997/98 data.**

The South Road Connector Station, being new, had no 1995/96 data available for model verification. Figure 6.39 shows the comparison between modelled and monitored values for the 1997 and 1998 data. In this case eleven of the 16 parameters were modelled within 15 percent. The exceptions were filterable reactive phosphorus, aluminium, cadmium, chromium and mercury.

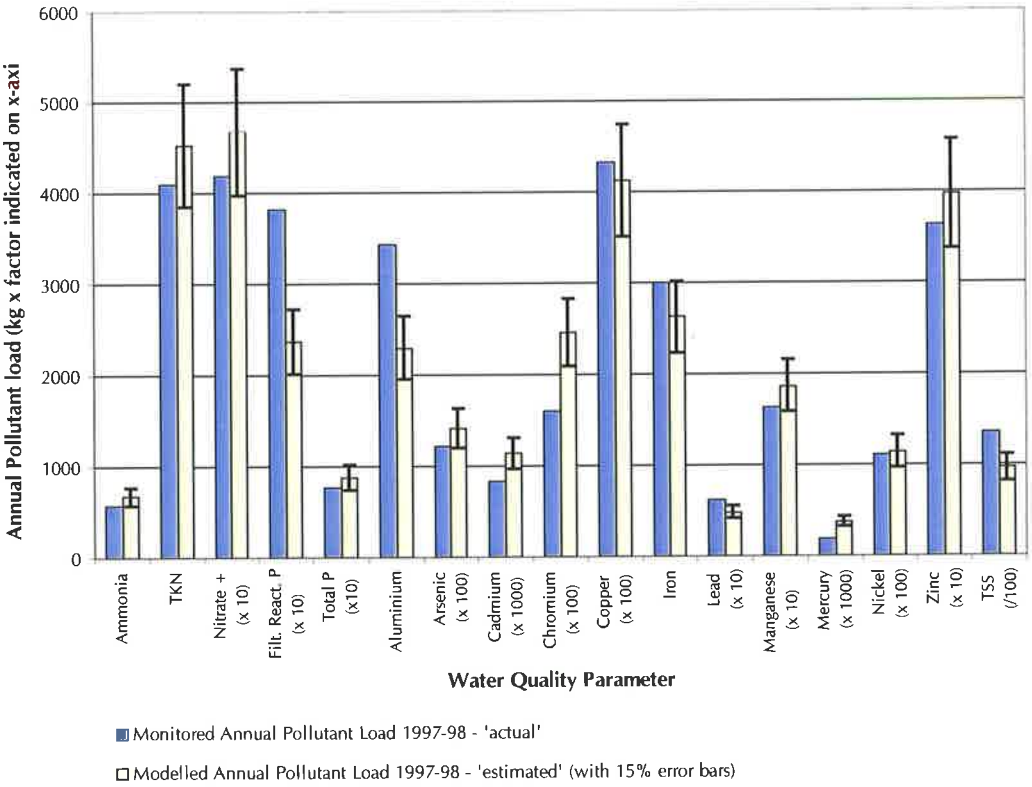


Figure 6.39 Comparison of monitored and modelled annual loads at South Road Connector outflow station using 1997/98 data.

With a majority of parameters at both stations being modelled within 15 percent of the observed values, this method was considered an appropriate technique for filling in gaps in the data set.

6.4.2 Data Estimation using other Parameters

Some contaminants in urban stormwater exhibit inter-relationships. These correlations can be the result of similarities in the pollutant source, or similarities in the transport medium such as suspended particulate matter (Williams, 1997). Correlations between water quality parameters were investigated by Williams (1997) with the aim of reducing monitoring requirements in the future to save on monitoring costs. In this section, parameter correlations were investigated as a means of filling in gaps in the data set when certain water quality information were not available.

A common correlation between turbidity, which can be measured continuously with relative ease, and suspended solids concentration is often investigated. The potential of using optical turbidity meters to estimate suspended solids concentration was investigated by Gippel (1995). He found that an adequate relationship between turbidity measured in the field and suspended solids concentration should be expected in most situations. Although some variations due to particle size and composition and water colour are likely, these can be tolerated as the continuous estimation of suspended solids content overcomes the problems associated with infrequent sampling or, in this case, malfunctioning sampling equipment.

The relationship between the two parameters at the North Arm East Drain is given in Figure 6.40. The blue scatter points are the results from sequential samples tested in 1997-1998, while the green points are those determined by Williams (1997) during 1995-1996. This figure shows that the two data collection periods yielded a very similar relationship. The graph indicates that for turbidities and TSS concentrations up to around 300 NTU or mg/L, the relationship is approximately 1-to-1. Past this point the relationship changes, probably due to the influence of particle size. Higher suspended solids concentrations typically include larger particles which have a significant effect on the mass of particles in suspension, but not necessarily the turbidity.

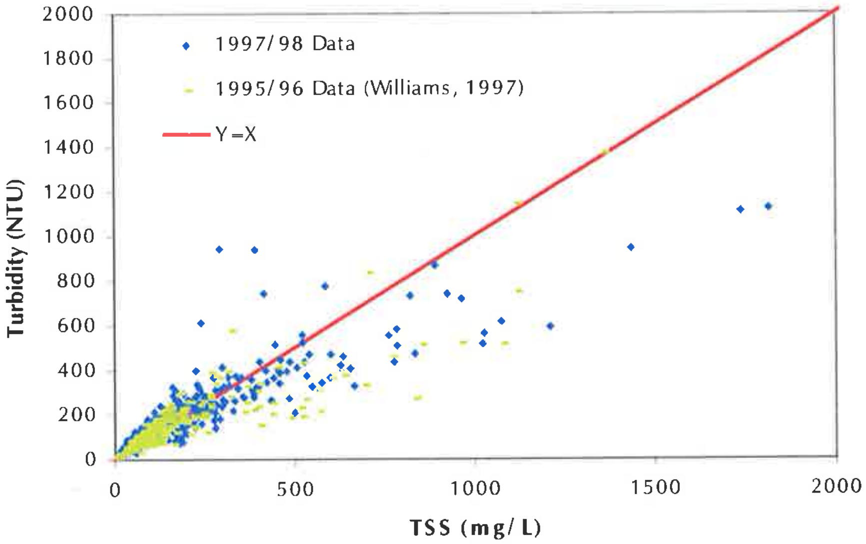


Figure 6.40 TSS vs. turbidity relationship (determined from sequential samples)

The usefulness of such a correlation is of course limited by the reliability and accuracy of the relationship, which is dependent on the size of the data set. Figure 6.41 shows the relationship determined for the entire data set; 1995-1998, which consists of some 750 points. The data set has been split into two separate graphs, TSS < 300 mg/L (left) and TSS > 300 mg/L (right). The left hand graph shows an almost 1-to-1 relationship for TSS concentrations less than 300 mg/L, while the right hand graph shows a different relationship for TSS concentrations greater than 300 mg/L. Gippel (1995) states that an r^2 value of at least 80 percent is desirable for a predictive model between TSS and turbidity. The r^2 value of the relationship for TSS > 300 mg/L is significantly less than 0.8 as some points are closer to the 1-to-1 relationship. Although this method of estimation would be better than having no data at all, it is likely to be a little unreliable unless other factors such as particle size are taken into consideration.

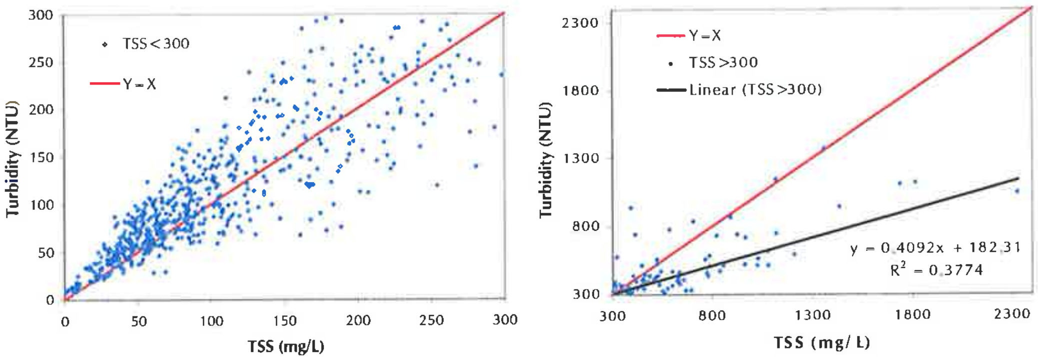


Figure 6.41 (left) Turbidity vs. TSS relationship for TSS<300mg/L; (right) turbidity vs. TSS relationship for TSS>300mg/L

Some inter-relationships were found between parameters during composite sample analysis. Figure 6.42 shows a strong relationship between total Kjeldahl nitrogen and total phosphorus, with an r^2 value of 77 percent.

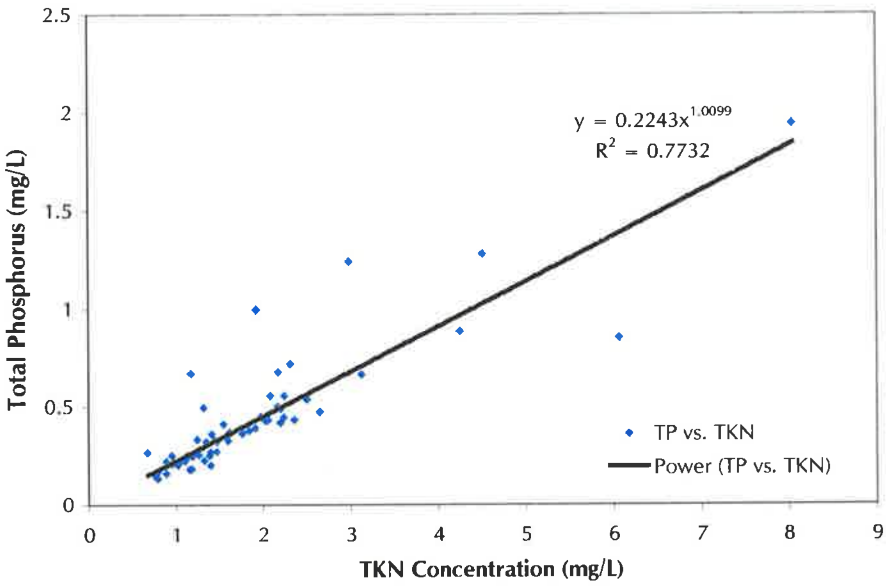


Figure 6.42 Total phosphorus vs. total Kjeldahl nitrogen relationship (determined from composite samples)

Another even more significant correlation was found between aluminium and iron, as shown in Figure 6.43.

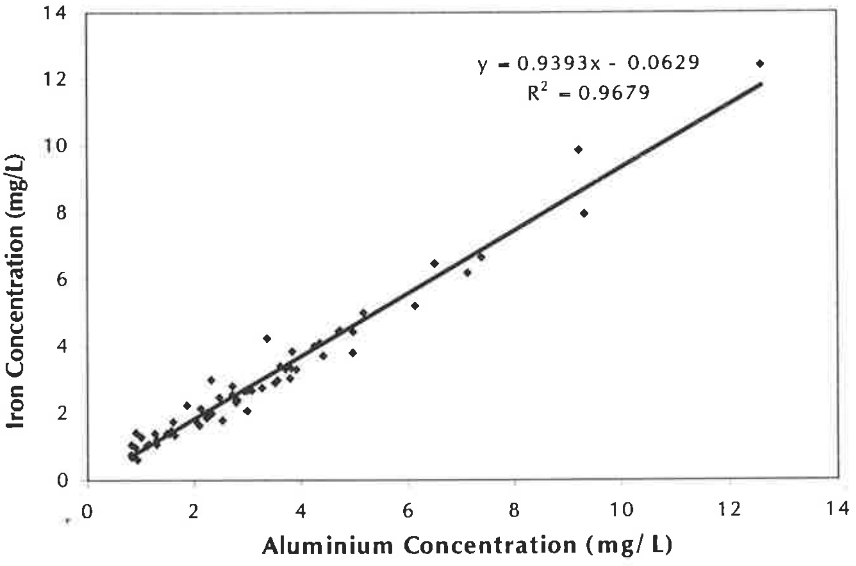


Figure 6.43 Iron versus aluminium relationship (determined from composite samples)

The inter-relationships between pollutant species may be due to similarities in transport mechanisms, or common sources. This fact was summarised by pH Environment (1995).

"Correlations between metals and metals could indicate a common source in industry or land use. Correlations between metals and organic matter are more likely to reflect the potential of organic surfaces to adsorb and transport metals. Correlations between metals and nutrients may be a result of the total nitrogen content of organic matter but correlations between nutrients would probably indicate a common source."

It is well recognised that a significant amount of pollutants are transported by stormwater as sediment bound contaminants. Mann and Hammerschmid (1989) conducted a study on urban runoff from two Australian catchments in Hawkesbury / Nepean Basin and found strong relationships between total suspended solids (TSS) and total phosphorus (TP), total nitrogen (TN), and chemical oxygen demand (COD). A similar study by Ball and Abustan (1995) also found a high correlation between TSS and TP. Williams (1997) investigated parameter correlations such as turbidity versus total phosphorus, and TSS versus total phosphorus, but failed to find any relationships of significance in the Barker Inlet Catchment.

While the relationships found above may be useful for reducing testing requirements, they are not able to estimate all the parameters required.

6.4.3 Data Estimation Using Mean Annual Concentration

Millar (1996) states that the problem with mass export/runoff volume approaches to data estimation is spurious correlation and the false confidence it instils in the predictive ability. He goes on to say that there may be no improvement to simply using mean concentration.

Schueler (1987) prepared a manual in which he makes reference to a "Simple Method" for estimating pollutant export from urban developments. He carried out a statistical analysis of over 300 events at eight locations in Washington and over 2000 events at 22 sites covering a much larger area. The conclusions from his analysis indicated that there was no significant difference in the average pollutant concentrations between the widely different urban locations, and there was no significant correlation between pollutant concentration and event volume or intensity. His modelling approach was therefore based on rainfall, runoff coefficients and mean annual concentration of pollutants (averaged from empirical studies). A similar approach was investigated in this study.

Out of the 54 events monitored over the 12-month period being discussed, only 5 events were not monitored successfully at the inlet while seven events were missed at the outlet. These "missed" events represent only 10 percent and 13 percent of the total number of events respectively. Using the mean concentration of events monitored successfully, it is possible to estimate the annual loads flowing in and out of Pond 4. This method is not likely to be as reliable for individual events.

The mean annual concentrations determined for the North Arm East and South Road Connector Stations during the 12-month period (1st August 1997 - 31st July 1998) are summarised in Table 6.9.

Table 6.9 Mean annual concentrations for North Arm East and South Road Connector Stations (determined from data collected 1st August 1997-31st July 1998)

Water Quality Parameter	NAE Inflow Mean Annual Concentration (mg/L)	SRC Outflow Mean Annual Concentration (mg/L)
Ammonia	0.227	0.216
Nitrate + Nitrite as N	0.284	0.164
TKN	1.68	1.52
Filterable Reactive P	0.131	0.133
Total P	0.430	0.289
Aluminium	3.66	1.23
Arsenic	0.0046	0.0046
Cadmium	0.0009	0.0003
Chromium	0.015	0.0065
Copper	0.047	0.016
Iron	3.26	1.10
Lead	0.154	0.022
Manganese	0.083	0.064
Mercury	0.0001	< 0.0001
Nickel	0.0058	0.004
Zinc	0.455	0.137
TSS	188.5	48.4

The data estimation method has been validated in the following section.

6.4.3.1 Validation

The modelling technique was validated for the North Arm East site once again using the 1995 and 1996 data collected in an earlier study by Williams (1997). The same data were not available for the South Road Connector Site and so no verification could be carried out for this site. The comparison between monitored and modelled loads is given in Figure 6.44, which also shows15 percent error bars.

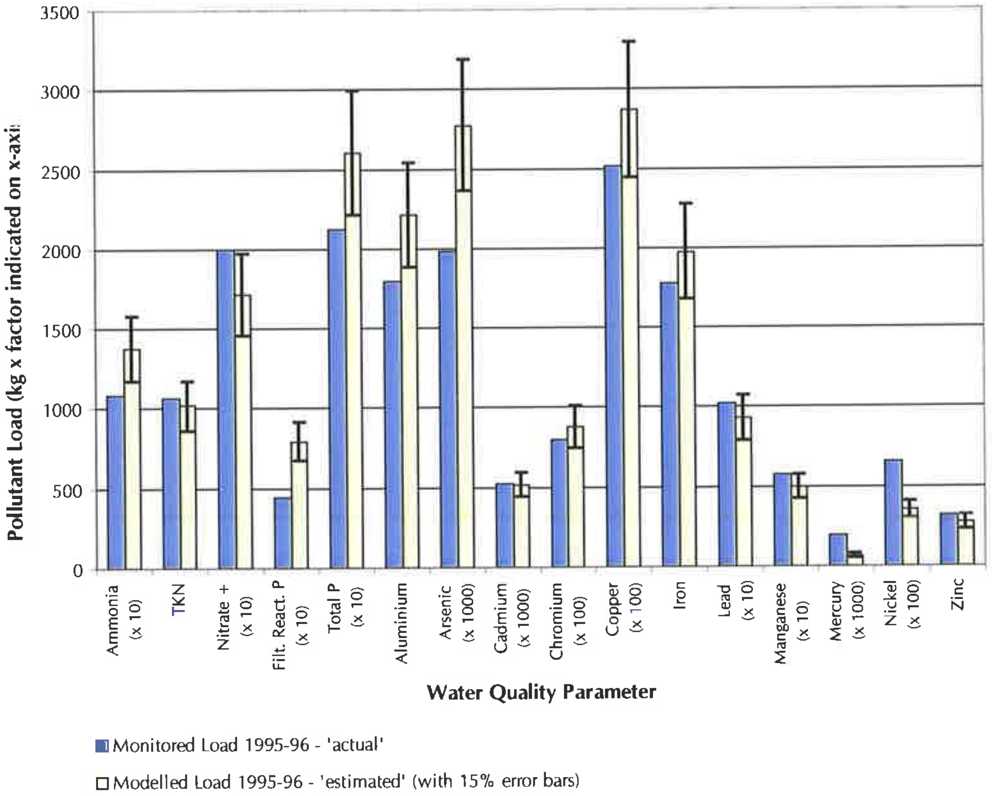


Figure 6.44 Comparison between 1995/96 monitored NAE loads and loads modelled using mean annual concentrations.

As can be seen, there is some difference in the model capability, however this is not to say that it is any better than the volume/load relationships. Table 6.10 compares the errors obtained for both the volume/load and the mean annual concentration modelling approaches for each of the water quality parameters.

Table 6.10 Errors obtained for the volume/load and mean annual concentration modelling approaches.

Water Quality Parameter	Load/Volume Relationship Modelling Error (%)	Mean Annual Concentration Modelling Error (%)
Ammonia	33.0	27.0
Nitrate + Nitrite as N	**12.5**	**14.1**
TKN	**7.8**	**4.6**
Fil. Reactive P	28.8	79.6
Total P	**12.8**	22.6
Aluminium	**9.3**	23.2
Arsenic	**14.7**	40.1
Cadmium	**4.6**	**0.96**
Chromium	**3.5**	**10.6**
Copper	**6.9**	**14.3**
Iron	**13.7**	**11.3**
Lead	34.2	**8.5**
Manganese	22.6	**13.4**
Mercury	51.0	66.5
Nickel	49.3	46.1
Zinc	16.3	**12.6**
Average Error	18.88	23.26

The table shows that both of the predictive methods model nine of the parameters better than the other. It also shows that the two methods each predict the same number of parameters within the 15 percent margin (shown in **bold**), although the parameters differ between the models. The load/volume relationships provide a smaller average error across all the parameters modelled.

The mass export/runoff volume approach to data estimation was chosen to be used in this study as it was capable of predicting event loads based on the amount of runoff rather than just relying on mean annual concentration. It should be noted that there is not expected to be any *significant* difference in the annual loads if the other predictive model had been chosen.

6.5 Annual Loads

Although looking at discrete events gives an indication of short term pond performance, results can be misleading due to the complex physical, chemical and biological reactions which readily occur in wetland and pond systems. As discussed in Chapter 3 (Wetland Processes), nutrient levels in particular fluctuate and may decrease during flow events but be re-released and actually increase during quiescent conditions. Lawrence and Breen (1998b) suggest that a minimum of 30 days following an event needs to be examined to determine the full sediment and algal response to the storm event.

To determine the longer-term performance of the pond, a full 12 months of data were examined, eliminating the effects of seasonality, which may otherwise provide misleading results.

Figure 6.45 displays the event flows for the 12-month period commencing August 1st 1997. They are represented on a pie chart as a percentage of the total annual flow to put into perspective the relative magnitude of each of the events, in particular the October 30 event. The chart shows that around 50 percent of the annual flow was contributed by just seven of the 54 events which were, in descending order of contribution:

1. October 30th 1997
2. April 19th 1998
3. April 11th 1998
4. June 11th 1998.
5. December 18th 1997
6. June 6th 1998
7. August 30th 1997

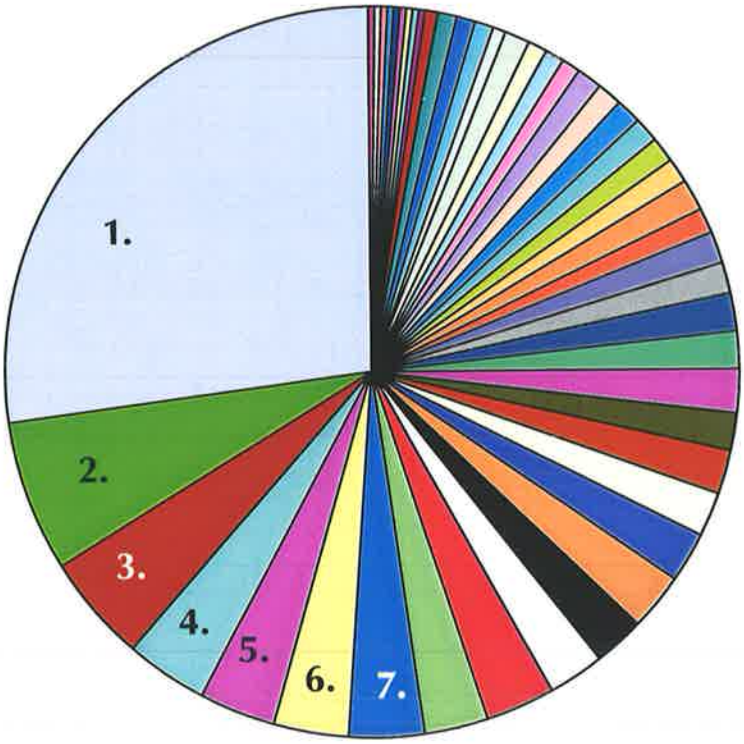


Figure 6.45 Event flow for 12-month period 1st August 1997 – 31st July 1998 displayed as a percentage of total annual flow

A number of authors (Cullen, 1989; Raisin *et al.*, 1997, Murphy *et al.*, 1998; Lawrence & Breen, 1998b) have noted the importance of large events in transporting a majority of pollutants from the catchment. This phenomenon is clearly demonstrated by Figure 6.46 to Figure 6.50. To give an example of their application, the distribution of total phosphorus (inflow) shown in Figure 6.46 indicates that 65 percent of the total annual total phosphorus load is contributed by only 20 percent of the inflow events. Another way of looking at this is that if a system is designed to capture only 80 percent of inflow events (bypass the high flows), 65 percent of the total phosphorus transported from the catchment would bypass the system untreated.

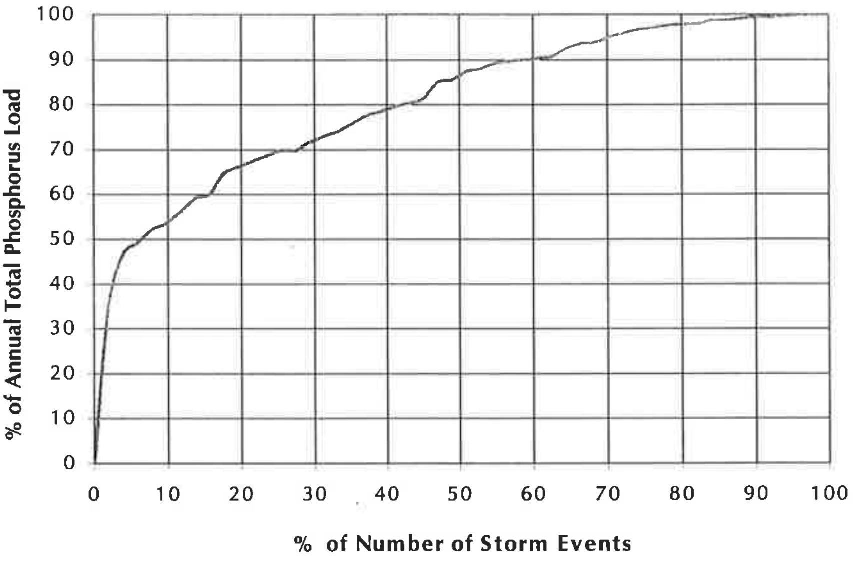


Figure 6.46 Distribution of total phosphorus event loads at inflow to Pond 4.

A similar distribution was obtained for lead as shown in Figure 6.47. Again, 60 percent of the annual lead load entering the pond was contributed by just 20 percent of the events. At the outlet however, almost 80 percent of the lead exiting the pond was contributed by the top 20 percent of events. This demonstrates the much smaller degree of treatment obtained by the larger event with their shorter residence times.

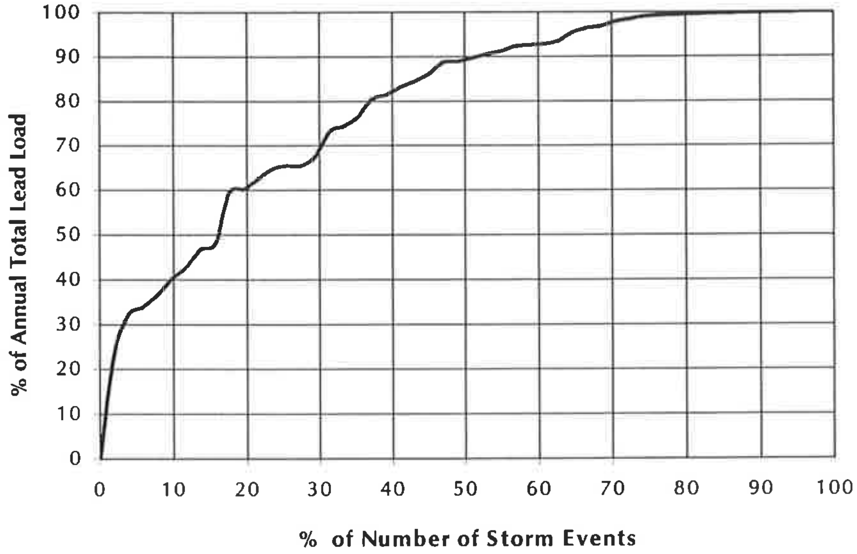


Figure 6.47 Distribution of lead event loads at inflow to Pond 4.

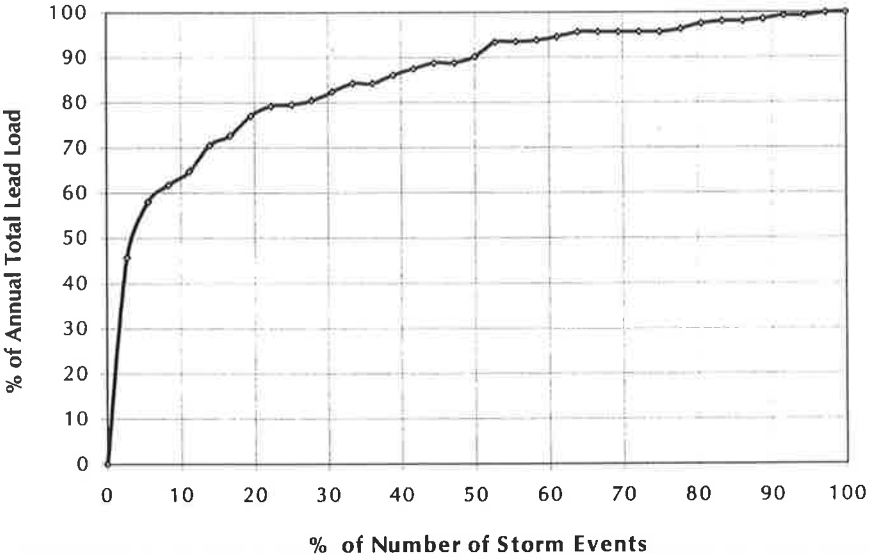


Figure 6.48 Distribution of lead event loads at outflow to Pond 4.

The same trends were exhibited for total suspended solids (Figure 6.49 and Figure 6.50) which is not surprising considering the close association of lead and sediment particles as has already been discussed in Section 6.1.2.

Similar findings have been noted by Cullen (1989) who reported 69 percent of phosphorus was transported by nine percent of flows, and Raisin *et al.* (1997) who found 41 percent of phosphorus was transported by 24 percent of the annual discharge.

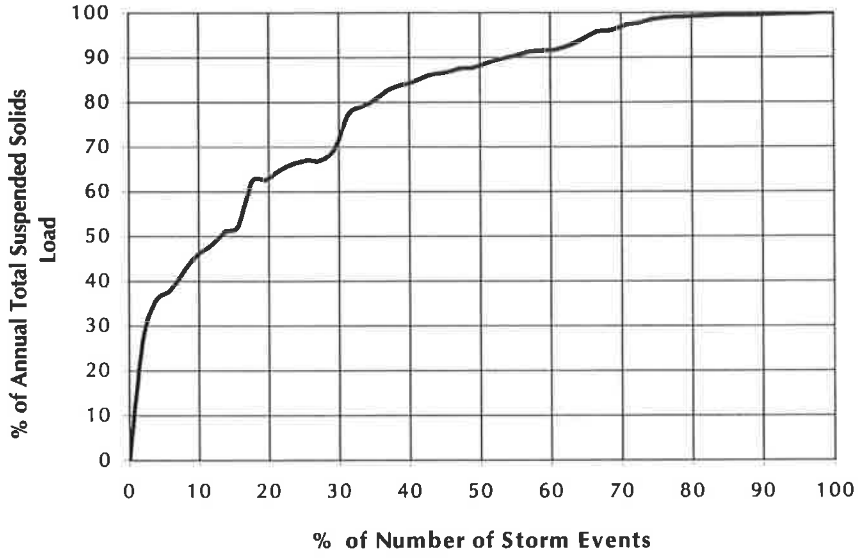


Figure 6.49 Distribution of total suspended solids event loads at inflow to Pond 4.

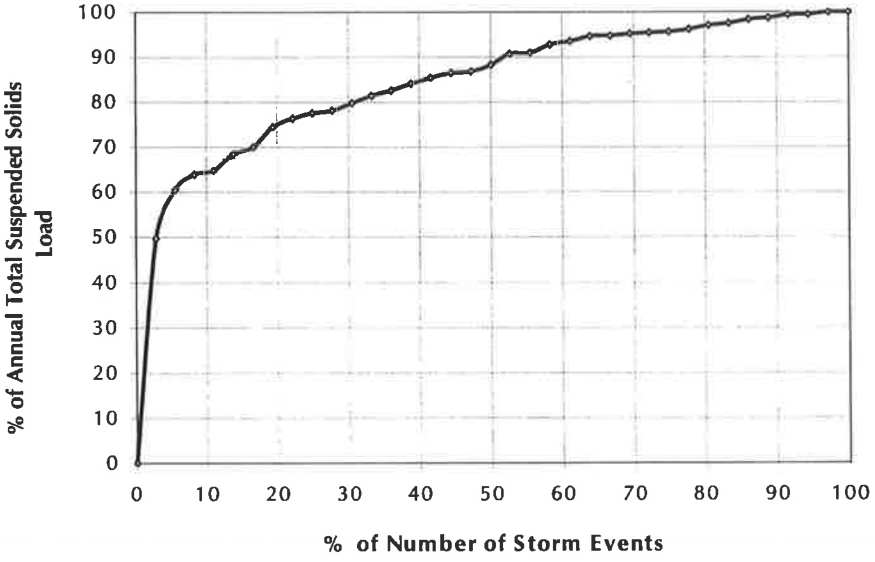


Figure 6.50 Distribution of total suspended solids event loads at outflow to Pond 4.

The total annual loads for the North Arm East and South Road Stations are given in Table 6.11.

Table 6.11 Annual loads for North Arm East and South Road Connector Stations.

Parameter	NAE Inflow Load (kg)	SRC Outflow Load (kg)
Ammonia	637	653
Nitrate + Nitrite	795	495
TKN	4707	4596
Organic N	4070	3943
Total N	5502	5091
Filt. Reactive P	367	403
Particulate P	838	472
Total P	1205	875
Aluminium	10238	3720
Arsenic	13	14
Cadmium	2	1
Chromium	41	20
Copper	133	50
Iron	9141	3324
Lead	432	68
Manganese	232	193
Mercury	0.3	0.26
Nickel	16	12
Zinc	1275	414
Total Suspended Solids	528 T	146 T

T = tonnes

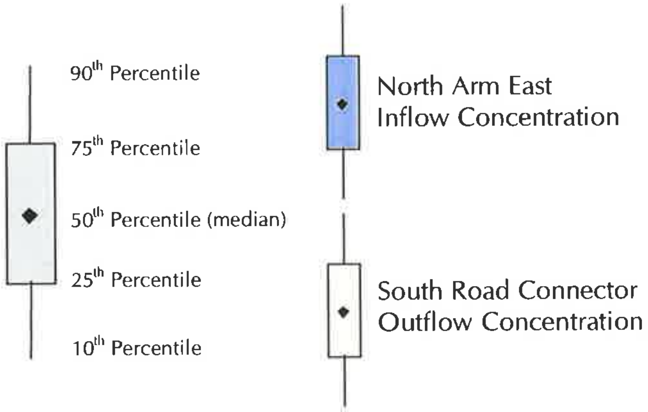
6.6 Pond Performance

Data on both an event by event and an annual basis has been discussed in the previous sections. This section will present the results in terms of pond performance, which was after-all the main aim of the research.

It has been noted by Wu *et al.* (1996) that it is not uncommon in water pollution control ponds, to experience complete or negative removals at different times, at the same site. Therefore, they go on to suggest that performance should be based on long term removals of pollutant mass loadings rather than performance of a single event basis. The following results will provide estimates of long term performance as well as estimations of response to flow based on the extended data set.

Following is a series of "box and whisker" plots which display the inflow and outflow concentrations for the various pollutants. The water quality parameters have been grouped with individual pollutants being graphed alongside other pollutants exhibiting concentrations of a similar magnitude (for example iron and aluminium). An explanation of the graphs is shown below. Briefly, they show the 90th, 75th, 50th (median), 25th, and 10th percentile values of the concentrations side by side to give an indication of how the pond reduces (or increases) the pollutant concentrations over the long term. The median (50th percentile) values are shown by the ◆ and are the best parameter to use as a performance indicator.

EXPLANATION OF FIGURES



The first plot to examine, Figure 6.51, shows the inflow and outflow concentrations for the various species of nitrogen analysed.

As well as showing the difference between the inflow and outflow annual concentrations, Figure 6.51 shows the relative concentrations of each of the species of nitrogen. The ammonia and nitrate+nitrite concentrations were quite low while the TKN and TN concentrations were quite high. Since TKN is the sum of organic and and ammonia-nitrogen, a majority of the nitrogen in the system must be in the form of organic nitrogen. Studies have in fact shown that the majority of nitrogen in stormwater is in organic form (Yousef *et al.*, 1986). Although organic material, or vegetation, would appear to be a major source of organic nitrogen in stormwater, a CRC for Catchment Hydrology study (Allison *et al.*, 1997) has shown that vegetation is not a major source of nutrients compared to other sources. The CRC monitoring program indicated that the potential total phosphorus and total nitrogen loads in stormwater contributed by vegetation were about two orders of magnitude lower than the loads actually measured in the stormwater samples.

Figure 6.51 also shows that the median of the annual ammonia concentration is higher at the outlet than the inlet, while the mean annual concentration (shown in Table 6.9) was slightly lower at the outlet. An increase in ammonia concentration was demonstrated in the individual events (Section 6.1) and in fact displayed a negligible decrease over the long term. Duncan (1995a) has stated that nitrogen exhibits complex behaviour in storage as it is not a conservative parameter. Chapter 3.2 described the various processes in the nitrogen cycle which were fixation, nitrification, denitrification, ammonification, and assimilation. On occasions when the organic nitrogen (or TKN) component decreases while the ammonia component increases, it is possible that the rate of ammonification is exceeding the rates of nitrification and assimilation which are the dominant pathways for the removal of ammonia. In other words more ammonia is being produced by the microbial breakdown of organic nitrogen that can be converted to nitrate in the nitrification process or assimilated by plants and algae.

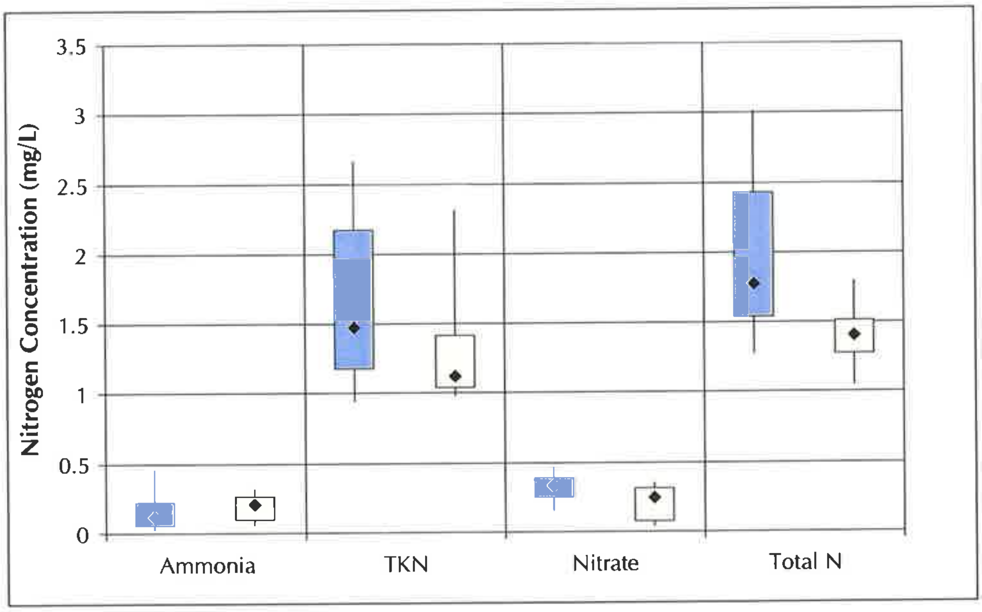


Figure 6.51 Inflow (blue) and outflow (yellow) concentration ranges for nitrogen species

Figure 6.52 demonstrates that both the particulate and total forms of phosphorus decrease in storage over the long term. The median of the dissolved phosphorus was shown here to also decrease, although it will be shown shortly that there was very little change in the inflow and outflow *loads* for the 12-month period. This discrepancy occurs as the dissolved phosphorus in the pond water periodically was higher than the inflow concentration, while on other occasions it was lower. Erratic behaviour with regard to phosphorus concentrations has been noted by a number of authors. Mulhern and Steel (1988) noted the phenomenon of phosphorus generation in a study of a water quality pond in Denver, USA. Here it was attributed to remobilisation of phosphorus from bottom sediments during periods of algal growth and associated high oxygen demand, a process described in more detail in Kadlec and Knight (1996). Similar findings were also reported by Jacobi and Murphy (1996). Phosphorus remobilisation was also discussed by Böstrom *et al*. (1982) who stated that many factors also contribute to the release of phosphorus from sediments under aerobic conditions. The release of phosphorus from sediment to overlying water is mainly a result of dissolved phosphorus existing in the pore water that is directly exchangeable and highly mobile. Release can be readily induced by diffusion, wind turbulence, or bioturbation (Prescott & Tsanis, 1997).

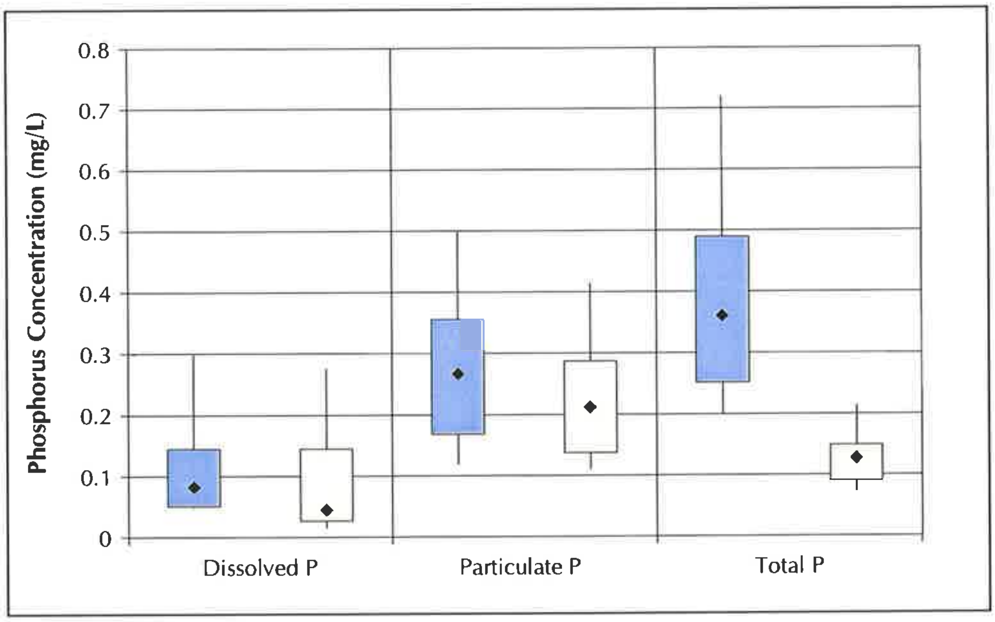


Figure 6.52 Inflow (blue) and outflow (yellow) annual concentrations for phosphorus species

Duncan (1997) reported that while suspended solids can be a significant pollutant in its own right, it is even more important when one considers the association with many other pollutants like heavy metals. Because of their association with particulate matter, effective suspended solids reduction will also lead to significant metal reduction. Lead and zinc are shown to reduce substantially within the pond by Figure 6.53, while the removal of manganese is somewhat lower. This provides an indication of the affinity of these metals for sediment particles which subsequently settle out. While lead and zinc are primarily particle bound nutrients, manganese has a more significant dissolved fraction. This behaviour was noted by Duncan (1997) who classified water quality parameters according to their behaviour in storage. He found a group, which he called the settling group, to be comprised of suspended solids, lead and zinc.

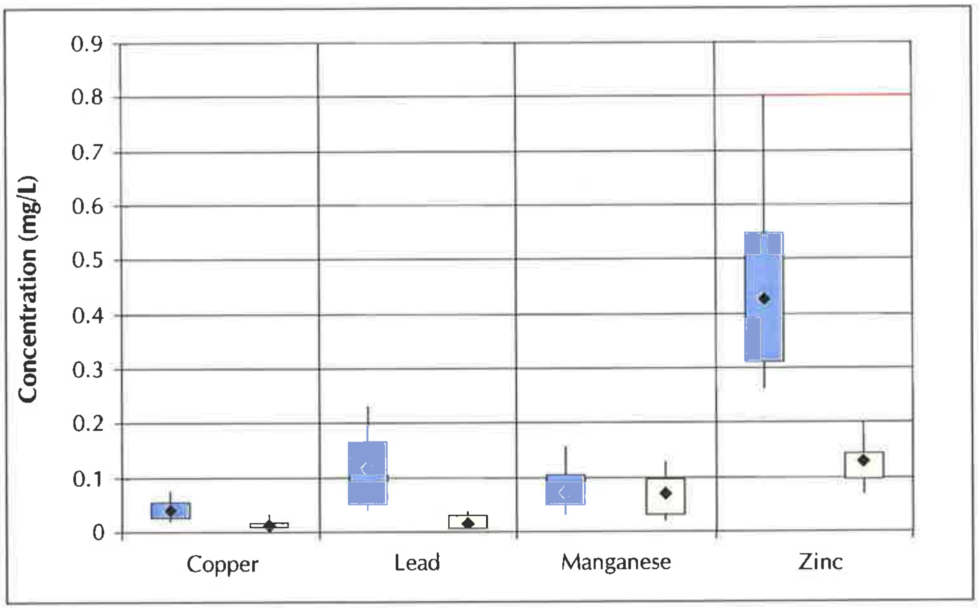


Figure 6.53 **Inflow (blue) and outflow (yellow) annual concentrations for copper, lead, manganese, and zinc.**

Figure 6.54 indicates the variation in the concentrations of iron and aluminium experienced over the study period. The similarity between the two parameters is not surprising considering the correlation that was shown to exist between the two pollutants in Figure 6.43. The close correlation as well as the similarity in the behaviour in storage (both were removed to a similar degree, cf. Table 6.12) is likely to be an indication of a common source as well as a common transport mechanism. If iron and aluminium were attached to similar sized particles and were present in simlar concentrations, it would be expected that they would be removed to a similar degree.

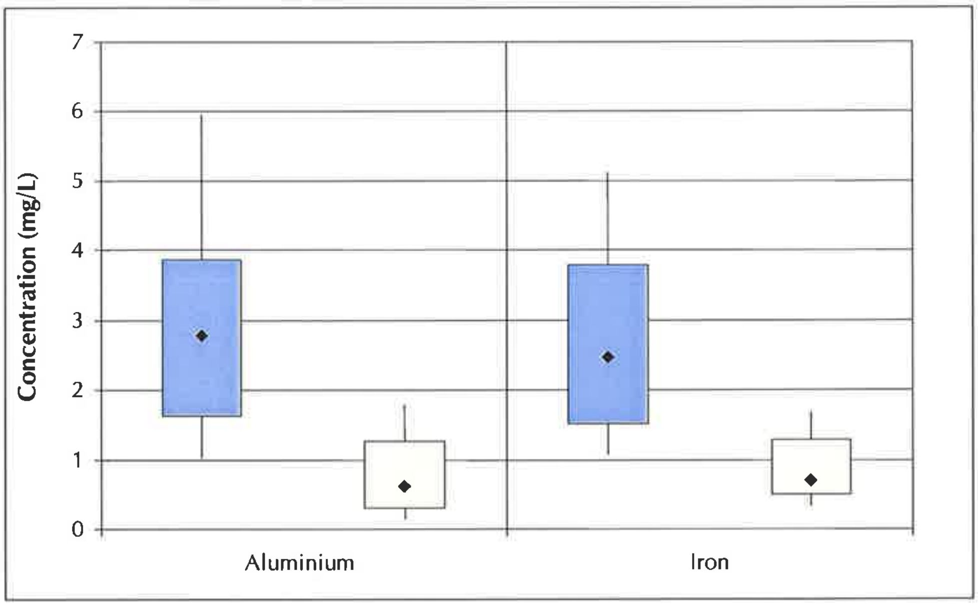


Figure 6.54 **Inflow (blue) and outflow (yellow) annual concentrations for aluminium and iron.**

The behaviour of the remainder of the pollutants analysed are shown in Figure 6.55.

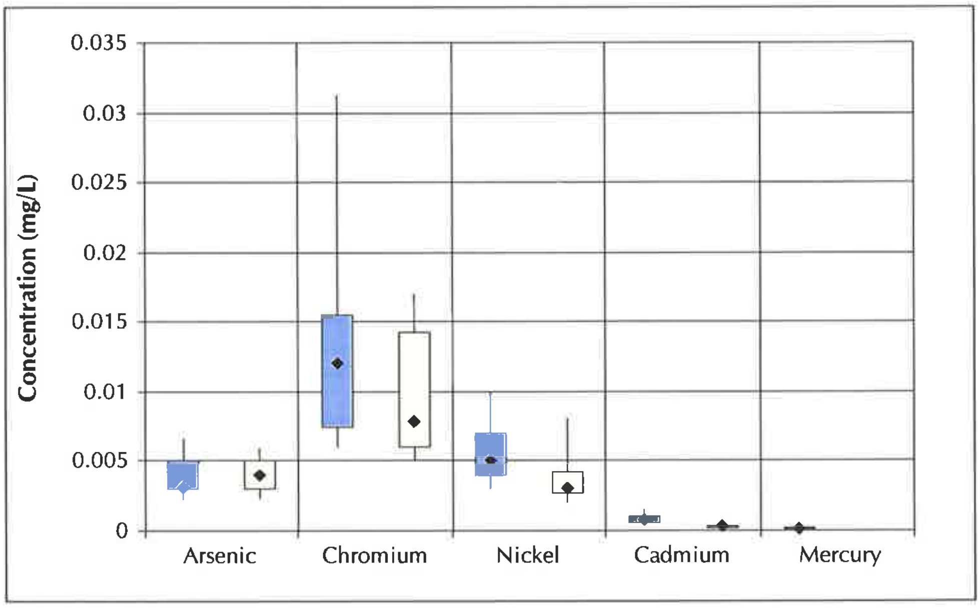


Figure 6.55 **Inflow (blue) and outflow (yellow) annual concentrations for arsenic, chromium, nickel, cadmium and mercury.**

While it is common, in the determination of pond effectiveness, to express performance in terms of event mean concentrations, it has been suggested by other authors that load is a more relevant parameter (ARC, 1992; NSWDH, 1993; ANZECC, 1996).

Table 6.12 summarises the total pollutant loads that entered and exited BIW Pond 4 during the 12-month study period. The NAE inflow loads have been factored up by 10% to account for the Henschke Street contribution.

Table 6.12 Total inflow and outflow loads from BIW Pond 4 including annual load reduction

Parameter	Total inflow to BIW Pond 4 (kg)	Total outflow from BIW Pond 4 (kg)	Reduction in pollutant load (%)
Ammonia	701	653	6.8
Nitrate + Nitrite	874	495	43.4
TKN	5117	4596	11.2
Organic N	4477	3943	11.9
Total N	5326	5091	15.9
Filt. Reactive P	404	403	0.2
Particulate P	750	472	48.8
Total P	1326	875	34.0
Aluminium	11262	3720	67.0
Arsenic	14	14	0
Cadmium	3	1	54.5
Chromium	45	20	55.6
Copper	146	50	65.8
Iron	10055	3324	66.9
Lead	476	68	85.7
Manganese	255	193	24.3
Mercury	0.34	0.26	13.3
Nickel	18	12	33.3
Zinc	1403	414	70.5
Total Suspended Solids	581 T	146 T	75.0

T = tonnes

The annual inflow and outflow loads are also shown in graphical form in Figure 6.56.

The graph shows that all parameters, with the exception of one, exhibit removal to some degree. Some removals are clearly much greater than others, filterable reactive, or dissolved, phosphorus was the only parameter that was not removed successfully over the one year period. Also indicated by Figure 6.56, is the fact that the expression of pond performance in terms of concentration can sometimes be misleading. While Figure 6.51 appeared to suggest that more ammonia left the pond than entered it over the 12-month period, Figure 6.56 in fact showed that there was a net reduction of ammonia, albeit small. Similarly, Figure 6.51 showed significant reductions in the median concentrations of TKN and dissolved phosphorus, while the reductions in terms of loads were actually quite small.

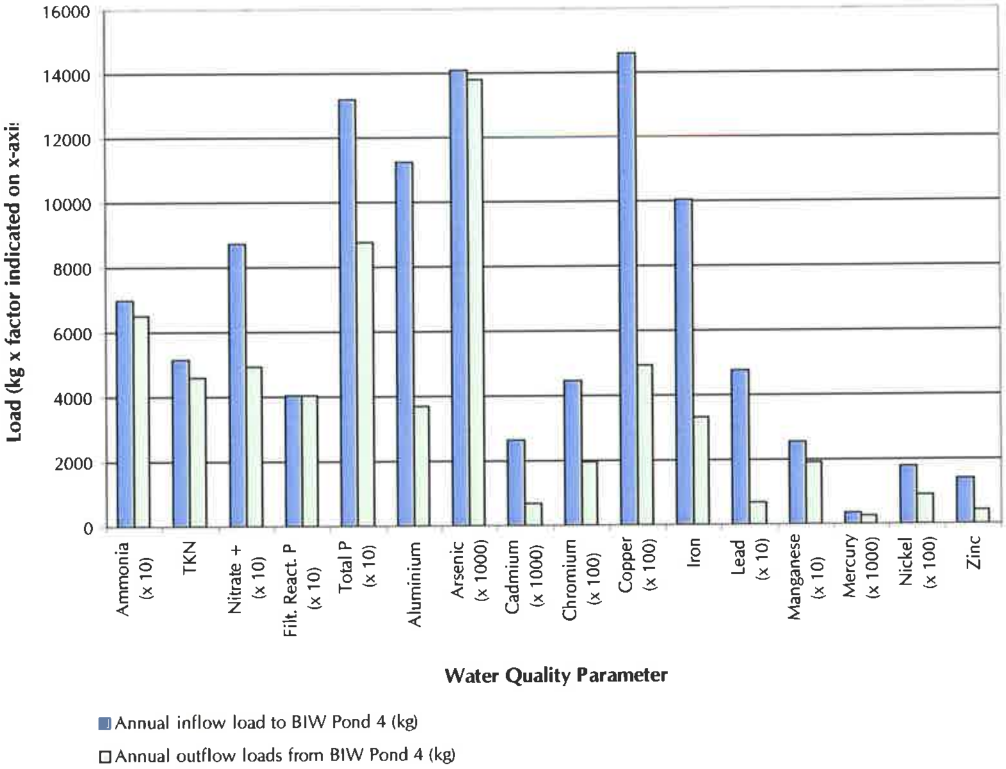


Figure 6.56 **Inflow (blue) and outflow (green) loads to BIW Pond 4 over 12-month period. Some parameters have been multiplied by a factor (indicated on x-axis) for clear display on the graph.**

The highest removal was achieved for lead (85%), total suspended solids (75%), zinc (70%), aluminium (67%) and iron (67%), which agrees with results reported by other researchers. Once again these are the pollutants commonly associated with particulate matter which is removed readily from the water column via sedimentation.

6.6.1 The Effect of Event Size on Pond Performance

It is a well documented fact that flow conditions within ponds and wetlands play an important role in the determination of their ultimate performance. It has also been suggested by a number of authors that, for this reason, it is desirable to divert high flows to prevent resuspension of previously deposited pollutants. The effect of event size on the performance of Pond 4 was discussed briefly in Section 6.1 which presented results of three individual events of varying magnitude. Following are a series of plots which aim to quantify the effect of event volume on the performance of Pond 4.

The relationships developed in Section 6.4.1 for load in and out versus event volume, have been used to develop a series of "load in/load out" curves, examples of which are given by Figure 6.57 through to Figure 6.59. The entire set of load in and load out plots are given in Appendix F.

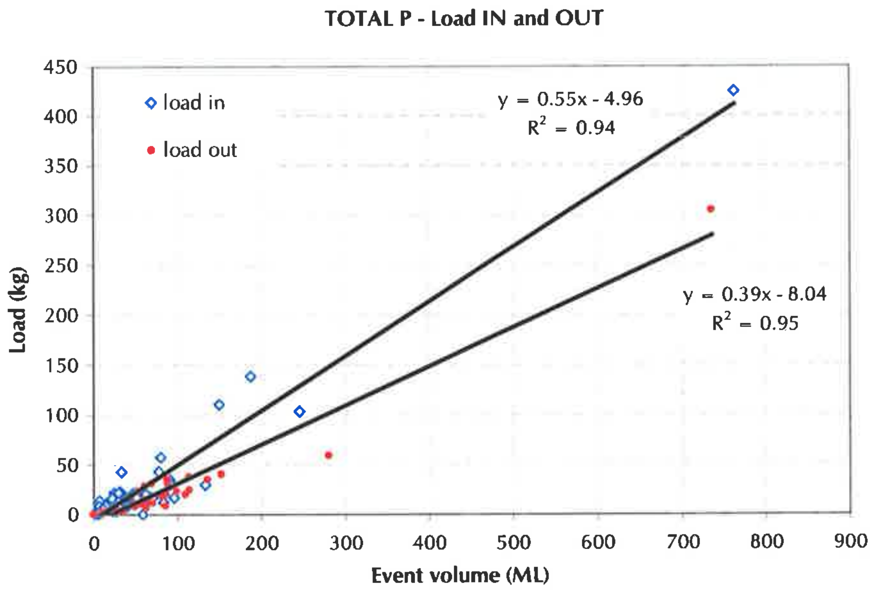


Figure 6.57 Total phosphorus load in and load out of BIW Pond 4 related to event volume

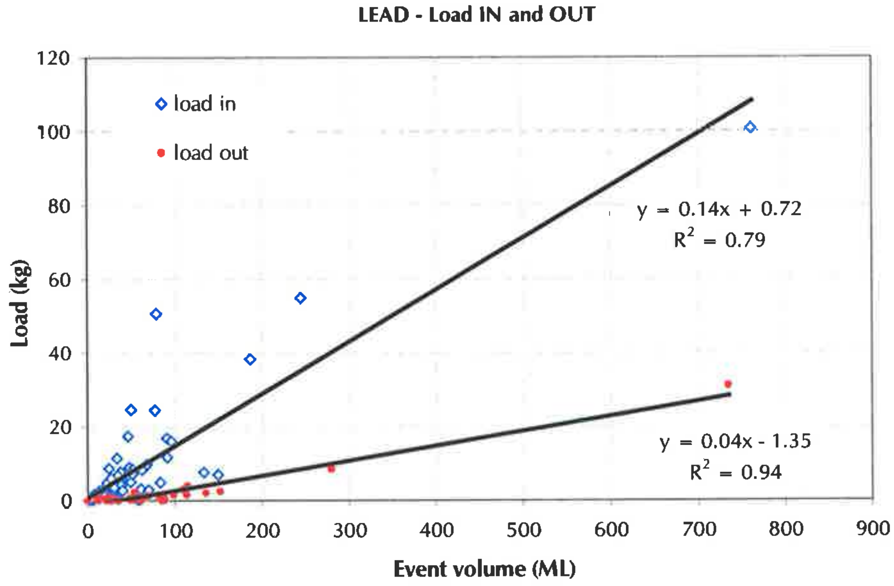


Figure 6.58 Lead load in and load out of BIW Pond 4 related to event volume

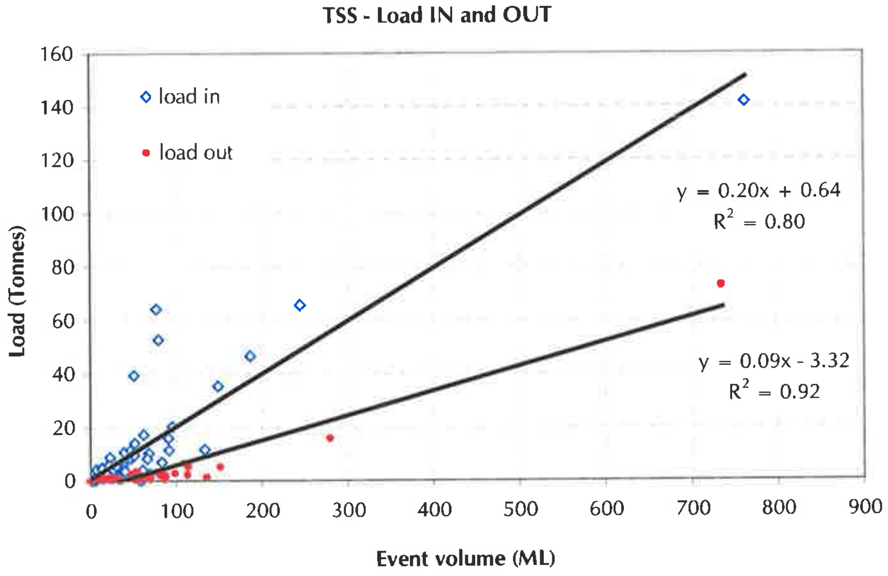


Figure 6.59 **Total suspended solids load in and load out of BIW Pond 4 related to event volume**

The plots give an indication of the loads that might be expected to enter and exit Pond 4 for a given storm volume. These relationships can further be used to develop a single line graph, which relates percent removal of pollutants to normalised event volume. Normalised event volume simply expresses the event volume in terms of pond size. For example, for a 10 ML pond, an event of 20 ML would represent a normalised storm size of 2. Normalising the event size makes the relationships easy to apply to other similar situations.

The removal of total phosphorus, total lead, and total suspended solids within Pond 4 is shown as a function of normalised storm size in Figure 6.60.

Figure 6.60 shows that for event volumes up to approximately four times the pond volume, the degree of pollutant removal is significantly influenced by the size of the event. Past this point the performance curve flattens out and indicates that a similar amount of treatment will be obtained regardless of the treatment time. While the results collected at the Barker Inlet Wetland site to date suggest this, some caution should be exercised when using the curves for storm sizes in excess of three or four times the pond volume. Only one storm has been monitored outside this range and further data collection is really needed to verify the results.

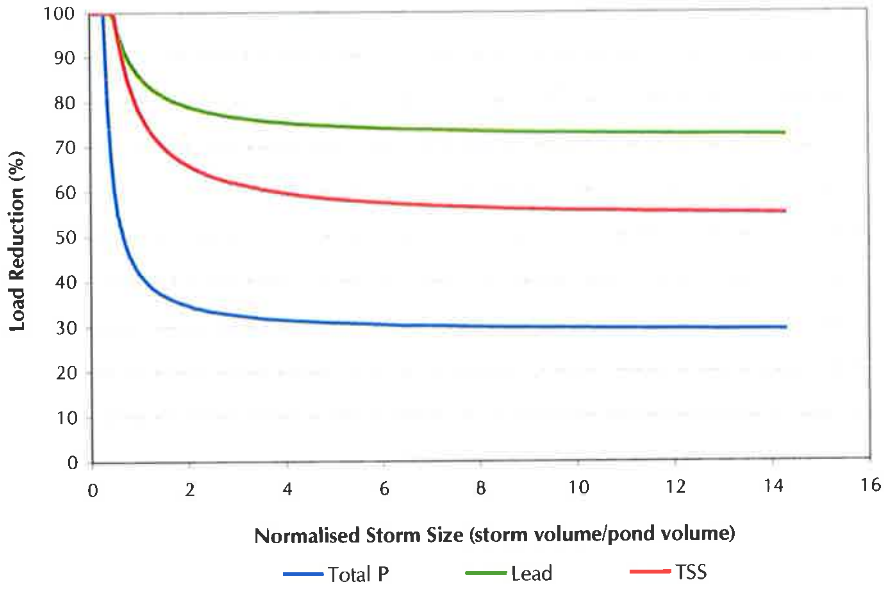


Figure 6.60 **Predicted trends in reduction of Total P, lead and TSS loads within BIW Pond 4 related to normalised storm size**

The effect of storm size on water quality improvement has been discussed by other authors such as Lawrence (1986) and Tomlinson *et al.* (1993) who related performance to residence time. This is in fact just an expression of event size (larger events produce shorter residence times). Findings of this study are consistent with those reported by the aforementioned authors who found that performace decreases with decreasing residence time.

6.6.2 Effect of initial concentration

In addition to event volume playing a role in determining pond performance, the effect of initial concentration has been noted by authors including Duncan (1995a) and Randall *et al.* (1982).

Duncan (1997) stressed the importance of reporting initial concentrations when quoting treatment efficiencies in storage as the removal is often highly dependent on the initial concentrations. Randall *et al.* (1982) explain that higher input concentrations tend to lead to a higher removal *rate*. This is likely to be because these higher concentrations are typically a result of the presence of larger particles that settle out more rapidly (Ferrara & Witkowski, 1983).

Randall *et al.* (1982) conducted a series of settling column experiments to determine the efficacy of sedimentation in the removal of stormwater pollutants. Figure 6.61 demonstrates the effect of initial TSS concentration on removal of sediment as found by Randall for 6-hour sedimentation time.

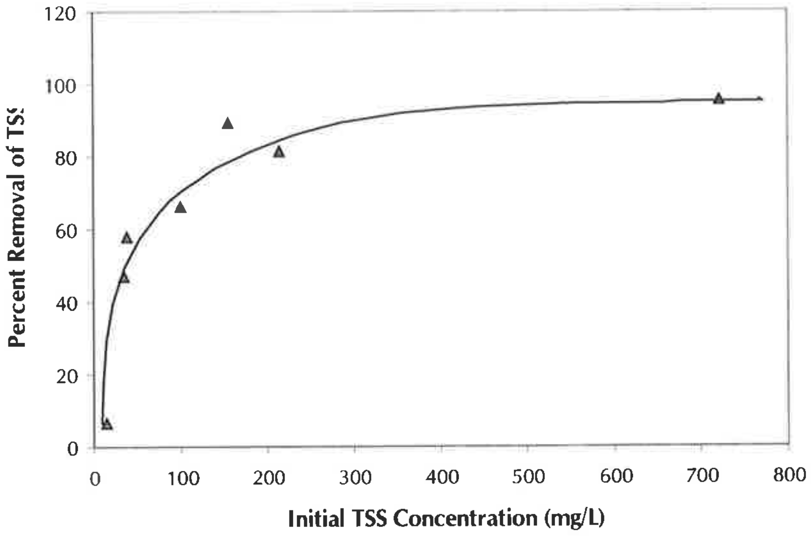


Figure 6.61 **The effect of particle concentration on TSS removal by sedimentation from study by Randall *et al.* (1982), based on 6-hour sedimentation time**

The effect found during the study of Pond 4 was a better match to the 12-hour sedimentation results of Randall as seen in Figure 6.62.

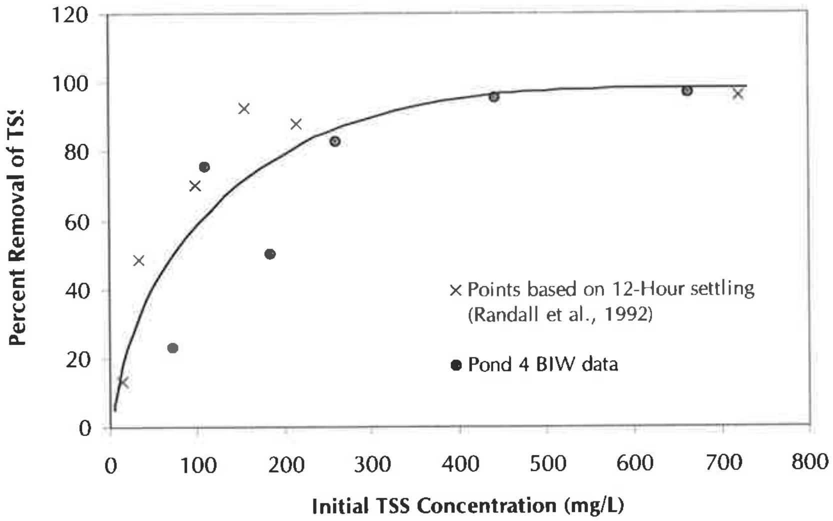


Figure 6.62 **The effect of initial concentration on TSS removal from Barker Inlet Wetland (Pond 4), compared to 12-hour sedimentation results of Randall *et al.* (1982)**

The results show that Pond 4 of the Barker Inlet Wetland is not quite as efficient, which may be due to a number of reasons. Firstly, Pond 4 may experience residence times less than12 hours, which is quite likely during large events, but not so likely during smaller events. Secondly, and perhaps more importantly, the conditions are probably not as ideal for settling as those experienced in the settling column used for the Randall experiments.

Another reason may be the particle size of the sediment entering the pond. Sediment from the Barker Inlet catchment may in fact be finer than the sediment used by Randall, and thus will have a slower settling velocity. The average particle size distribution for the North Arm East catchment is shown in Figure 6.63 as determined by analysis in a Malvern Mastersizer.

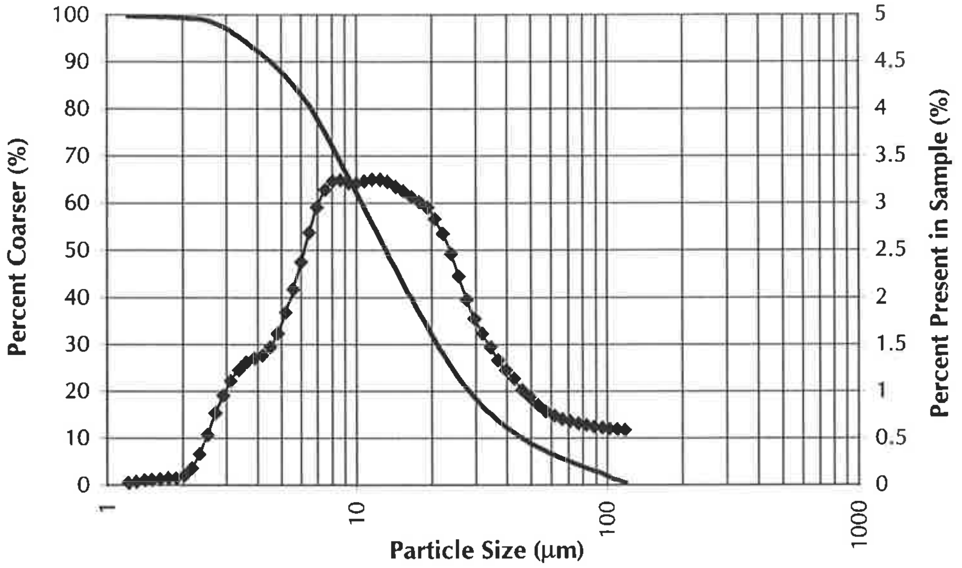


Figure 6.63 Average particle size distribution for North Arm East inflow station (during event 28th July 1998).

Wong and Walker (1998) collated data relating to the particle size distribution of sediment in street runoff derived from various researchers in a number of locations both in Australia and overseas. They found the sediment derived from Australian catchments to be finer than most data. Although the quantity of Australian data was limited, a distinct difference in sediment characteristics was noted. A study by Ball and Abustan (1995) found that 70 percent of sediment sampled from runoff in Sydney was finer than 62 μm, while studies of European and American catchments indicated that typically only 20 around percent was less than 100 μm. Lloyd *et al.* (1998) stated that while it is true that the results of Ball and Abustan (1995) are not necessarily representative of the sediment characteristics of all Australian capital cities, *"one would expect sediments in runoff from Australian catchments to contain a greater proportion of fine graded particles than for northern hemisphere countries."*

Randall *et al.* (1982) concluded from their study that while much of the TSS were removed during the theoretical settling time, in all experiments a large number of fine particles remained suspended in the water column at the end of the period. While substantial removal of these fine particles did occur, he attributed this to agglomeration, and found that it occurred at a much slower rate. The fine particles are of considerable concern from the viewpoint of water quality because, as has already been discussed, it is the fine particle fraction (< 20μm (French, 1999)) which provides for the attachment of other pollutant species (Randall *et al.*, 1982).

The study by Randall also found that total nitrogen removal was related to the initial TSS concentration. Removal was substantially higher for initial TSS concentrations in excess of 100 mg/L, and was also influenced by the initial nitrogen concentrations. No correlation between initial TN concentration and removal efficiency was found during this study.

6.7 Pond Performance In Perspective

The following sections compare the performance of the pond to other studies of water pollution control ponds. Reference is also made to the ANZECC Australian Water Quality Guidelines (ANZECC, 1992) to determine whether or not Pond 4 is acting to reduce pollutants to the Guidelines recommended for protection of aquatic ecosystems.

6.7.1 Comparison to ANZECC Guidelines

The following section details a comparison between the results of this study and the Australian Water Quality Guidelines (ANZECC, 1992) for protection of aquatic ecosystems. It has been noted in Chapter 2 that current guidelines do not reflect the bioavailability of pollutants, and with this in mind, this comparison is not intended to give any indication of ecosystem health. It is simply given to put into perspective the magnitudes of the concentrations of each of the parameters monitored to give the reader some indication as to whether the concentrations might be significant or not.

For information on the toxicological implications of the deposited sediment in the Barker Inlet Wetland system the reader is referred to literature by Jenkins *et al.* (1998) and Gorrie *et al.* (1998).

Figure 6.64 shows the comparison between the South Road Connector outflow concentrations (flow weighted mean annual) and the ANZECC guidelines for the protection of aquatic ecosystems.

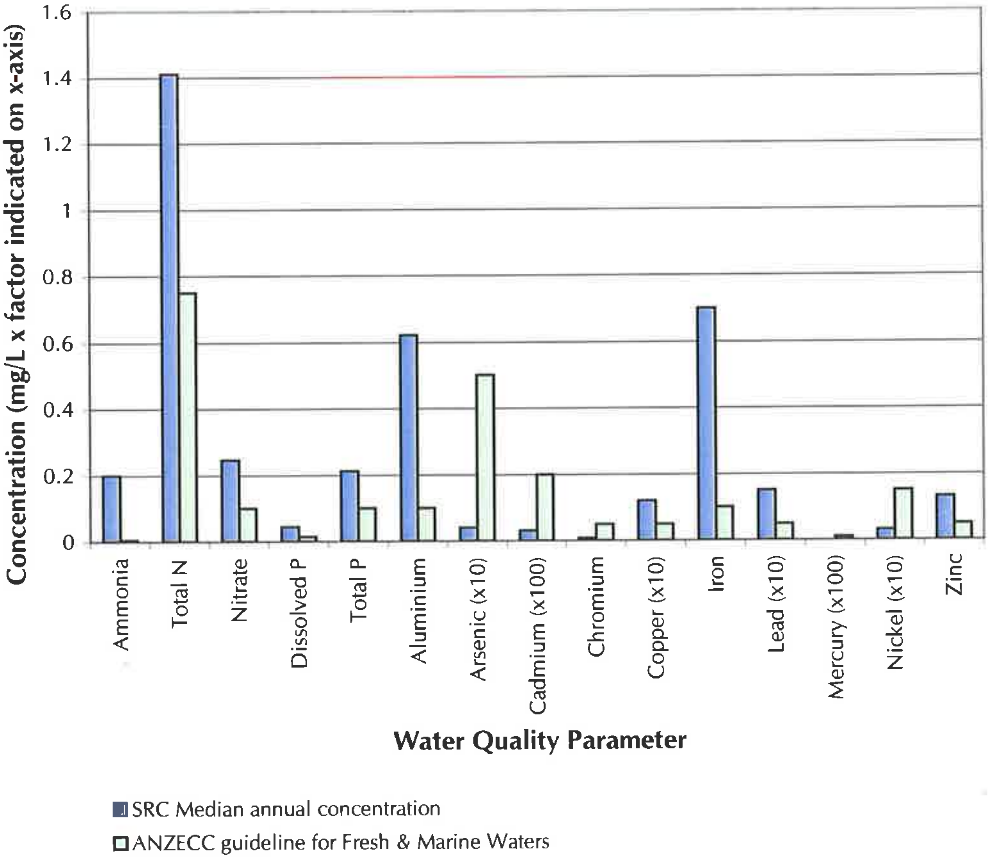


Figure 6.64 **South Road Connector outflow annual median concentrations compared to ANZECC Guidelines for the protection of aquatic ecosystems**

As Figure 6.64 shows, ten of the fifteen parameters displayed are above the recommended guidelines. Some parameters, for example, ammonia, aluminium, and iron are significantly higher.

6.7.2 Comparison With Other Studies

The NURP study was one of the most comprehensive studies of urban stormwater runoff carried out in the United States, and involved the comprehensive monitoring of nine pollution control ponds of differing types. The general findings of the study were the following:

- > 90% removal of TSS and lead
- 65 % removal of TP
- 50 % removal of BOD, COD, TKN, copper and zinc

While Pond 4 did not achieve greater than 90 percent removal efficiency, the removal of lead and total suspended solids were the highest obtained for all the water quality parameters. Copper and zinc removal was slightly higher, while nutrient removal was significantly lower.

The report by Duncan (1997) presented a "statistical overview of urban stormwater treatment by detention in on-stream storage." The author compared and analysed the results of investigations at approximately 50 separate locations in four countries, reported in urban stormwater quality literature.

Duncan classified eleven water quality parameters into three groups based on their behaviour in storage - a settling group, a proportional group, and a rate-limited group. He found that for the settling group, which included suspended solids, total lead and total zinc, the output concentration was roughly proportional to the square root of the area ratio. For the proportional group, made up of COD, dissolved and total phosphorus, and all forms of nitrogen except oxidised nitrogen, the output concentration was proportional to the input concentration and decreased very slowly as the area ratio increased. For the rate-limited group, which consisted solely of oxidised nitrogen (nitrate plus nitrite), the output concentration was proportional to the input concentration to the power of 1.6, and decreased slowly as the area ratio increased.

The Barker Inlet Wetland Pond 4 has a surface to area ratio of approximately 0.5 % (catchment area 2200 ha, surface area 10ha) and has been shown in relation to the trends established by Duncan in Figure 6.65 (total phosphorus), Figure 6.66 (lead) and Figure 6.67 (total suspended solids).

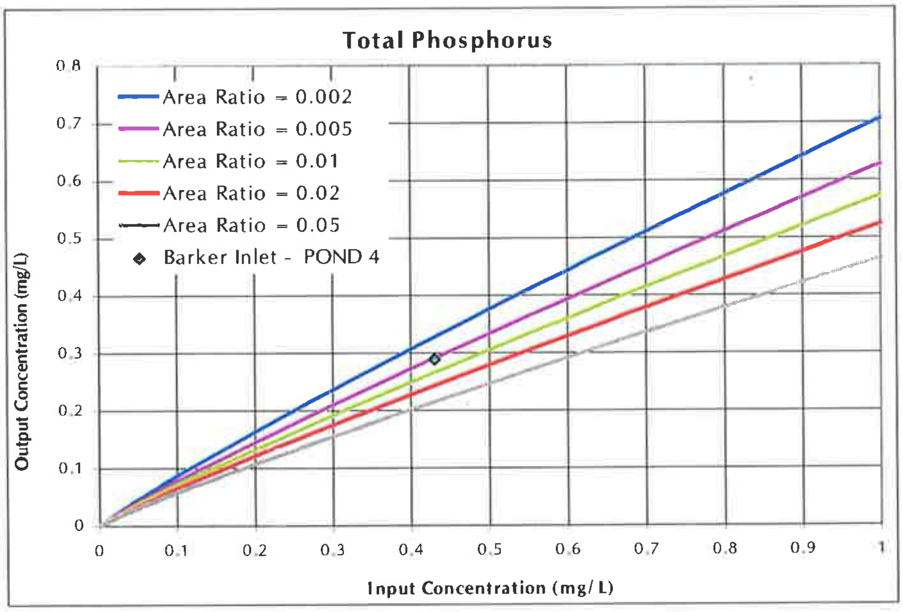


Figure 6.65 BIW Pond 4 performance (total phosphorus) compared to study on SAR by Duncan (1997)

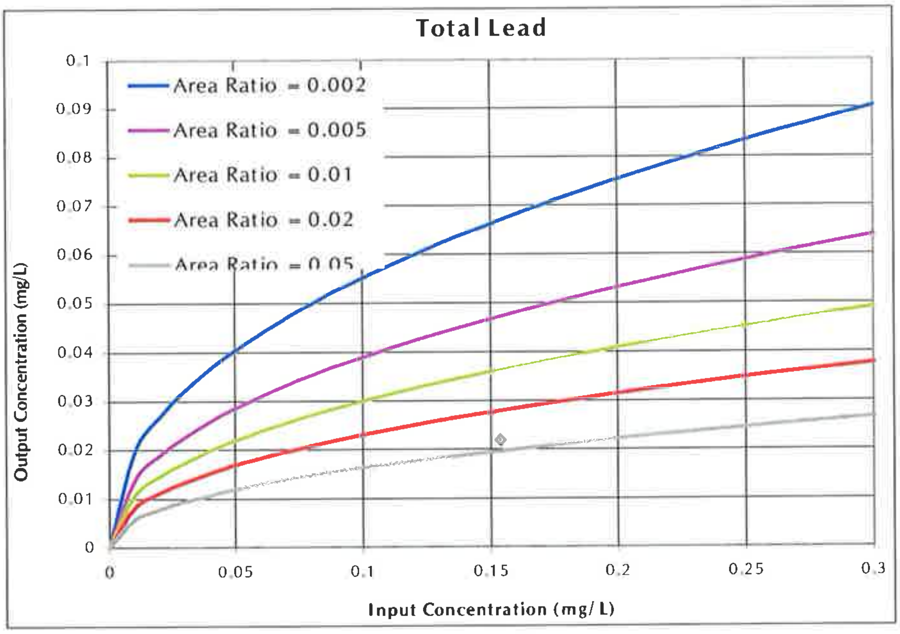


Figure 6.66 BIW Pond 4 performance (total lead) compared to study on SAR by Duncan (1997)

Figure 6.67 BIW Pond 4 performance (total suspended solids) compared to study on SAR by Duncan (1997)

As these figures show, there is little consistency in the performance of BIW Pond 4 compared to the combined results of world wide studies. The removal of total phosphorus (refer Figure 6.65) lies on the SAR=0.5 line, however the removal of lead within BIW Pond 4 (refer Figure 6.66) is closer to the efficiency for a SAR of five percent. Total suspended solids removal efficiency lies somewhere between a SAR of one and two percent. It is not surprising that the Barker Inlet results do not lie exactly on the 0.5 percent line as much variability was encountered in the analysis of Duncan (1997).

The Paddocks Wetland in South Australia is located reasonably close to the Barker Inlet Wetlands and therefore has some similarities in terms of rainfall patterns and catchment characteristics. The sampling methods used in the study (Tomlinson *et al.*, 1993) were also similar which may allow a comparison to be made. It should be noted however, that the Paddocks system is quite a bit smaller and is fed from a much smaller catchment. The Paddocks Wetland also has a grassed swale drain upstream of the Wetland inlet.

Table 6.13 compares the Paddocks removal efficiencies to those obtained from the study of Pond 4. The paddocks removals have been calculated from a flow weighted event mean concentration for the entire study period. This is the same as comparing inflow and outflow loads provided the inflow and outflow volumes are the same.

Table 6.13 Comparison between removals experienced at the Paddocks Wetland and Barker Inlet Wetland Pond 4.

Water Quality Parameter	Paddocks (1)	BIW Pond 4
Ammonia	39	6.8
Nitrate	75	43.4
TKN	32	11.2
TN	43	15.9
Filt Reactive P	54	0.2
TP	62	34.0
Cadmium	33	54.5
Chromium	-	55.6
Copper	> 92	65.8
Iron	12.5	66.9
Lead	89	85.7
Manganese	-42	24.3
Mercury	-	13.3
Nickel	37.5	33.3
Zinc	93	70.5
TSS	88	75.0

(1) Tomlinson et al. (1993)

While the Barker Inlet Wetlands were established in 1994, the Paddocks Wetland has been in operation since 1975 (Tomlinson *et al.*, 1993). The vast difference in the maturity of the two systems, and in particular the aquatic vegetation which play an important role in water treatment, is reflected by these results. The comparison indicates that the Paddocks Wetland achieves a higher degree of nutrient removal, while the removal of metals and TSS, achieved predominantly by sedimentation, is comparatively similar. It is expected that as the Barker Inlet Wetland system matures a higher removal efficiency will be achieved.

The removal efficiencies reported in the literature examined, which were summarised in Table 2.4, have been graphed (see Figure 6.68) to show the relative performance of Pond 4. The vertical bars show the maximum and minimum values reported, and the balck dots show the median of the data set examined. The performance of Pond 4 is shown by the blue dots.

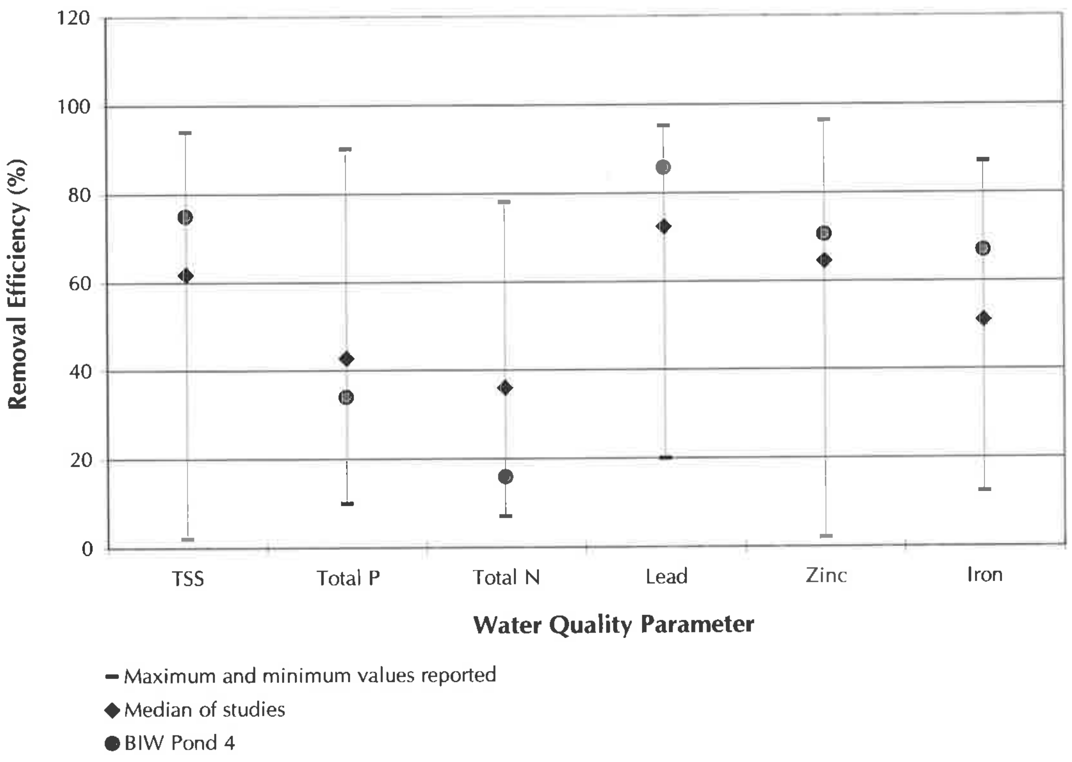


Figure 6.68 **Maximum, minimum and median removal efficiencies reported in literature compared to annual performance of BIW Pond 4**

Figure 6.68 demonstrates that the results obtained from the study are typical of those found elsewhere. Some of the pollutants experienced removals above the median of the data set, while others were below. Not surprisingly, the lower removals were obtained for the nutrients nitrogen and phosphorus, a reflection of the immaturity of the system. Better than average removals were experienced for the particle bound pollutants (TSS, lead, zinc and iron) probably as a result of the very long flow path through the pond.

7

Water Quality Computer Modelling

The benefit of water quality computer modelling was highlighted in Chapter 2. It enables the prediction of water quality behaviour, without the need for comprehensive water quality monitoring which can be both time consuming and expensive. Many models are available today which have been calibrated and successfully applied to various situations in Australia and other regions of the world. Despite this, each individual catchment is unique. Sufficient data is therefore necessary to calibrate models for each situation to which they are to be applied in order to maintain the models' integrity.

Data collection has been carried out at the Barker Inlet Wetland site for approximately three years and now represents a database sufficient for model calibration. While estimates of the performance of Pond 4 have already been made for the 12-month period discussed in the previous chapter, application of a model able to predict pond performance will enable continuing estimates of performance to be made without the need for further water quality monitoring.

The model chosen for the purposes of this study, as discussed in Section 2.6.3, is the Pond Water Quality Model developed by the CRC for Fresh Water Ecology. The input requirements of this model are simply daily inflow and daily concentration for a range of pollutants. Daily inflow can be calculated using rainfall data, and a method of predicting daily inflow concentration for Pond 4 was described in Section 6.4.1.

7.1 Model Background

Research undertaken by the CRC for Freshwater Ecology has indicated that ponds and wetlands comprise a number of compartments, with transfers between these compartments occurring as a result of the physical, chemical, biological and microbial processes supported by the pond environment. The dominant pollutant interception and transfer processes have been identified making it possible to model the responses of wetlands and ponds to certain inflow conditions.

A model was developed by the CRC for Freshwater Ecology in 1998, which described the dominant transfers and transformations of pollutants in terms of physical, chemical, biological and microbial thermodynamics. Other compartments within the model structure simulate the factors which drive the transfers and transformations.

A number of modifications were made to the model during the course of this study that gave improved results in some instances, and worse results in other instances. The results from two versions of the model are given in the following sections. The versions used are the original release version (1998) and the current release as of September 1999.

7.2 Model Structure

Two models were developed by the CRCFE, a pond model and a wetland model. These models are comprised of several sub-models, which are illustrated in Figure 7.1. Each sub-model is represented by a separate worksheet in the EXCEL spreadsheet described below. Only the pond model was used in this study.

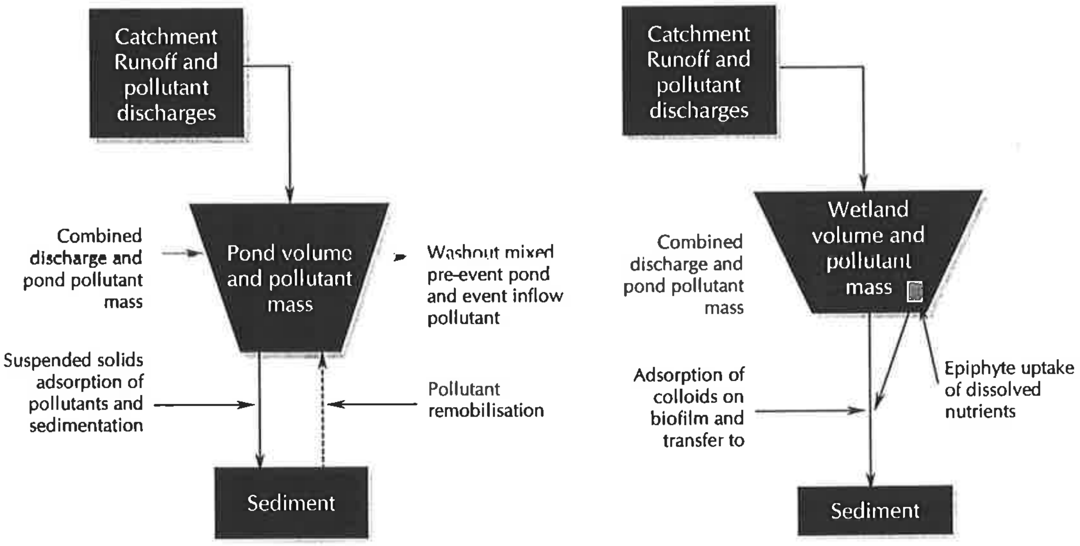


Figure 7.1 **Major components of pond and wetland pollutant washout, retention, interception and remobilisation processes**

(Source: Lawrence & Breen, 1998)

7.2.1 Initialisation (Worksheet A)

Worksheet A contains the information necessary for the initialisation step which precedes the running of the sub-models. The user must specify the physical characteristics of the pond, the initial or in-pond water quality, the sediment characteristics, and the sediment redox properties. Information required is listed below:

Physical Parameters:	Pond volume
	Pond surface area
	Average pond depth
	Pond transverse side slopes*
	Spillway equivalent length*
Initial in-pond water quality:	Total suspended solids
	Total phosphorus
	Total nitrogen*
	Dissolved oxygen
	Nitrate-N
	Temperature (°C)
	BOD
	Chlorophyll-a
Sediment grading (%):	$> 50\ \mu m$
	$15 - 50\ \mu m$
	$4 - 50\ \mu m$
	$< 4\ \mu m$
Sediment Composition:	Fe (III)
(% sediment weight)	SO_4
Initial sediment BOD	SOD/day
& aeration conditions:	BOD initial
	wind speed

* *indicates this parameter was not included in first release of model*

7.2.2 Water and Constituents Budgets Sub-Model (Worksheet B)

In this sub-model the user is required to enter daily inflow and water quality for the pond or wetland. Daily budgets track the changes in mass of constituents of the inflows to, and the outflows from the pond water column by assuming a Continuously Stirred Tank Reactor (CSTR) system. Water in the pond in the CSTR system is assumed to be fully mixed over the course of the day.

This aspect of the model underwent significant modifications during the time over which this research was carried out.

In the original model, calculations in the spreadsheet were based on an expansion of the mass balance relationship, described by Equation 7.1, which applied the daily inflow, Q, in five incremental (0.2Q) steps.

$$C_{pd}' = C_{pd} + {}^{0.2Q}\!/_{V}\left(C_{in} - C_{pd}\right)$$ **Equation 7.1**

where: C_{pd} = Pollutant concentration of pond water at start of period
C_{in} = Inflow pollutant concentration (mg/L)
Q = Daily inflow (ML/day)
V = Pond Volume (ML)

The expansion was as follows:

$$C_{pd}^{1} = C_{pd} + 0.2Q/V(C_{in}-C_{pd})$$
$$C_{pd}^{2} = C_{pd}^{1} + 0.2Q/V(C_{in} - C_{pd}^{1})$$
$$C_{pd}^{3} = C_{pd}^{2} + 0.2Q/V(C_{in} - C_{pd}^{2})$$
$$C_{pd}^{4} = C_{pd}^{3} + 0.2Q/V(C_{in} - C_{pd}^{3})$$
$$C_{pd}^{5} = C_{pd}^{4} + 0.2Q/V(C_{in} - C_{pd}^{4})$$

To give:

$$C_{pd}^{5} = C_{pd} + 5R(C_{in} - C_{pd}) - 10R^{2}(C_{in}-C_{pd}) + 10R^{3}(C_{in}-C_{pd}) - 5R^{4}(C_{in}-C_{pd}) + R^{5}(C_{in}-C_{pd})$$

where: $R = 0.2Q/V$ **Equation 7.2**

This step-wise based CSTR washout algorithm used in the original model was only accurate for cases where inflow volumes were less than the pond volume. This was replaced by an exponential algorithm, providing convergence for higher inflows.

$$C_{pd}' = C_{pd} + \left(1 - \frac{1}{\exp\left({}^{Q_{in}}\!/_{V_{pd}'}\right)}\right)\left(\frac{V_{pd}'}{V_{pd}''}\right)\left(C_{in} - C_{pd}\right)$$ **Equation 7.3**

where: C_{pd} = pond pollutant concentration at start of period
C_{pd}' = pond pollutant concentration at end of period
Q_{in} = pond inflow during period
V_{pd}' = pond volume during period
V_{pd}'' = pond volume during following period
C_{in} = inflow pollutant concentration

The model also computes losses and gains on a daily basis, these being transfers between the water column and sediment compartments, between the water column and the algal compartments, and between the water column and the atmosphere.

7.2.2.1 Extended detention basin analysis

Another new feature of the updated model was the incorporation of an extended detention basin analysis. This was to account for the increased surface area and volume associated with storm event inflows. A broad crested weir algorithm was included to inter-relate inflow, pond water level, volume and surface area. Transverse side slopes of the pond and a spillway length are entered into the Initialisation spreadsheet.

The model undertakes a daily computation of pond drawdown for the days following an event. The discharge, volume, depth and surface area prevailing each day are used in the washout, sedimentation, BOD loading and remobilisation sub-models rather than the average conditions used in the previous version of the model.

7.2.3 Adsorption and Sedimentation Sub-model (Worksheet C)

Lawrence and Breen (1998) report that in urban stormwater pollution control ponds, a majority (up to 95%) of exports occur during storm events with runoff characterised by high levels of suspended solids. CRC research has supported other reports that the fine suspended solids have a high adsorption capacity for nutrients, metals, organic material, and bacteria. An important response to runoff within the pond system is therefore the adsorption of pollutants to suspended solids and the physical settling of the particles.

Particulates found in urban stormwater have a substantial capacity to adsorb dissolved pollutants. This adsorption often takes place very rapidly, within seconds or minutes of contact (Lawrence & Breen, 1998). As a large amount of particulate matter settles out under quiescent conditions, the degree of removal of adsorbed pollutants will depend on size of particles to which they are attached and the sedimentation efficiency of each size fraction. The CRCFE has analysed a range of particulates to determine what percentage of pollutants are attached to the various particle sizes. Based on current knowledge:

$$TP = 0.7d^{-0.2}$$

$$TN = 11d^{-0.2}$$

Where TP and TN are the mass (in μg) of nutrients adsorbed per gram of suspended matter and d is the particle diameter in μm.

The rate of settling of suspended solids is dependent on the particle size and the pond eddy diffusion conditions. The settling velocity is calculated in the model using Stokes Law, identified in Equation 7.4 (Vanoni, 1977).

$$V_s = \frac{gD^2}{18\nu}\left(\frac{\gamma_s - \gamma}{\gamma}\right)\eta_s$$

Equation 7.4

Where: V_s = settling velocity (m/s)
g = acceleration due to gravity (9.81 m/s^2)
D = particle diameter (m)
ν = kinematic viscosity (m^2/s)
γ_s = specific weight of particle
γ = specific weight of fluid
η = settling efficiency

The medium silt and larger particles usually settle within a few hours to two days, while most of the finer material will settle within 10 to 20 days. There is a loss of settling efficiency associated with small eddies and currents induced by flow, wind and thermal gradients, with non-spherical particles, and with short circuiting of the available storage capacity (Lawrence & Breen, 1998). The sedimentation efficiency, or η term, in Equation 7.4 is defined in Table 7.1 and becomes more important for the finer particles.

Table 7.1 Sedimentation efficiency and specific weight as a function of particle size

(Source: Lawrence & Breen, 1998)

Particle Size & Description (µm)		**Specific Weight**	**Settling Efficiency (%)**
10500	course sand	2.6	100
250	medium sand	2.5	100
170	fine sand	2.5	90
90	very fine sand	2.5	90
50	medium silt	2.3	90
15	fine silt	2.0	80
5	clay	1.7	70
0.7	colloids	1.7	60

The sedimentation sub-model calculates the fraction of pollutants attached to each of the sediment sizes and the loss of each particle size from the water column.

The proportion of each of the size fractions removed from the water column is calculated on a daily basis. For the medium silt it is assumed that there is a 95 percent removal efficiency, as described by Equation 7.5.

$$L = 0.95\ C_{pd}V \qquad \textbf{Equation 7.5}$$

Where: L = load sedimented (kg)

The load for the other fractions is computed by Equation 7.6.

$$L = C_{pd}V\left(1 - \frac{1}{0.5V_s \frac{A}{Q^{0.5}}}\right) \qquad \textbf{Equation 7.6}$$

where: Q = daily flow (ML)

A = pond area (ha)

C_{pd} = pond concentration (mg/L)

V = pond volume (ML)

In addition to TSS, the degree of sedimentation of total phosphorus, total nitrogen and BOD are also calculated in this way.

7.2.4 Sediment Reduction and Oxidation Sub-model (Worksheet D)

Stormwater runoff is typically high in organic material, and as it settles out in the pond it can impose a significant biochemical oxygen demand on the sediments (Lawrence & Breen, 1998). For a period of time after a storm event the water will be well mixed and oxygen will be transferred readily from the atmosphere to the sediments, off setting the depletion of oxygen by bacteria growth (a response to the organic matter carbon source). During quiescent conditions (between storm events) however, the pond may stratify (particularly when solar radiation is high) and hamper the transfer of O_2 to the sediments. It is under these conditions that O_2, cations and anions may be reduced in the pond sediments leading to the re-mobilisation of nutrients and, in particular, metals in a highly bio-available (soluble) form. Although thermal stratification is unlikely to occur in the shallow water of the Barker Inlet Wetland Pond 4 (maximum depth ~ 2m) the model tracks the sediment redox conditions.

The sediment redox sub-model calculates the daily BOD based on the sedimentation of organic matter from the sedimentation sub-model, and the depletion of O_2 and nitrate followed by reduction of ferric iron and sulphate. As reduction proceeds, either all the organic carbon is used up resulting in the cessation of BOD growth, or the aeration rate increases to a level that exceeds the BOD rate. Under either of these conditions, re-oxidation of sediments begins which leads to:

- oxidation of Fe^{2+} to Fe^{3+} causing precipitation of phosphate (PO_4);
- oxidation of hydrogen sulphide (H_2S) to SO_4^-;
- oxidation of ammonium (NH_4^+) to nitrate NO_3^-; and
- more dissolved oxygen in the water column.

The daily release of nutrients resulting from the transformations are calculated by the model along with oxidation of Fe^{2+} and SO_4^- and precipitation back to the sedimentation sub-model via a molecular diffusion equation.

7.2.5 Algal Growth Sub-model (Worksheet D)

The sediment redox computations also include an algal growth sub-model component. As mentioned above, if sediments undergo chemical reduction they release nutrients (ammonium and phosphate) back into the water column in a highly bio-available form for algal uptake (Lawrence & Breen, 1998). The algal model assumes that the release of nutrients is the dominant nutrient pathway sustaining algal growth and that the growth is a reflection of either the daily release of phosphorus (Equation 7.7), or the doubling rate coefficient (Equation 7.8), which ever is smaller.

$$\text{Algal biomass(chlorophyl-}a) = \Delta P \times 0.5 \times \frac{1}{d} \times 0.7$$

Equation 7.7

where: ΔP = release of phosphorus from sediments (mg/L)
0.5 indicates remobilisation across 50% of pond area
d = average depth of pond (m)
0.7 is the chlorophyll a to TP ratio

$$\text{Algal biomass(chlorophyl-}a) = y_o^{0.3n} / 1000$$

Equation 7.8

where: y_o = initial chlorophyll a value (μg/L)
n = an estimate of the number of dry days since the last storm inflow
0.3 increase in initial algal level per day for a doubling time of 5 days

The model assumes an algal decay rate of ten percent of the biomass/day, a loss that is linked to the BOD computation through its contribution to the net BOD. The algal growth sub-model also calculates the daily photosynthesis oxygen generation, which is linked to the computation of dissolved oxygen in the Water and Constituent Budgets sub-model (Worksheet B).

7.2.6 Mixing (Oxygen Transfer) Sub-Model (Worksheet D)

The oxygen transfer, or mixing, sub-model has four components.

1. Physical mixing of water column with uptake of O_2 at the water surface and transfer to sediments;
2. Physical mixing of water by inflow as a result of advective eddy diffusion forces;
3. Sediment O_2 demand plus daily microbial BOD demand in sediment due to microbial growth; and
4. Direct transfer of O_2 through macrophyte stems to rhizome and root zones.

The mixing sub-model determines the sediment aeration rates on the basis of:

- Local wind conditions;
- Volume of inflow relative to the pond volume; and
- Computation of O_2 transfer by macrophytes.

The total daily aeration is given by Equation 7.9.

$$W = 0.8 + 0.3u^{1.64} + 4V^{0.5}/d^{0.5}\times(C_{sat}-C_{pd}) \qquad \textbf{Equation 7.9}$$

where: W = the oxygen transfer rate in g/m^2/day
u = the diurnally averaged wind velocity (m/s)
v = the average flow velocity through the pond (m/s)
C_{sat} = the oxygen saturation concentration for the water temperature
C_{pd} = the concentration of oxygen in the pond water

7.3 Model Calibration

Site specific parameters were entered into the Initialisation worksheet (Worksheet A). Table 7.1 details the input values for Pond 4.

Flow and water quality data were entered into the initialisation worksheet (Worksheet B) using a combination of actual monitored values and estimated values based on the modelling approaches described in Section 6.4. As a first input, the regression equations developed in Section 6.4 were entered into the worksheet, and then for significant events where actual EMC data were available, alternative values were entered manually. Daily flows entered into the spreadsheet was based on a combination of actual event volumes calculated from rating tables, and modelling procedures developed by Daniell *et al.* (1999).

Table 7.2 Local values used in model calibration

	Parameter		Local Value
Physical Descriptors	Pond volume		69 ML
	Pond surface area		9.5 ha
	Average pond depth		0.73 m
	Pond transverse side slopes		1 in 8
	Equivalent spillway length		32 m
In-pond Water Quality	Suspended Solids		15 mg/L
	Total phosphorus		0.25 mg/L
	Inorganic nitrogen		0.4 mg/L
	Organic nitrogen		1.0 mg/L
	Dissolved oxygen		7.0 mg/L
	Nitrate-N		0.2 mg/L
	Temperature		15 °C
	Chlorophyll-a		0.01 mg/L*
	BOD		1.0 mg/L*
Sediment Grading	> 50 µm		9.0 %
	15-50 µm		33.9 %
	4-15 µm		48.6 %
	< 4 µm		8.5 %
Sediment Composition (% sediment weight)	Top layer	Fe^{3+}	2.0 %*
		SO_4	0.20 %*
	Second layer	Fe^{3+}	1.0 %*
		SO_4	0.10 %*
Initial Sediment BOD and Aeration Conditions	SOD/day		0.0 g*
	BOD initial		2.0 g*
	Wind speed		2.0 m/s*

* *default values used, no local data available*

A similar approach was taken when calibrating the output results. Regression equations were combined with actual monitored EMC values, which were consequently converted to export loads.

The model was run for the 12-month period 1st August 1997- 31st July 1998 and results were converted from daily pond concentration to daily load exiting the pond to allow the use of the regression equations developed in Section 6.4.1.

7.4 Model Results

Results for two versions of the model are presented in the following section. Results have been plotted as a time series graph and a scatter plot to demonstrate the relationship between modelled and monitored results.

7.4.1 Total Phosphorus

The initial version of the model made somewhat erratic predictions about the behaviour of phosphorus in the pond. The fluctuations in total phosphorus concentrations were a function of the particulate fraction, which subsequently settled out, and the dissolved fraction which could be bound to sediments or released from sediments under the right (or wrong!) conditions.

This erratic behaviour was traced to the sediment reduction and oxidation sub-model, which was predicting massive releases of dissolved phosphorus from the sediments. The problem identified was the modelling of BOD inflows to the pond. This parameter was not monitored during the study, and hence default values in the pond model were used. The default equation for predicting the BOD inflow concentration is given by Equation 7.10

$$C_{BOD} = 6 + 0.1 \times Q$$ **Equation 7.10**

where: Q = daily inflow to pond (ML)

This method of prediction produced spikes in the BOD concentration, which lead to the eventual release of phosphorus from sediments. The process by which this happens was described by Lawrence and Breen (1998) and is summarised below:

The chemical reduction of sediments is biologically mediated by heterotrophic bacteria, which use organic carbon from stormwater discharges as their energy source. As the populations grow, they consume oxygen, firstly from the water column and then from nitrate and nitrite. This chemical reduction produces ammonia under high reducing conditions. If organic carbon remains after all the DO is used up, the microbial populations continue to grow but now reduce the insoluble ferric iron (Fe^{3+}) to soluble ferrous iron (Fe^{2+}), and sulphate (SO_4^{2-}) to hydrogen sulphide (H_2S). Iron, along with manganese, is a key part of the formation of $FePO_4$ (↓) and iron hydroxides to which phosphorus and a range of heavy metals become fixed or attached as complexes (Lawrence and Breen, 1998). If the ferric iron becomes ferrous the phosphorus and metals are released. The reduction of sulphate releases sulphide which is one of the preferred anions for complexing the released metals in bioavailable form.

In the current release of the model, a new method of estimating BOD concentration, C_{BOD}, is provided as described by Equation 7.11.

$$C_{BOD} = 9 \times C_{TN}$$ **Equation 7.11**

where: C_{TN} = total nitrogen inflow concentration (mg/L)

This method provided much more reliable results, as demonstrated in Figure 7.2. This graph shows that the two lines, each representing daily load, are relatively close, however, Figure 7.3 gives a clearer understanding of the accuracy of the model.

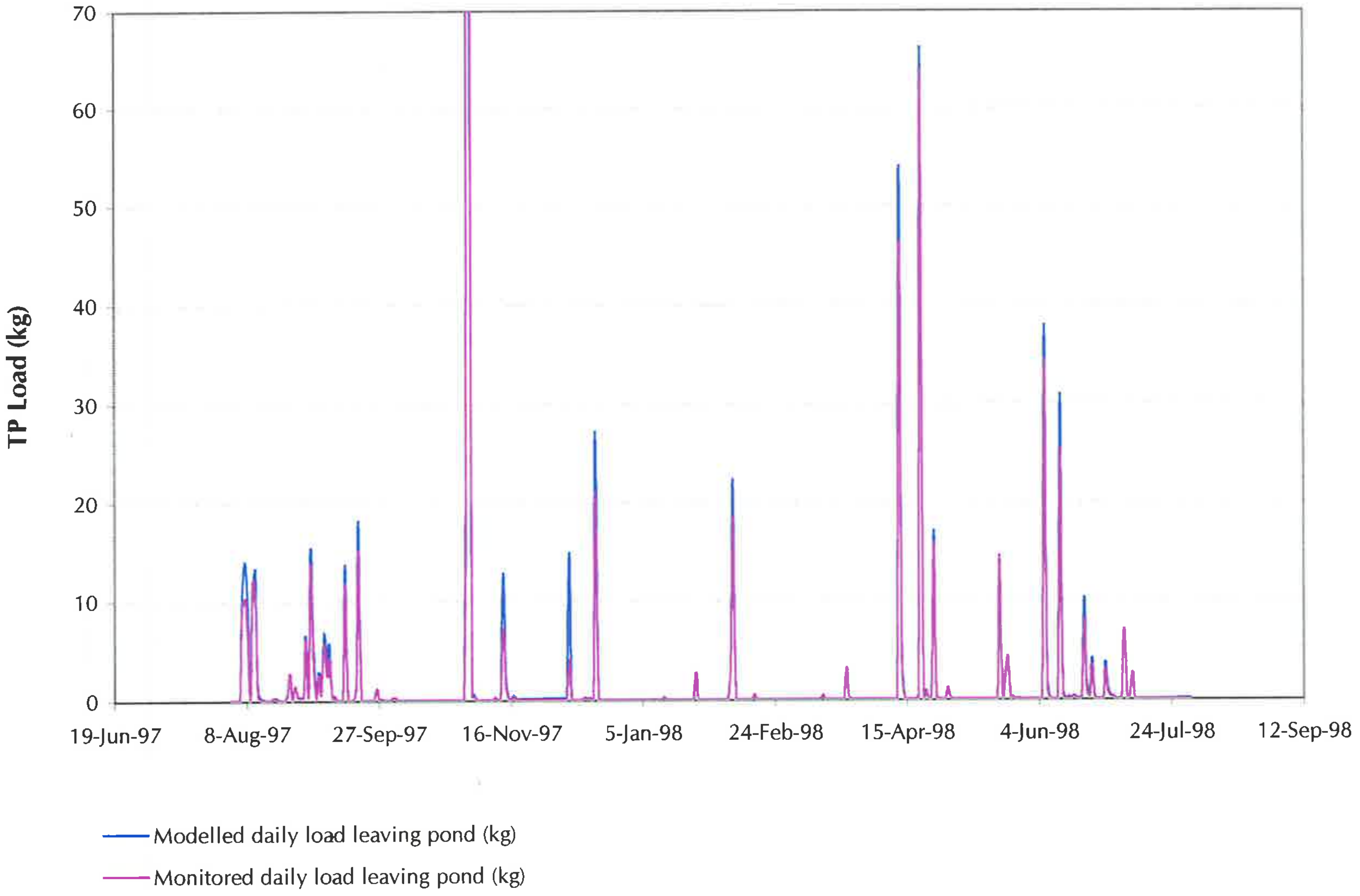


Figure 7.2 OLD MODEL: Comparison of monitored (pink) and modelled (blue) daily total phosphorus loads exiting the pond

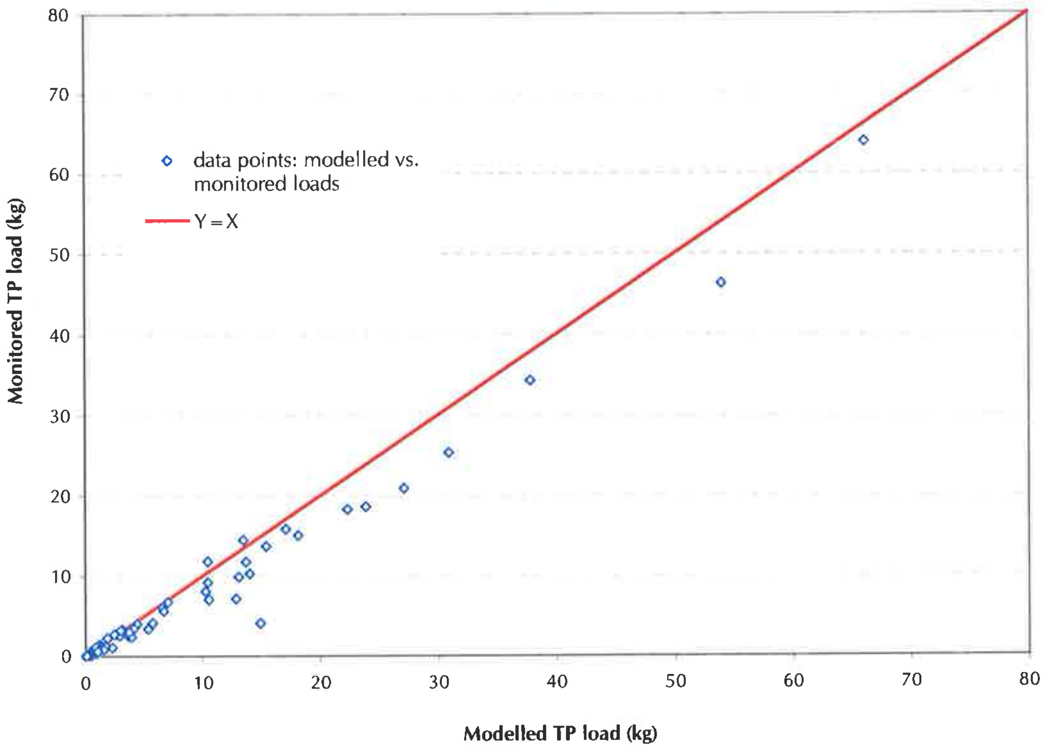


Figure 7.3 **OLD MODEL: Monitored vs. modelled daily total phosphorus loads exiting the pond compared to y = x line**

Figure 7.3 indicates that almost a 1-to-1 relationship was achieved between the monitored and modelled daily export loads.

In addition to the modifications made to the model already described, modifications were made that directly affected the prediction of phosphorus behaviour.

The extended detention basin analysis performed by the current model has been incorporated to account for the increased pond volume and surface area during storm event inflows. Due to the alternative wetting and drying of the sediments, there is a significant reduction in the nutrients available for remobilisation (Lawrence & Breen, 1998). This addition to the model has provided significant improvements in pond pollutant interception. This is demonstrated by Figure 7.4 and Figure 7.5.

Although the new version of the model predicted a linear relationship between the modelled and monitored values, it was not a 1-to-1 relationship, as demonstrated by Figure 7.5.

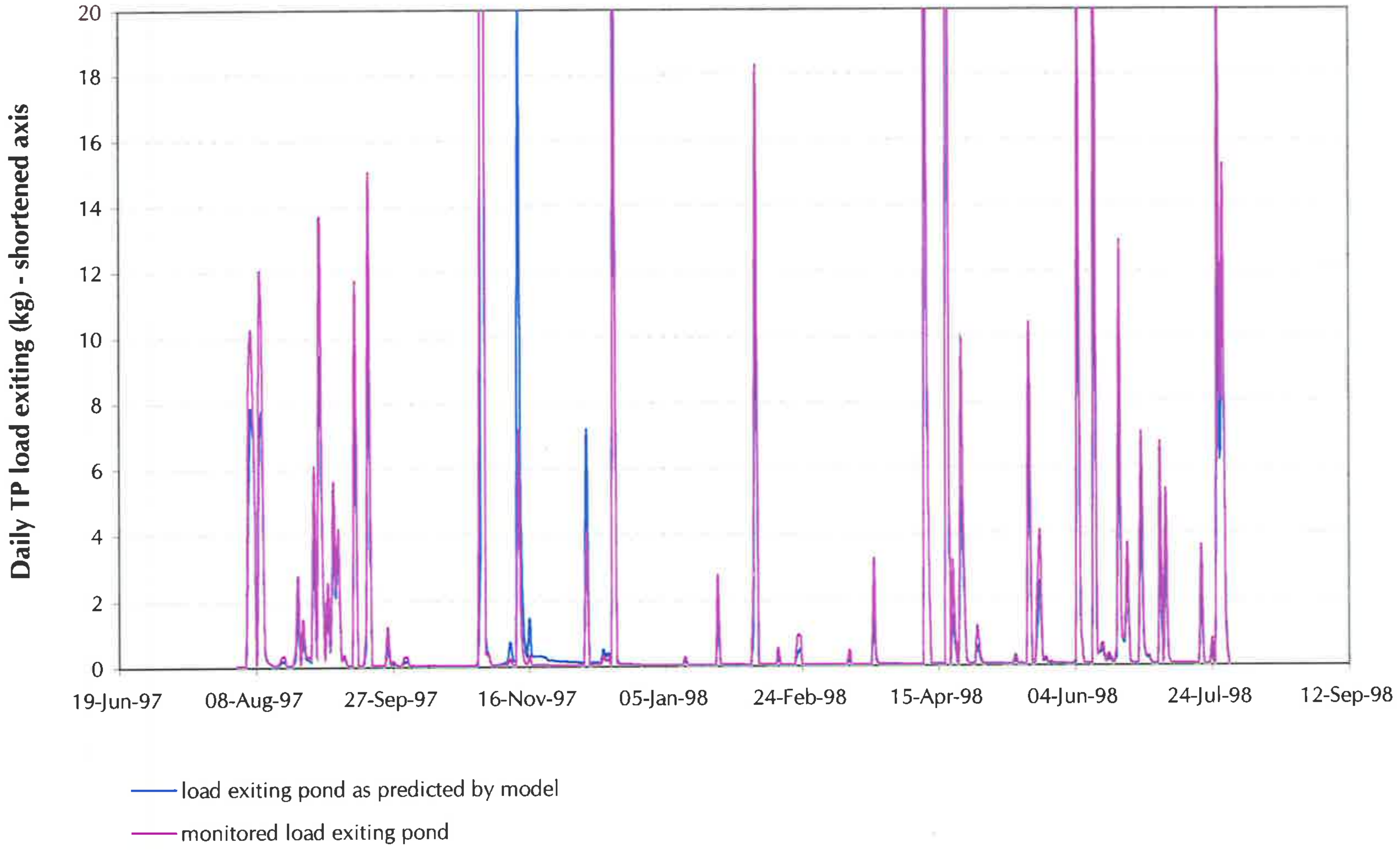


Figure 7.4 NEW MODEL: Comparison of monitored (pink) and modelled (blue) daily total phosphorus loads exiting the pond

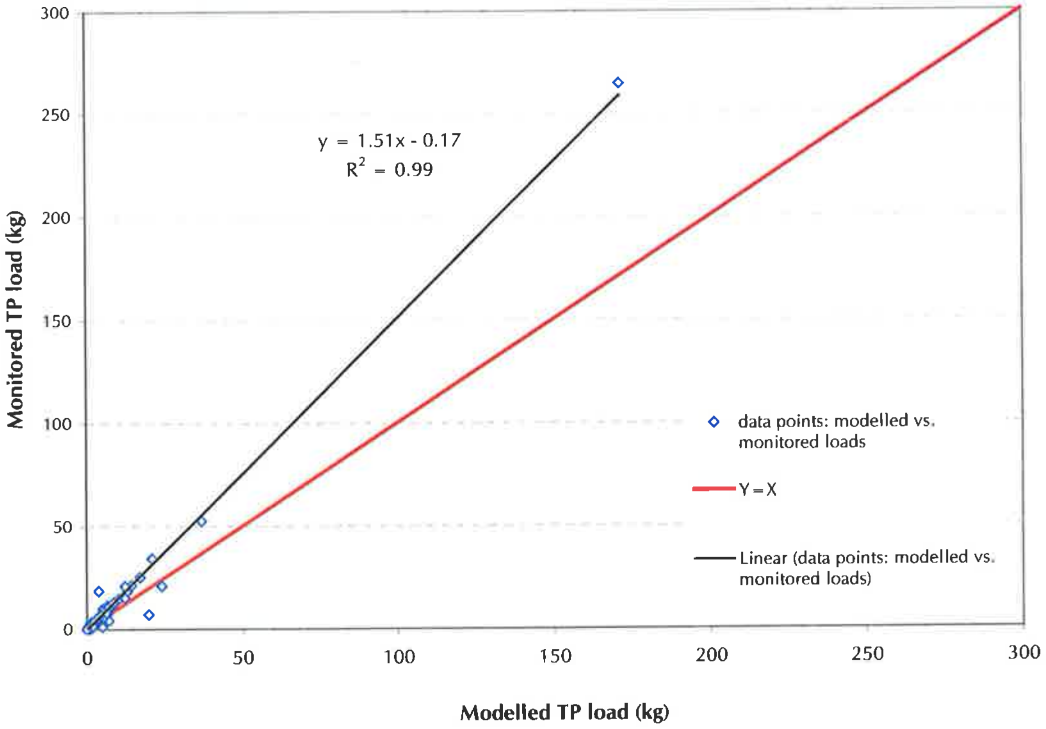


Figure 7.5 **NEW MODEL: Monitored vs. modelled daily total phosphorus loads exiting the pond compared to y = x line**

The initial version of the model thus proved to have the better predictive capability with regard to total phosphorus export loads.

7.4.2 Total Nitrogen

Nitrogen was not modelled at all satisfactorily in the initial version of the model. Figure 7.6 demonstrates that there was no correlation found between modelled and monitored results. Nitrogen, in the initial version of the model, was modelled as inorganic and organic fractions. Removal of the organic fraction was modelled using the same sedimentation model as for TSS and TP. This early version of the model focussed largely on TSS, TP and algal uptake processes, however the current version has been modified to include nitrogen transfer and transformation processes.

Nitrogen is now modelled as total nitrogen (TN), rather than organic and inorganic components. The major loss pathway is via adsorption and sedimentation, and a sediment remobilisation-water column diffusion link has been incorporated. The total nitrogen model predictions compared to monitored behaviour are shown in Figure 7.8. Figure 7.7 compares the correlation found between modelled and monitored results to the 1:1 relationship.

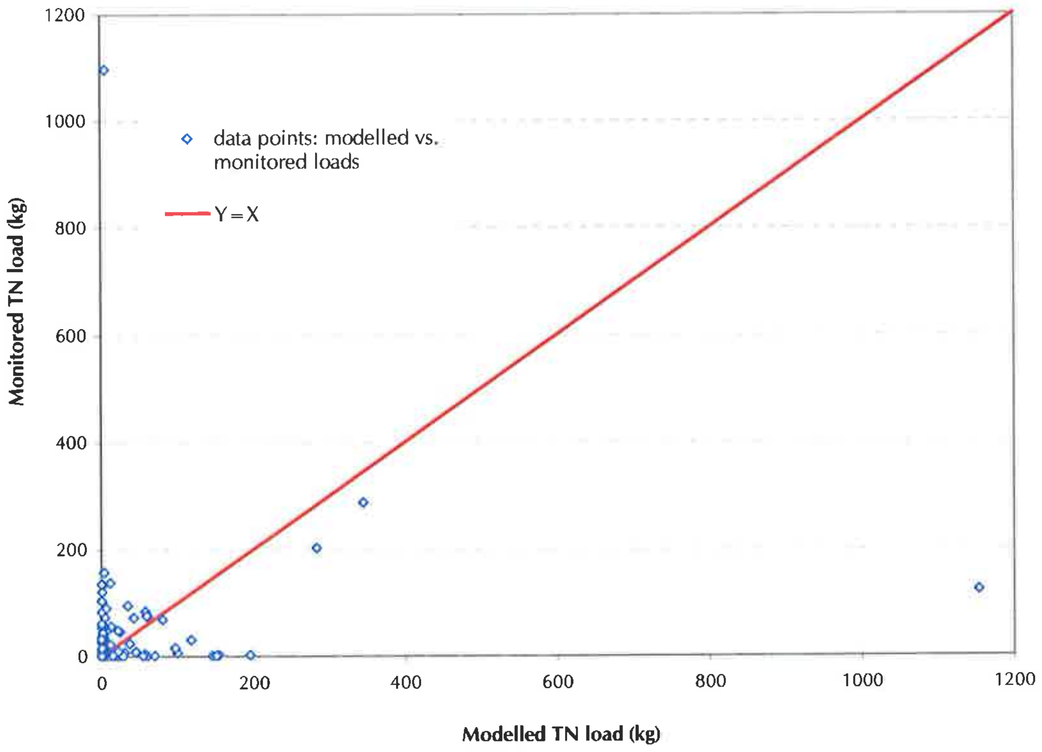


Figure 7.6 **OLD MODEL: Monitored vs. modelled daily total nitrogen loads exiting the pond compared to y=x line**

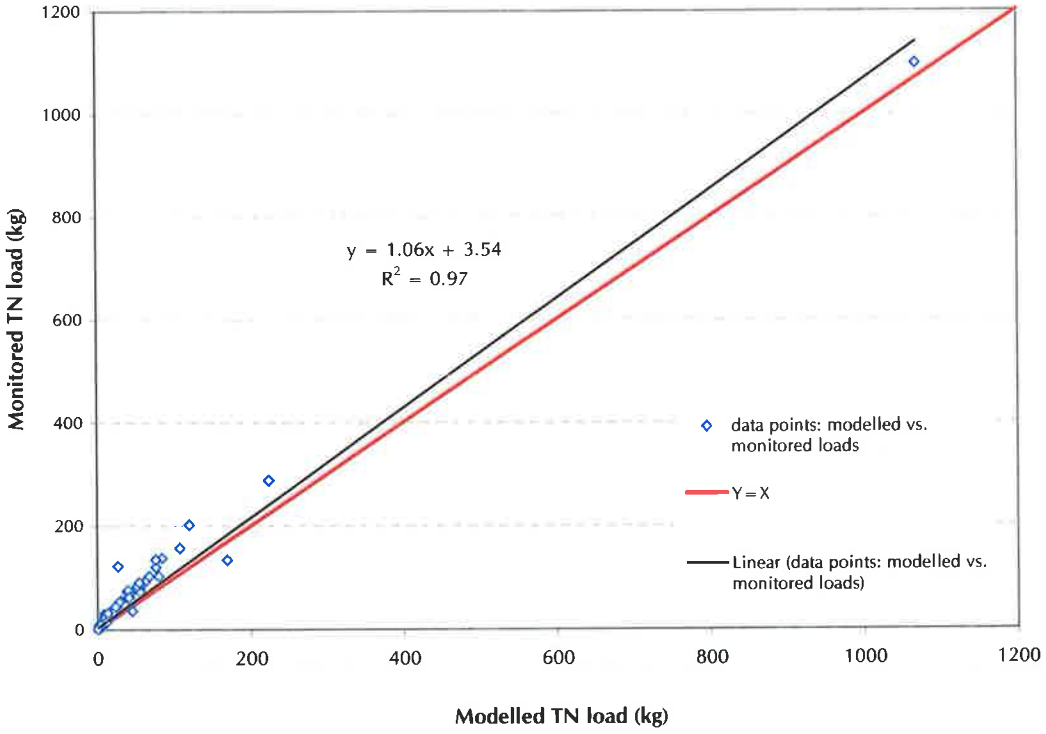


Figure 7.7 **NEW MODEL: Monitored vs. modelled daily total nitrogen loads exiting the pond compared to y=x line**

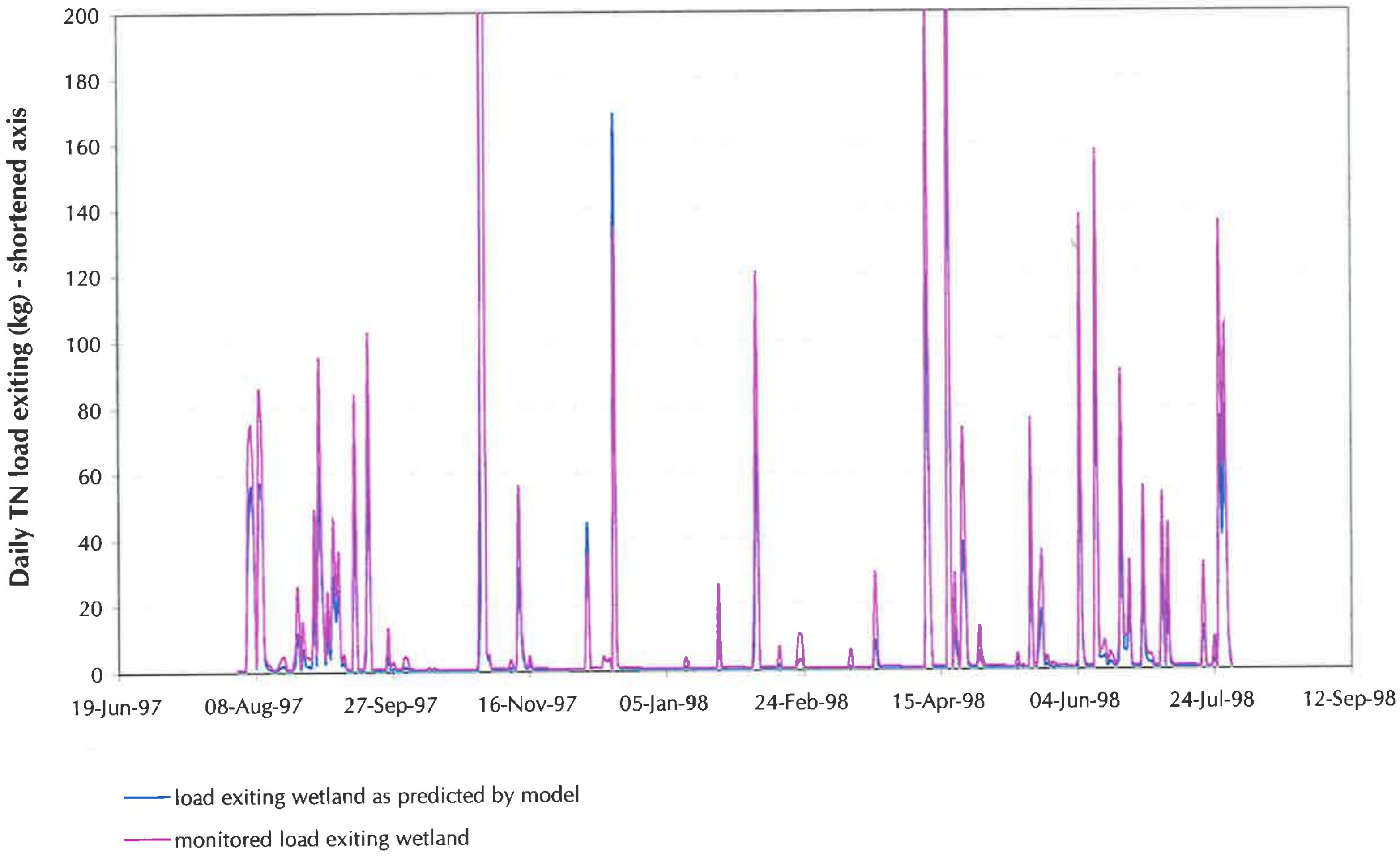


Figure 7.8 **NEW MODEL: Comparison of monitored (pink) and modelled (blue) daily total nitrogen loads exiting the pond**

Figure 7.7 indicates that a 1-to-1 relationship was almost attained for total nitrogen.

These results clearly illustrate that total nitrogen behaviour was modelled much better by the new version of the model with its improved nitrogen model components.

7.4.3 Total Suspended Solids

The total suspended solids component of the modelling proved to be the most difficult. The initial version of the model produced the results shown in Figure 7.9. As the graph shows, the model produced large spikes in concentration (and thus load) on a number of occasions during the simulation period. Some of the values of data points on the plot have been given to demonstrate the difference in magnitude between the modelled and monitored values.

A number of possible explanations for this behaviour were investigated; firstly, the particle size distribution of inflowing sediment. The model uses a single set of particle size information to calculate the proportion of each size fraction present in the inflow. Therefore when an event with a high inflow concentration enters the pond, the model calculates the relative fractions present using the "average" particle size distribution entered by the user. Without modification, the model is unable to simulate the fact that for high inflow concentrations there is probably a higher proportion of larger sediment present than under normal circumstances. It was initially thought that this could be the reason why outflow loads were being overestimated for events with high initial concentrations. By increasing the proportion of medium silt, and re-examining the results however, it was realised that this made little difference to the outcome.

Included in the new release of the model, as explained in Section 7.2.2, is an exponential algorithm to replace the former step wise CSTR washout algorithm. The method used in the original model was only accurate for daily inflows less than the pond volume. The new algorithm provides convergence for higher flows and, as Figure 7.10 demonstrates, this new method has reduced the incidence of spikes in the outflow loads.

The spikes in Figure 7.9 corresponded to days with large inflows relative to the Pond volume. The largest spike produced by the new model was on August 7th 1997, which had a large inflow TSS concentration rather than a large inflow volume. The same spike appears in both model simulations, however its origin could not determined.

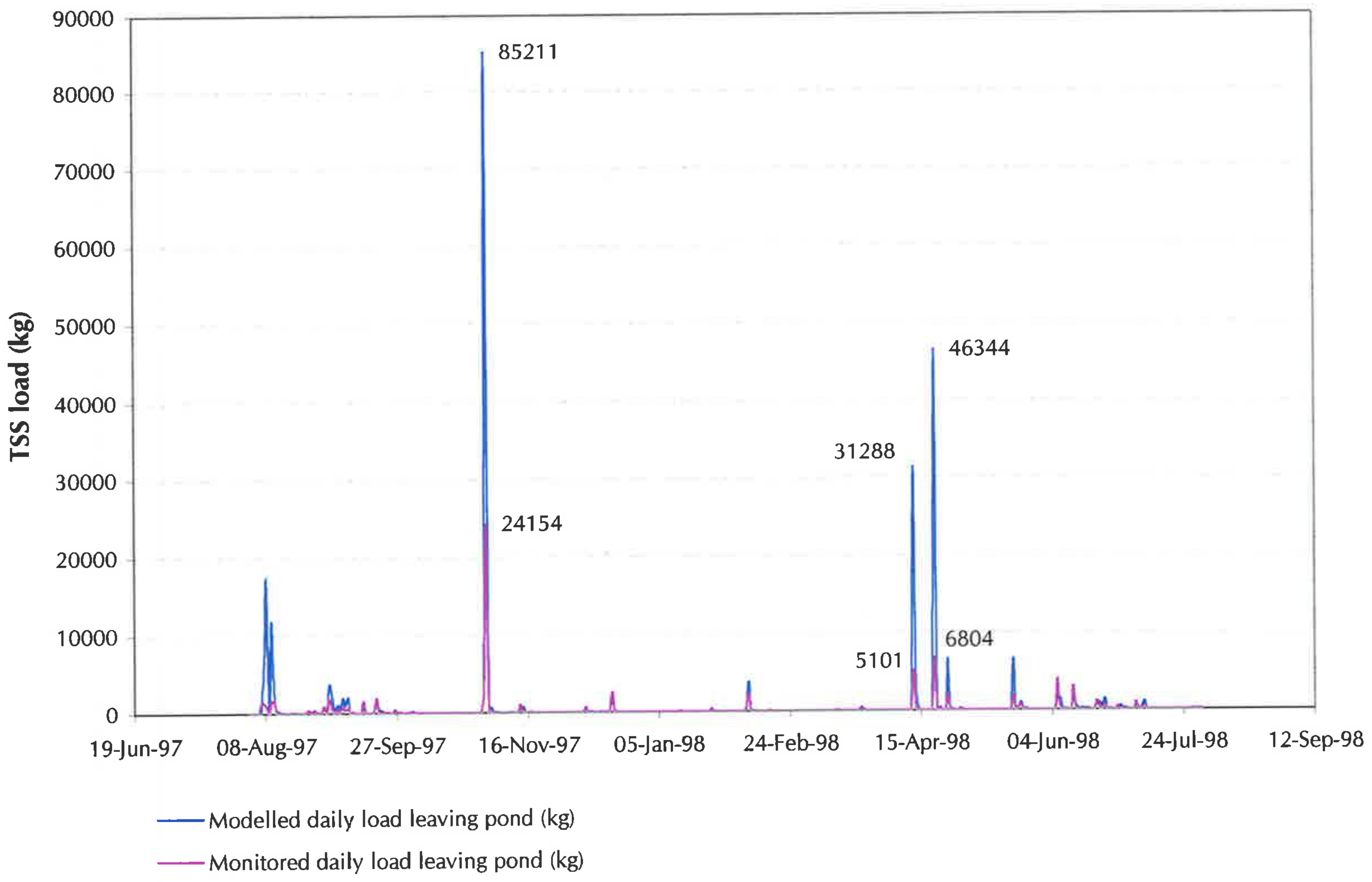


Figure 7.9 **OLD MODEL: Comparison of monitored (pink) and modelled (blue) daily total suspended solids loads exiting the pond**

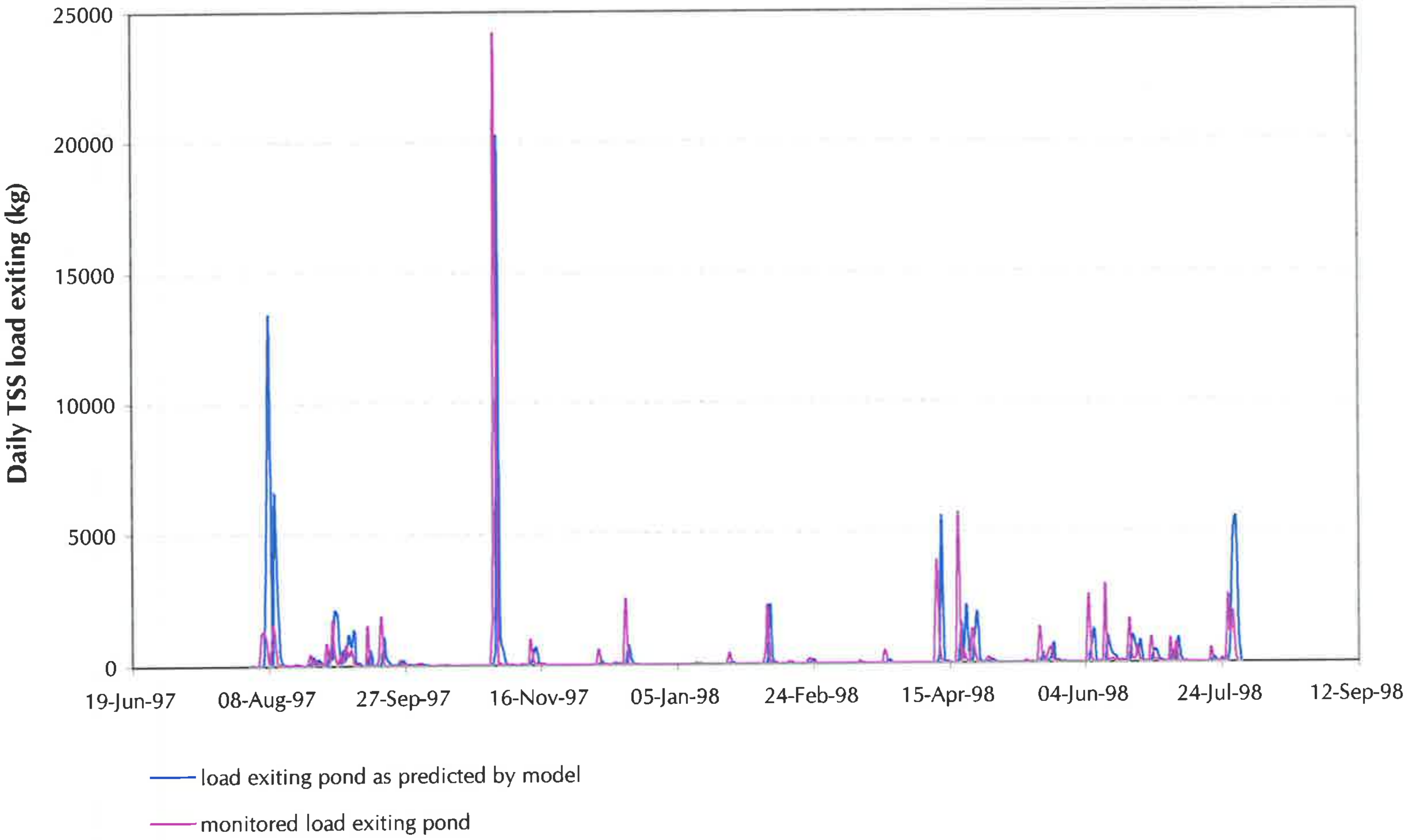


Figure 7.10 NEW MODEL: Comparison of monitored (pink) and modelled (blue) daily total suspended solids loads exiting the pond

Although Figure 7.10 appears to indicate a higher degree of model accuracy than was obtained for the old version of the model, Figure 7.11 shows that even the new model does not provide a significant correlation between modelled and monitored results.

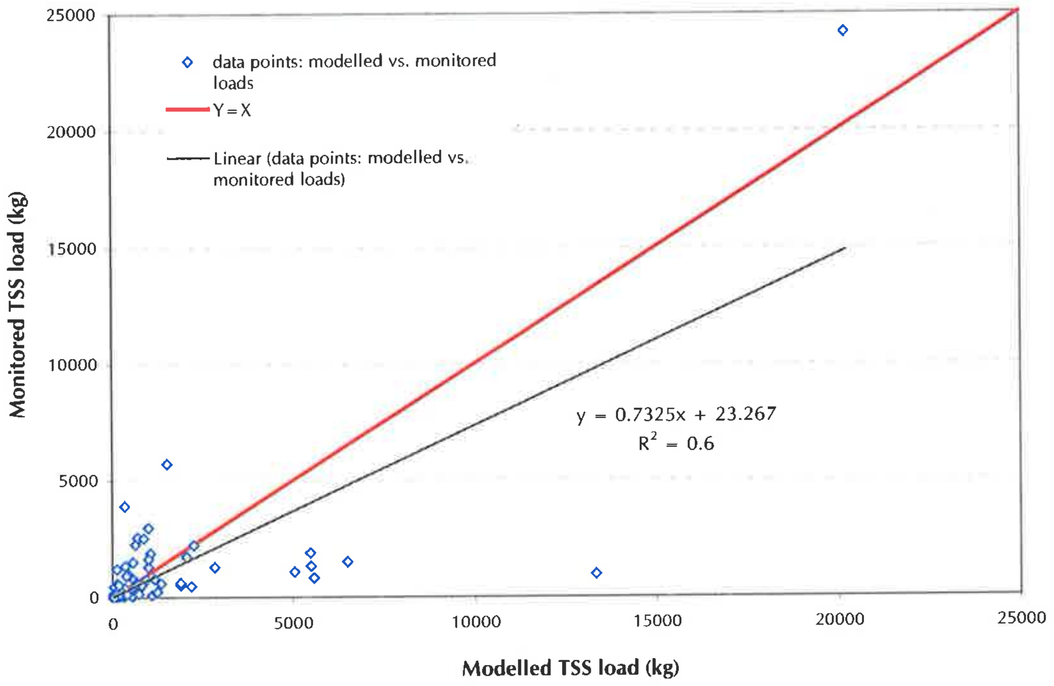


Figure 7.11 **NEW MODEL: Monitored vs. modelled daily total suspended solids loads exiting the pond compared to y=x line**

There is a significant amount of scatter in the data points when modelled and monitored results are plotted against each other.

The reason for the poor modelling of TSS could not be determined, and is surprising considering the same algorithm is used as for the particulate fractions of phosphorus and nitrogen. Despite the apparent scatter in the results, Section 7.6 will show that results over a longer period of time are more accurate.

7.5 Further Calibration

Attempts were made to further refine the predictions made by the model. This was done by altering some input parameters to see what effect they had on the results. The only parameter which was found to have a significant effect (and that was not a known fixed value such as volume) was the effective spillway length. The value used in the model was 32 metres, which is the length of the gabions at the South Road Connector site separating the main pond from the tailwater pond. In order to produce a 1:1 relationship between modelled and monitored total phosphorus results, an effective spillway length of 100 metres was required. It was not considered that this was a realistic number and so further refinements using this technique were abandoned. It is though that leakage through the gabions at the pond outlet may be affecting the retention period and therefore the model results.

The authors of the model suggest that other parameters defined in Worksheet A (Initialisation Worksheet) can be modified in the calibration process, however no local data were available for this calibration.

All aspects of the TSS model components were examined in detail in an attempt to improve its predictive capability, however no solution was found.

7.6 Model Predictions and Accuracy

The final worksheet in the model (Worksheet E) summarises the output results of the simulation and tabulates the daily inflow pollutant loads and daily export loads or interception. This enables the pond performance in terms of percent removal of pollutants to be determined.

The model was run over the 12-month period discussed in Chapter 6; 1st August 1997 to 31st July 1998. The estimated pond performance predicted by each version of the model is presented in Table 7.3. This can be compared to the estimate of pond performance made using results of the monitoring program, as shown in Table 7.3.

Table 7.3 Pond performance estimated by each of the model versions compared to monitored performance estimate

	Performance estimated by model (%)		Pond performance determined by monitoring (%)
Pollutant	**OLD**	**NEW**	
Suspended solids	42	81	75
Total phosphorus	46	79	34
Total nitrogen	na	71	16
BOD	34	46	-

na model results did not make sense

The table suggests that the new version of the model generally predicts a higher removal efficiency than was determined by the monitoring program. While TSS did not appear to be modelled accurately in the new model on a daily basis, Table 7.3 indicates a relatively close estimate of performance over a longer period of time.

A possible explanation for the discrepancy between the monitoring results and those of the modelling exercise is the immaturity of the pond system. Table 6.12 in the previous chapter presented a comparison between the performance of the Paddocks Wetland (in Salisbury, South Australia) and the Barker Inlet Wetland Pond 4. This table indicated that the Paddocks had a better removal efficiency, at which point it was suggested that this could be due to the more mature state of the pond system. It is possible that, in time, the Barker Inlet system will mature to provide treatment to the extent indicated by the model presented in this section.

7.7 Conclusions

This exercise has proven that this model provides a suitable method of predicting the performance of the Barker Inlet Wetland Pond 4, based on a set of simple parameters which can be easily estimated. While the predictive capability of the initial model proved to be more accurate for total phosphorus, the later developments provided a better estimation of total nitrogen and suspended solids behaviour. If this model were to be used in future studies on this particular site, consideration could be given to modifying the new model such that the old method of estimating TP reduction is incorporated.

Although the model did not appear to provide an adequate prediction of TSS behaviour on an event basis, the results presented in Table 7.3 suggest that the annual load of sediment removed by the pond was predicted to within six percent of that estimated by the monitoring program.

It is possible that further refinements to the model will be made in the future which may again improve the models ability to simulate Barker Inlet Wetland Pond 4 performance further.

In the mean time, investigation into some of the parameters that had to be assumed may be advantageous, for example characterisation of the particle size distribution at the pond inlet.

8

Summary and Conclusions

The study presented in this thesis has involved an investigation into the treatment efficiency of the primary pond in the Barker Inlet Wetland system in South Australia. Over an 18 month period, flow and water quality were monitored at both the inlet and outlet of the pond to gain an understanding of how well the pond is functioning in its early stages of development.

The results of the water quality monitoring were analysed on an event by event basis as well as over a longer period of time. The most complete 12 month section of the data set was chosen to make an estimate of the performance of the Pond over a one year period. This eliminated effects of seasonality on efficiency.

The results of the monitoring program are summarised in Table 8.1. The mean and median concentrations for each water quality parameter are given for the North Arm East and South Road Connector stations.

By comparing the values, it is evident that a majority of the pollutants have lower concentrations at the outlet of the pond. The exceptions are nitrogen species, and arsenic which is only present in trace amounts. No statistics could be given for mercury, as more often than not the concentration in the water samples were below the limits of detection. The erratic behaviour occurs as nitrogen is not a conservative parameter (it can be lost and gained from the system) and undergoes many complex transformations in storage.

Table 8.1 **Summary of North Arm East inflow and South Road Connector outflow mean and median concentrations for study period**

Parameter	North Arm East inflow		South Road Connector outflow	
	Mean (mg/L)	Median (mg/L)	Mean (mg/L)	Median (mg/L)
Ammonia (as N)	0.208	0.12	0.193	0.2
TKN (as N)	1.86	1.77	2.158	1.13
Nitrate + Nitrite (as N)	0.322	0.328	0.208	0.246
Total Nitrogen	2.17	1.77	2.34	1.41
Filt. Reactive Phosphorus	0.133	0.082	0.098	0.045
Particulate Phosphorus	0.307	0.266	0.141	0.126
Total Phosphorus	0.435	0.360	0.235	0.212
Aluminium	3.26	2.99	0.816	0.622
Arsenic (inorganic)	0.005	0.003	0.004	0.004
Cadmium	0.001	0.0008	0.0007	0.0003
Chromium	0.015	0.012	0.010	0.008
Copper	0.044	0.039	0.016	0.012
Iron	3.00	2.47	0.921	0.7
Lead	0.137	0.117	0.018	0.015
Manganese	0.088	0.072	0.072	0.069
Mercury	-	-	-	-
Nickel	0.006	0.005	0.004	0.003
Zinc	0.473	0.427	0.128	0.130
Total suspended solids	191.5	147.2	33.1	24.0

Mercury has not been included in the above table as concentrations were not significant.

The event analysis revealed a number of things previously noted by researchers in other locations;

- **The removal of many pollutants, in particular total suspended solids (TSS) is dependent on the initial concentration;**

  This phenomenon has been reported by Randall *et al.* (1982) and Ferrara and Witkowski (1983). Duncan (1997) stresses the importance of reporting initial concentrations when quoting removal efficiencies for this reason. Randall *et al.* (1982). explain that this phenomenon is attributable to the generally higher proportion of larger particles present in water with higher concentrations of TSS which settle out mode rapidly.

- **The removal efficiency is generally dependent on the event size, and efficiency decreases as event size increases;**
  This is simply due to the reduced residence time caused by larger inflow events. The effect of residence time on pond performance has been previously investigated by Tomlinson *et al.* (1993) and Lawrence (1986) who came to the same general conclusion.

- **Nutrient removal can be highly variable;**
  Ammonia and dissolved phosphorus in particular displayed erratic behaviour and on occasions were found to be present in higher concentrations at the pond outlet. This has been noted by other researchers such as Boström *et al.* (1982), Randall *et al.* (1982) and Holler (1989).

- **Metals have a high affinity for suspended solids and display very similar removal characteristics;**
  This is a well established fact noted many times before. Certain sediments provide sites for the attachment of metals, which results in metals being removed when the sediment drops out of suspension

An extreme event (approximately 1 in 50 year ARI rainfall event) was captured during the monitoring period, which was used to investigate the effect of extreme flows on pond performance. Although it is generally accepted that large flows should bypass pond systems to prevent resuspension of previously deposited sediments, it was found that in this case significant treatment was still provided by the pond. It is noted that this will not always be the case, particularly in smaller systems.

To examine the long term pond performance, loads entering and exiting the pond were determined for a 12 month period. Before this could be done, some gaps in the data set had to be filled in. A number of 'modelling' techniques were investigated, the most appropriate was found to be a mass export/runoff volume approach. This was able to predict pollutant loads for both the inflow and outflow stations based on event volume with reasonable accuracy.

The total loads for the period 1st August 1997 to 31st July 1998 are summarised in Table 8.2. An additional 10 percent was added to the estimated North Arm East loads to account for the Henschke Street input which could not be successfully monitored.

The table demonstrates that most water quality parameters exhibited at least some degree of removal. Metal and TSS removal were markedly higher than nutrients, although nitrate and particulate phosphorus were comparable.

Table 8.2 Annual loads entering and exiting Pond 4, including estimate of removal efficiency

Parameter	Total inflow to BIW Pond 4 (kg)	Total outflow from BIW Pond 4 (kg)	Reduction in pollutant load (%)
Ammonia	701	653	6.8
Nitrate + Nitrite	874	495	43.4
TKN	5117	4596	11.2
Organic N	4477	3943	11.9
Total N	5326	5091	15.9
Filt. Reactive P	404	403	0.2
Particulate P	750	472	48.8
Total P	1326	875	34.0
Aluminium	11262	3720	67.0
Arsenic	14	14	0
Cadmium	3	1	54.5
Chromium	45	20	55.6
Copper	146	50	65.8
Iron	10055	3324	66.9
Lead	476	68	85.7
Manganese	255	193	24.3
Mercury	0.34	0.26	13.3
Nickel	18	12	33.3
Zinc	1403	414	70.5
Total Suspended Solids	581 T	146 T	75.0

The estimation of removal efficiency determined for Barker Inlet Pond 4 was compared to the performance of other ponds. The most suitable comparison was determined to be the Paddocks in South Australia due to the similarities in climate and catchment characteristics. While the removal of TSS and metals in both ponds were comparable, the Paddocks achieves a higher removal of nutrient species. It is thought that this is a result of the greater maturity of the Paddocks System which was constructed in 1975. This is compared to the Barker Inlet System which was only completed a little over three years ago.

The secondary objective of this study was to model the pond behaviour with the aim of providing a tool for long term performance estimation without the need for further data collection. A model recently developed by the CRC for Freshwater Ecology was used for this exercise. The model was comprised of several sub-models which linked the various processes occurring in wetlands and ponds.

During the course of the study several modifications were made to the model to improve its predictive capability. Two versions, these being the initial release version and the current release at September 1999, were used to model Barker Inlet Pond 4. The same data were entering into both versions of the model to determine which provided the best results.

The initial model proved to be the best at predicting total phosphorus behaviour, while the new version was more appropriate for total nitrogen. Total suspended solids proved to be the most difficult to model, although the new model provided some improvement over the initial version.

The model generally predicted a higher removal efficiency than was estimated from the monitoring program. This may, once again, be a result of the immature state of the pond system.

The general conclusions of this study are that the Barker Inlet Wetlands are providing much needed protection to the Barker Inlet Aquatic Reserve through stormwater treatment. Through the investigation of the performance of the largest of the Southern ponds, it has been determined that a significant amount of stormwater treatment is being provided. Although other ponds may not be performing to the same extent, particularly Pond 2 (refer Figure 4.4) with its lack of vegetation, it should be kept in mind that Pond 4 receives the largest annual stormwater flow. The Northern Ephemeral Area and Marine Intertidal Wetland are also able to provide some final treatment for stormwater from the five southern ponds. It is thought that as the system matures further treatment may be provided through the increased biological activity and more stable ecosystem.

9

References

Allison, R., Chiew, F. and McMahon, T. (1997) "Stormwater Gross Pollutants." Industry Report, Cooperative Research Centre for Catchment Hydrology, Report 97/11, 17pp.

Anderssen, R.S., Dietrich, C.R. and Green, P.A. (1990) "Designing Artificial Lakes as Pollution Control Devices." *Mathematics and Computers in Simulation*, Vol. 32, Elsevier Science Publishers, B.V. North Holland, pp. 77-82.

ANZECC (1996) "Draft Guidelines for Urban Stormwater Management." National Water Quality Management Strategy. Australian and New Zealand Environment and Conservation Council.

ARC (1992) "Design Guideline Manual. Stormwater Treatment Devices." Prepared for Environment and Planning Division, Auckland Regional Council by Beca Carter Hollings and Ferner Ltd., 1st Edition.

Argue, J.R. and Scott, P (1996) "Residential Stormwater: Non-Gross Pollution." Progress Report, Urban Water Resources Centre, Adelaide.

Australian and New Zealand Environment and Conservation Council (1992) "Australian water Quality Guidelines for Fresh and Marine Waters." Australian and New Zealand Environment and Conservation Council, Canberra.

Bacher, G.J. and O'Brien, T.A. (1990) "The Sensitivity of Australian Freshwater Aquatic Organisms to Heavy Metals." SRS 88/018, Environment Protection Authority, Melbourne.

Ball, J.E. and Ferguson, D. (1994) "Enhancements to the Sotrmwater Management Model for Simulation of Urban Runoff." In *Water Downunder '94*, Volume 3, Joint IAH IEAust, Adelaide, pp. 293-298.

Ball J.E. and Abustan, I. (1995) "An Investigation of the Particle Size Distribution During Storm Events on an Urban Catchment." In Proceedings of the 2nd International Symposium on Urban Stormwater Management, Melbourne, Institution of Engineers Australia, pp. 531-535.

Benes, P. and Steinnes, E (1975) "Migration Forms of Trace Elements in Natural Fresh Waters and the Effect of Water Storage." *Water Research*, 9, pp. 741-749.

Berner, E.K. and Berner, R.A. (1987) "The Global Water Cycle." Prentice Hall, New Jersey.

Biesinger, K.E. and Christen, G.M (1972) "Effects of Various Metals on Survival, Growth, Reproduction and Metabolism of *Daphnia magna*." *Journal of the Fisheries Resources Board of Canada*, 29, pp. 1690-1700.

Bos, M.G. (1989) "Discharge Measurement Structures." 3rd Revised Edition, ILRI Publication 20, International Institute for Land Reclamation and Improvement, Wageningen, The Netherlands.

Boström,B., Jansson, M. and Forsberg, C. (1982) "Phosphorus Release from Lake Sediments." *Ergebnisse der Limnologie - Advances in Limnology*, Stuttgart, 18, 5-59.

Bowmer, K.H. (1993) "Wetlands in Urban Water Systems – Feasible or Fashionable?" CSIRO Division of Water Resources, Technical Memorandum 93/12.

Breen, P. (1992) "Natural and Artificial Wetlands for Treating Sewage and Urban Runoff Wastewaters." In *Catchments of Green*, Proceedings National Conference on Vegetation and Water Management, Greening Australia, Adelaide, pp. 229-234

Brown, R.T., Field, J.J, Zanoli, M.J. and Crite, R.W. (1994) "Modeling Pollutant Fate and Transport in Constructed Wetlands." In *Critical Issues in Water and Wastewater Treatment*, National Conference in Environmental Engineering, American Society of Civil Engineers, New York, pp. 523-539.

Canadian Council of Resource and Environment Ministers (1987) "Canadian Water Quality Guidelines." Health and Welfare Canada, Ottawa.

CCREM (1991) "Canadian Water Quality Guidelines." Canadian Council of Resource and Environment Ministers, Inland Water Directorate, Environment Canada, Ottawa.

Chen, C.W. and Shubinski, R.P (1971) "Computer Simulation of Urban Storm Water Runoff." *Journal of the Hydraulics Division*, Proceedings of the American Society of Civil Engineers, Vol. 97, No. 2, pp. 289-301.

Chiew, F.H.S. (1995)

Christensen, N., Mitsch, W.J. and Jørgensen, S.E. (1994) "A First Generation Ecosystem Model of Des Plaines River Experimental Wetlands." *Ecological Engineering*, 3, Elsevier Science Publishers, Amsterdam, pp. 495-521.

Connell, D.W. (1993) "Water Pollution: Causes and Effects in Australia and New Zealand." Third Edition, University of Queensland Press, 203 pp.

Cordery, I., Cloke, P.S. and Sands, J.R. (1997) "Integrated Water Monitoring: The Priority for Water Quality Information." *Australian Journal of Water Resources*, Technical Paper, Vol. 2, No. 1, Institution of Engineers Australia, pp. 29-33.

Cullen, P. (1989) "Social Choice, Risk and Determinism in Water Quality Management." *Hydrobiologia*, Kluwer Academic Publishers, Dordrecht, The Netherlands, pp. 1-5.

Daniell, T.M. and McCarty, D.J (1994) "Sedimentation in Artificial Urban Wetlands at Happy Valley, South Australia." In *Water Downunder '94*, Vol. 3, Joint IAH IEAust, Adelaide, pp. 341-346.

Daniell, T., Murphy, S. and French, R. (1999) "Barker Inlet Monitoring Final Report, Water Quality and Flow Monitoring." Report to former MFP DC.

Dannecker, W., Au, M. and Stechmann, H. (1990) "Substance Load in Rainwater Runoff from Different Streets in Hamburg." *Science of the Total Environment*, 93, Elsevier Publishing Company, Amsterdam, pp. 385.

Department of Territories (1986) "Pollutant Removal in an Urban Wetland." National Capital Development Commission.

Dillon, P. and Pavelic, P. (1995) "Draft Guidelines on the Quality of Stormwater and Treated Wastewater for Injection into Aquifers for Storage and Reuse." CSIRO Centre for Groundwater Studies Report No. 63.

Dørge, J. (1994) "Modelling Nitrogen Transformations in Freshwater Wetlands. Estimating Nitrogen Retention and Removal in Natural Wetlands in Relation to their Hydrology and Nutrient Loadings." *Ecological Modelling*, 75/76, Elsevier Science Publishers, Amsterdam, pp. 409-420.

Dorney, J.R. (1986) "Leachable and Total Phosphorus in Urban Street Tree Leaves." *Water Air and Soil Pollution*, 28, pp.439.

Driscoll, E.D. (1986) "Detention and Retention Controls for Urban Runoff." In B. Urbonas and L. Roesner (Eds.), *Urban Runoff Quality – Impact and Quality Enhancement Technology, Proceedings of an Engineering Foundation Conference*, New Hampshire, American Society of Civil Engineers, New York, pp. 381-393.

Driscoll, C.T., Baker, J.P., Bisogni, J.J. and Schofield, C.L. (1980) "Effect of Aluminium Speciation in Dilute Acidified Waters." *Nature* 284, Macmillan, London, pp. 161-184.

Duncan, H. (1995a) "A Review of Urban Stormwater Quality Processes." Cooperative Research Centre for Catchment Hydrology, Melbourne, Report No. 95/9.

Duncan, H.P. (1995b) "A Bibliography of Urban Stormwater Quality." Cooperative Research Centre for Catchment Hydrology, Report No. 95/8.

Duncan, H.P. (1997) "Urban Stormwater Treatment by Storage: A Statistical Overview." Cooperative Research Centre for Catchment Hydrology, Melbourne, Report No. 97/1.

Duncan, H.P. (1998) "Urban Stormwater Quality Improvement in Storage." In *HydraStorm '98*, Proceedings of the 3rd International Symposium on Stormwater Management and the 6th International Conference on Hydraulics in Civil Engineering, Adelaide, Institution of Engineers Australia, pp. 203-208.

Fair, G.M. and Geyer, J.C. (1954) "Water Supply and Waste-Water Disposal." Vol. 2, John Wiley and Sons, New York, 973 pp.

Ferrara, R.A. and Witkowski, P. (1983) "Stormwater Quality in Detention Basins." *Journal of Environmental Engineering*, Vol. 109, No. 2., American Society of Civil Engineers, pp. 428-447.

Finlayson, C.M. (1983) "Use of Aquatic Plants to Treat Wastewater in Irrigation Areas of Australia." Proceedings of AWWA 10th Federal Convention, Sydney, Australian Water and Wastewater Association, pp. 25.2-25.9.

Finlayson, C.M. and Mitchell, D.S. (1982) "Treatment of Rural Wastewater in Australia with Aquatic Plants." *Der Tropenlandwirt*, Oct, pp. 155-165.

Finlayson, C.M. and Chick, A.J. (1983) "Testing the Potential of Aquatic Plants to Treat Abattoir Effluent." *Water Research*, Vol. 17, No. 4, Pergamon Press, New York, pp. 415-422.

Fisher, P. and Stricker, J. (1992) "Constructed Wetland Systems in Water Pollution Control." In *Catchments of Green: A National Conference on Vegetation and Water Management*, Adelaide, Australia, Vol. 2, Greening Australia, pp. 91-102.

French, R.K. (1999) "Modelling Urban Runoff Volume and Pollutant Concentration of the Barker Inlet Wetland Catchment." Unpublished, The University of Adelaide.

Gamble, S. (1997) "Monitoring of the Nutrient Inflows and Outflows of a Constructed Urban Wetland." Thesis submitted for the degree of M.Eng.Sci., The University of Adelaide, Department of Civil and Environmental Engineering.

Gippel, C.J. (1995) "Potential of Turbidity Monitoring for Measuring the Transport of Suspended Solids in Streams." *Hydrological Processes*, 9, John Wiley & Sons Ltd., pp. 83-97.

Golterman, H.L. (1975) "Physical Limnology: An Approach to the Physiology of Lake Ecosystems." Elsevier Scientific Publishing Company, New York, 489 pp.

Gorrie, J.R., Jenkins, C.H. and Bidwell, J.R. (1998) "Toxicity Assessment of Sediments in a Constructed Wetland Designed to Receive Urban Stormwater." In HydraStorm '98, Proceedings of the 3rd International Symposium on Stormwater Management and the 6th International Conference on Hydraulics in Civil Engineering, Adelaide, Institution of Engineers Australia, pp. 445-450.

Graham, R. (1990) "Urban Stormwater Quality Improvement in a Detention Pond and Wetland System – The Wildlife Reserve, LaTrobe University." In S. Diez (Ed), *Wetlands: Their Ecology, Function, Restoration and Management, Wildlife Reserves*, LaTrobe University, Bundoora, Victoria, pp. 59-64.

Grizzard, T.J., Randall, C.W., Weand, B.L. and Ellis, K.L. (1986) "Effectiveness of Extended Detention Ponds." In Urban Runoff Quality – Impact and Quality Enhancement Technology, Proceedings of an Engineering Foundation Conference, New Hampshire, American Society of Civil Engineers.

Gupta, M.K., Agnew, R.W. and Kobriger, N.P. (1981) "Constituents of Highway Runoff, I: State of the Art Report." FHWA/RD-81/042. NTIS PB81-241895, US Federal Highway Administration.

Hart, B.T. (1982) "Australian Water Criteria for Heavy Metals." AWRC Technical Paper No. 77, Australian Government Publishing Service, Canberra.

Hensel, P.F., Day, J.W. Jr., Pont, D. and Day, J.N. (1998) "Short Term Sedimentation Dynamics in the Rhône River Delta, France: The Importance of Riverine Pulsing." *Estuaries*, Vol. 21, No. 1, Estuarine Research Federation, Columbia, pp. 52-65.

Holcombe, G.W. and Andrew, R.W. (1978) "The Acute Toxicity of Zinc to Rainbow and Brook Trout. Comparisons in Hard and Soft Water." EPA-600/3-78-094, Environmental Protection Agency, Washington DC.

Holler, J.D. (1989) "Stormwater Detention Basin Nutrient Removal Efficiency." *Journal of Water Resources Planning and Management*, Vol. 115, No. 1, American Society of Civil Engineers, pp.52-63.

Hynes, H.B.N. (1974) "The Biology of Polluted Waters." Liverpool University Press, 202 pp.

Jacobi, D. and Murphy, S.E. (1996) "Evaluation of an Urban Stormwater Wetland." Unpublished, Department of Civil and Environmental Engineering, The University of Adelaide, 167 pp.

James, A. and Elliot, D.J. (1991) "The Modelling of Stormwater Pollution." In *Water Pollution: Modelling, Measuring and Prediction*, Computational Mechanics Publications, Elsevier Applied Science, Southhampton, pp. 155-165.

Jenkins, C.H., Gorrie, J.R., Bidwell, D.E., Mulcahy, D.E. and Manning, P. (1998) " Integrated Chemical and Toxicological Assessment of Heavy Metals in A Constructed Stormwater Wetland." In HydraStorm '98, Proceedings of the 3rd International Symposium on Stormwater Management and the 6th International Conference on Hydraulics in Civil Engineering, Adelaide, Institution of Engineers Australia, pp. 457-462.

Jewel, T.K. and Adrian, D.D. (1981) "Development of Improved Stormwater Quality Models." *Journal of the Environmental Engineering Division*, Proceedings of the American Society of Civil Engineers, Vol. 107, No. 5, pp. 957-974.

Jones, G. (1994) "Bloom-Forming Blue-Green Algae (Cyanobacteria)." In G.R. Sainty and S.W.L. Jacobs, *Water Plants in Australia: A Field Guide*, Third Edition, CSIRO Division of Water Resources, pp. 267-284.

Jørgensen, S.E. (1988) "Fundamentals of Ecological Modelling." Elsevier, Amsterdam, 391 pp.

Jørgensen, S.E. (1995) "State of the Art of Ecological Modelling in Limnology." *Ecological Modelling*, 78, Elsevier Science Publishers, Amsterdam, pp. 101-115.

Kadlec, R.H. and Hammer, D.E. (1988) "Modeling Nutrient Behaviour in Wetlands." *Ecological Modelling*, 40, Elsevier Science Publishers, Amsterdam, pp. 37-66.

Kadlec, R.H and Hey, D.L. (1994) "Constructed Wetlands for River Water Quality Improvement." Water Science and Technology, Vol. 29, No. 4, Elsevier Science, pp. 159-168.

Kadlec, R.H. and Knight, R.L. (1996) "Treatment Wetlands." Lewis Publishers, New York, 893 pp.

Keefer, T.N., Simons, R.K. and McQuivey, R.S. (1979) "Dissolved Oxygen Impact from Urban Storm Runoff." EPA-600/2-79-156.NTIS PB80-138266, US Environmental Protection Agency.

Kemp, W.M., Wetzel, R.L., Bonython, W.R., D'Elia, C.F. and Stevenson, J.C. (1982) "Nitrogen Cycling and Estuarine Interfaces: Some Current Concepts and Research Directions." In V.S. Kennedy (Ed.), *Estuarine Comparisons*, Academic Press, New York, pp. 209-230.

Kemp, W.M., Sampou, P., Caffrey, J., Mayer, M., Henrikson, K. and Bonython, W.R. (1990) "Ammonium Recycling Versus Denitrification in Chesapeake Bay Sediments." *Limnology and Oceanography*, 35, pp. 1545-1563.

Koehn, J.D. and O'Connor, W.G. (1990) "Biological Information for Management of Native Freshwater Fish in Victoria." Department of Conservation and Environment, Melbourne.

Kramer, J.R., Herbes, S.E. and Allen, H.E. (1972) "Phosphorus: Analysis of Water, Biomass, and Sediment." In H.E. Allen and J.R. Kramer (Eds.), *Nutrients in Natural Waters*, Environmental Science and Technology Series, John Wiley & Sons Inc., New York, pp.51-100.

Laidler, G. (1991) "Environmental Chemistry – An Australian Perspective." Second Edition, Longman Cheshire, Melbourne, 233 pp.

Lawrence, I. (1986) "Source and Fate of Urban Runoff Constituents and Their Management." Water Research Federation of Australia, Illawarra Regional Committee 12th Symposium - *Stormwater Quality in Urban Areas*, The University of Wollongong, Australia.

Lawrence, I. and Baldwin, S. (1997) "Development of an Urban Pond Water Quality Model." *Australian Journal of Water Resources*, Vol. 2. No. 1, Institution of Engineers Australia, pp. 15-18.

Lawrence, I. and Breen, P. (1998) "Guidelines for Designing Pollution Control Ponds." *Water; Journal of the Australian Water and Wastewater Association*, Vol. 25, No.3, AWWA Inc., pp. 25-26.

Lawrence, I. and Breen, P. (1998 (b)) "Design Guidelines: Stormwater Pollution Control Ponds and Wetlands." Edition1, Cooperative Research Centre for Freshwater Ecology.

Listowski, A., Cartwright, T., Phillips, B.C. and Lawrence, A.I. (1997) "A Stormwater Management Strategy for the Urban/Commercial Renewal of Homebush Bay, Australia." Proceedings 24th Hydrology and Water Resources Conference, Auckland, pp. 234-239

Livingston, E.H. (1986) "Stormwater Regulatory Program in Florida." In *Urban Runoff Quality – Impact and Quality Enhancement Technology, Proceedings of an Engineering Foundation Conference*, Henniker, NH, June 23-27 1986, pp. 249-256.

Livingston, E.H. (1991) "Use of Wetlands for Urban Stormwater Management." In D.A. Hammer (Ed.) *Constructed Wetlands for Wastewater Treatment, Municipal, Industrial and Agricultural*, Lewis Publishers, pp. 253-262.

Lloyd, S.D., Wong, T.H.F., Liebig, T. and Becker, M. (1998) "Sediment Characteristics in Stormwater Pollution Control Ponds." In *HydraStorm '98*, Proceedings of the 3rd International Symposium on Stormwater Management and the 6th International Conference on Hydraulics in Civil Engineering, Adelaide, Institution of Engineers Australia, pp. 209-214.

Lung, W. and Light, R.W. (1996) "Modelling Copper Removal in Wetland Ecosystems." *Ecological Modelling*, 93, Elsevier Science, pp. 89-100.

Makepeace, D.K., Smith, D.W. and Stanley, S.J. (1995) "Urban Stormwater Quality: Summary of Contaminant Data." *Critical Review in Environmental Science and Technology*, Vol. 25, No. 2, CRC Press, pp. 93-139.

Mann, R. and Hammerschmid, K. (1989) "Physical and Chemical Characteristics of Urban Runoff from Two Catchments in Hawkesbury / Nepean Basin." Australian Water and Wastewater Association, 13th Federal Convention, Canberra, 6-10th March 1989.

Manning, P. and Stevens, R. (1991) "Onkaparinga Estuary Stormwater Wetlands. A Discussion of Guidelines." In M. Makestas (Ed.) *Seminar on Managing Impacts on the Water Environment in the 1990's*, Australian Water and Wastewater Association (SA Branch), Adelaide.

Martin, E.H. (1988) "Effectiveness of an Urban Stormwater Detention Pond-Wetlands System." *Journal of Environmental Engineering*, Vol. 114, No. 4, American Society of Civil Engineers, pp. 283-291.

McAlister, T., Syme, B., Bycroft, B. and Mack, P. (1995) "The Application of a Common Australian Stormwater Quality and Quantity Model in a Sub-Tropical Environment." In Proceedings of the Second International Symposium on Urban Stormwater Management, Institution of Engineers Australia, Melbourne, Australia.

McMahon, T.A. and Finlayson, B.L. (1991) "Australian Surface and Groundwater Hydrology. Regional Characteristics and Implications." In *Water Allocation for the Environment*, Proceedings of an International Seminar and Workshop, University of New England, Armidale, Australia, pp. 21-40.

MFP Australia (1995) "Revised Initial Patawalonga Catchment Management Plan." Patawalonga Catchment Water Management Board, Adelaide, 59 pp.

MFP Development Corporation (1996) "Annual Report 1995/1996." Adelaide.

Millar, R.G. (1996) "Spurious Correlation and Pollutant Export Relations." *Water; Journal of the Australian Water and Wastewater Association*, Vol. XX, No.X, AWWA Inc., pp. 33-35.

Miller, G.T. Jr. and Armstrong, P. (1982) "Living in the Environment: Concepts, Problems and Alternatives." International Edition, Wadsworth International Group, Belmont California.

Milne, S.J. and Clark, R.D.S. (1991) "A Feasibility Study for the Establishment of Wetlands on Urbanising Catchments Upstream of the Happy Valley Reservoir." Engineering and Water Supply Department, Report 91/18, Adelaide.

Mitchell, C. (1995) "Essential Wastewater Microbiology." In IWES '95 Course Notes – *Artificial Wetlands: Principals, Design and Management*, University of Queensland.

Mitsch, W.J. (1983) "Ecological Models for Management of Freshwater Wetlands." In S.E. Jørgensen and W.J. Mitsch (Eds.), *Application of Ecological Modelling in Environmental Management, Part B, Developments in Environmental Modelling*, 4B, Elsevier, Amsterdam, pp. 283-310.

Mitsch, W.J., Wu, X. and Wang, N. (1993) "Modelling the Des Plaines Experiemental Wetlands: An Integrative Approach to Data Management and Ecosystem Prediction." Report to USEPA Region V, Contract 91044, Project 769641/725753, Wetland Research Inc., Chicago, Illinois.

Moreno-Grau, S., García-Sánchez, A., Moreno Clavel, J., Serrano-Aniorte, J. and Moreno-Grau, M.D. (1996) "A Mathematical Model for Waste Water Stabilization Ponds with Macrophytes and Microphytes." *Ecological Modelling*, 91, Elsevier Science Publishers, Amsterdam, pp. 77-103.

Morrison, G.M.P., Revitt, D.M., Ellis, J.B., Svensson, G. and Balmer, P. (1984) "Variations of Dissolved and Suspended Solid Heavy Metals Through an Urban Hydrograph." *Environ. Technol. Lett.*, 7, pp. 313.

Mount, D.I. (1966) "The Effect of Total Hardness and pH on Acute Toxicity of Zinc to Fish." *Air and Water Pollution*, 10, pp. 49-56.

Mulhern, P.F. and Steele, T.D. (1988) "Water Quality Ponds – Are They the Answer?" In L.A. Roesner, B. Urbonas, and M.B. Sonnen (Eds.) *Design of Urban Runoff Quality Controls, Proceedings of an Engineering Foundation Conference on Current Practice and Design Criteria for Urban Quality Control*, American Society of Civil Engineers, New York, pp. 203-213.

Murphy, S.E., Daniell, T.M. and Walker, D.J. (1998) "The Dynamic Effectiveness of a Primary Pond in a Stormwater Wetland." In HydraStorm '98, Proceedings of the 3rd International Symposium on Stormwater Management and the 6th International Conference on Hydraulics in Civil Engineering, Adelaide, Institution of Engineers Australia, pp. 445-450.

Neeley, R.K. and Baker, J.L. (1989) "Nitrogen and Phosphorus Dynamics and the Fate of Agricultural Runoff." In A.G. van der Valk (Ed.), *Northern Prairie Wetlands*, Iowa State University Press, pp. 92-131.

New South Wales Department of Housing (1993) "Soil & Water Management for Urban Development." Second Edition, New South Wales Government, pp. 63-96.

Nixon, S.W. (1981) "Remineralisation and Nutrient Cycling in Coastal Marine Ecosystems." In B.J. Nielson and L.E. Cronin (Eds.), *Estuaries and Nutrients*, The Human Press, Clifton, New Jersey, pp. 111-138.

Novotny, V, Sung, H.M., Bannerman, R and Baum, K. (1995) "Estimating Non-point Pollution from Small Urban Watersheds." *Journal of the Water Pollution Control Federation*, Vol. 57, No. 4, pp. 339-348.

NSWEPA (1997) "Managing Urban Stormwater – Treatment Techniques." New South Wales Environment Protection Authority, Chatswood.

O'Loughlin, G.G. (Ed.) (1992) "Proceedings of the International Symposium on Urban Stormwater Management." Institution of Engineers Australia, Sydney, Australia.

O'Shea, M.L. and Field, R. (1991) "Detention and Disinfection of Pathogens in Storm-generated Flows." *Canadian Journal of Microbiology*, Vol. 38, No. 4, pp. 267.

Passfield, G.R.A. and Phillips, S.C. (1996) "Suspended Sediment Charcteristics, Dry Creek Catchment, South Australia." Unpublished, Department of Civil and Environmental Engineering, The University of Adelaide, 103 pp.

pH Environment (1995)

Phillips, B., Lawrence, A.I. and Bogiatzis, T. (1992) "An Integrated Water Quality and Streamflow Model Suite." In Proceedings of the International Symposium on Urban Stormwater Management, Institution of Engineers Australia, Sydney, Australia.

Pope, L.M. and Hess, L.G. (1988) "Load-Detention Efficiencies in a Dry-Pond Basin." In L.A. Roesner, B. Urbonas, and M.B. Sonnen (Eds.) *Design of Urban Runoff Quality Controls, Proceedings of an Engineering Foundation Conference on Current Practice and Design Criteria for Urban Quality Control*, American Society of Civil Engineers, New York, pp. 258-267.

Prescott, K.L. and Tsanis, J.K. (1997) "Mass Balance Modelling and Wetland Restoration." *Ecological Engineering*, 9, Elsevier Science, Amsterdam, pp. 1-18.

Raisin, G.W., Mitchell, D.S. and Croome, R.L. (1997) "The Effectiveness of a Small Constructed Wetland in Ameliorating Diffuse Nutrient Loadings from an Australian Rural Catchment." *Ecological Engineering*, 9, Elsevier Science, pp. 19-35.

Randall, C.W. (1982) "Stormwater Detention Ponds for Water Quality Control." In W. DeGroot (Ed.) *Stormwater Detention Facilities – Planning Design Operation and Maintenance*, Henniker, New Hampshire, American Society of Civil Engineers, pp. 200-204.

Randall, C.W., Ellis, K., Grizzard, T.J. and Knocke, W.R. (1982) "Urban Runoff Pollutant Removal by Sedimentation." In W. DeGroot (Ed.) *Stormwater Detention Facilities – Planning Design Operation and Maintenance*, Henniker, New Hampshire, American Society of Civil Engineers, pp. 205-219.

Reddy, K.R. and D'Angelo, E.M. (1994) "Soil Processes Regulating Water Quality in Wetlands." In W.J. Mitsch (Ed.) *Global Wetlands: Old World and New*, Elsevier Science B.V., The Netherlands, pp. 309-324.

Reddy, K.R., Patrick, W.H, and Phillips, R.E. (1980) "Evaluation of Selected Processes Controlling Nitrogen Loss in a Flooded Soil." *Soil Science Society of America Journal*, 40, Soil Science Society of America, Madison, pp. 1241-1246.

Reddy, K.R. and Patrick, W.H. (1984) "Nitrigen Transformations and Loss in Flooded Soils and Sediments." *CRC Critical Reviews in Environmental Control*, 13, pp. 273-309.

Reddy, K.R. and DeBusk, W.F. (1987) "Nutrient Storage Capabilities of Aquatic and Wetland Plants." In K.R. Reddy and W.H. Patrick (Eds.) *Aquatic Plants for Water Treatment and Resource Recovery*, Magnolia Publishing Inc., Orlando, Florida, pp. 337-357.

Reed, J.L (1921) "On the Correlations Between Any Two Functions and Its Application to the General Case of Spurious Correlation." *J. Wash Acad. Sci.*, 11, pp. 449-455.

Reeder, S., Demayo, A. and Taylor, M. (1979) "Guidelines for Surface Water Quality." Volume 1, Inorganic Chemical Substances. Cadmium, Environment Canada, Ottawa.

Reinelt, L.E. and Horner, R.R. (1995) "Pollutant Removal from Stormwater Runoff by Palustrine Wetlands Based on Comprehensive Budgets." *Ecological Engineering*, 4, Elsevier Science, pp. 77-97.

Richardson, C.J. and Marshall, P.E. (1986) "Processes Controlling Movement, Storage, and Export of Phosphorus in a Fen Peatland." *Ecol. Monogr.*, Vol. 56, No. 4, pp. 279-302.

Rowe, P.N. (1963) "Correlation of Engineering Data." *The Chemical Engineer*, March CE 69-74.

Rushton, B., Miller, C. and Hull, C. (1995) "Residence Time as a Pollutant Removal Mechanism in Stormwater Detention Ponds." Proceedings of the 4th Biennial Stormwater Research Conference, Clearwater, Florida, Southwest Florida Water Management District, Brooksville, Florida, pp. 210-221.

Ryan, P.A. (1991) "Environmental Effects of Sediment in New Zealand Streams: A Review." *New Zealand Journal of Marine and Freshwater Resources*, Vol. 25, pp. 207-221.

Sainty, G., Jacobs, S. and Adcock, P. (1994) "Waterplants to Purify Water." In G.R. Sainty and S.W.L. Jacobs, *Waterplants in Australia: A Field Guide*, CSIRO Division of Water Resources, pp. 310-316.

Schofield, C.L. and Trojnar, J.R. (1980) "Aluminium Toxicity to Brook Trout (*Salvelinus frontinalis*) in Acidified Waters." International Conference on Environmental Toxicity, May 21-22 1979, Rochester.

Schueler, T.R. (1987) "Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban BMPs." Metropolitan Washington Council of Governments, Washington DC.

Scott, A. and Davis, R. (1996) "Review of Urban Stormwater Research in Australia." Technical Memorandum 96.9, CSIRO Division of Water Resources, Canberra.

Shatwell, T. and Cordery, I. (1998) " A Phosphorus and Sediment Balance of an Urban Pond." In *HydraStorm '98*, Proceedings of the 3rd International Symposium on Stormwater Management and the 6th International Conference on Hydraulics in Civil Engineering, Adelaide, Institution of Engineers Australia, pp. 221-226.

Smith, R.A., Alexander, R.B. and Lanfear, K.J. (1994) "Stream Water Quality in the Conterminous United States—Status and Trends of Selected Indicators During the 1980's." In *National Water Summary 1990-1991 – Stream Water Quality*, US Geological Survey water Supply Paper 2400.

Snelder, T. and Trueman, S. (1995) "The Environmental Impacts of Urban Stormwater Runoff." ARC Technical Publication No. 53, Auckland Regional Council, 95 pp.

Somes, N.L.G. and Wong, T.H.F. (1994) "Design Criteria for a Trial Wetland to Improve the Water Quality from a Small Rural Catchment." Water Down Under '94, Adelaide, Australia, pp. 403-408.

Somes, N.L.G. and Wong, T.H.F. (1998) "Rainfall Analysis for Stormwater Wetland Design in Australian Capital Cities." In HydraStorm '98, Proceedings of the 3rd International Symposium on Stormwater Management and the 6th International Conference on Hydraulics in Civil Engineering, Adelaide, Institution of Engineers Australia, pp. 361-366.

Spear, P.A. and Pearce, R.C. (1979) "Copper in the Aquatic Environment: Chemistry, Distribution, and Toxicology." NRCC No. 16454, Associate Committee on Scientific Criteria for Environmental Quality, National Research Council of Canada, Ottawa.

Swanson, P. (1994) "Performance Evaluation of a Constructed Wetland for Urban Runoff Treatment." Australian Water Technologies for the (Sydney) Water Board, Sydney, 92/70.

Tomlinson, G.W., Fisher, A.G. and Clark, R.D.S. (1993) "Stormwater – A Community Resource. The Paddocks." Engineering and Water Supply Department, EWS Report No. 93/11, 52 pp.

Unidata (1996) "STARFLOW Ultrasonic Doppler Instrument User's Manual" Revision D, MacLaren Enterprises, Australia, 101 pp.

US Environmental Protection Agency (1976) "Quality Criteria for Water." Environmental Protection Agency, Washington DC, U.S., 256p

US Environmental Protection Agency (1983) "Final Report of the Nationwide Urban Runoff Program." Prepared by the Water Planning Division, WH 554, Washington DC.

US Environmental Protection Agency (1986) "Quality Criteria for Water - 1986." Environmental Protection Agency, Office of Water, EPA 440/5-86-001, Washington DC, U.S.

US Environmental Protection Agency (1987) "Methodology for Analysis of Detention Basins for Control of Urban Runoff Quality." US Environmental Protection Agency, EPA 440/5-87-001.

Vanoni, V.A. (1977) "Sedimentation Engineering." American Society of Civil Engineers.

Ward, N.I. (1990) "Multielement Contamination of British Motorway Environments." *Science of the Total Environment*, 93, Elsevier Publishing Company, Amsterdam, pp. 393.

Warnick, S.L. and Bell, H.L. (1969) "The Acute Toxicity of Some Heavy Metals to Different Species of Aquatic Insects." *Journal of the Water Pollution Control Federation*, 4, pp. 280-284.

Weir, J. (1976) "Natural and Agricultural Control." Proceedings of AWRC Symposium on Eutrophication, Australian Government Publishing Service, Canberra, pp. 55-67.

Wetzel, R.G. (1983) "Limnology." Second Edition, W.B. Saunders Company, New York, 767 pp.

White, M.J and Cattell, F.C.R. (1992) "AUSQUAL, An Australian Water Quality Management Model." International Symposium on Urban Stormwater Management, Sydney, Australia, Institution of Engineers Australia, pp.396-401.

Williams, B.G. (1997) "Monitoring Urban Stormwater Inflows to the Barker Inlet Wetland in Adelaide, South Australia." A Thesis for the Degree of Master of Engineering Science, Department of Civil and Environmental Engineering, The University of Adelaide.

Wong, T.H.F. and Geiger, W.F. (1997) "Adaptation of Wastewater Surface Flow Wetland Formulae for Application in Constructed Stormwater Wetlands." *Ecological Engineering*, 9, pp. 187-202.

Wong, T.H.F. and Walker, T.A. (1998) "Effectiveness of Street Sweeping in Stormwater Pollution Control." CRC for Catchment Hydrology Report.

Wong, T.H.F. and Somes, N.L.G. (1997) "The Contribution of Stormwater Wetlands and Swales to Urban Catchment Management." Proceedings 24th Hydrology and Water Resources Conference, Auckland, 89-94.

Wu, J.S., Holman, R.E. and Dorney, J.R. (1996) "Systematic Evaluation of Pollutant Removal by Urban Wet Detention Ponds." *Journal of Environmental Engineering*, Vol. 122, No. 11, American Society of Civil Engineers, pp. 983-988.

Yousef, Y.A., Wanielista, M.P. and Harper, H.H. (1986) "Design and Effectiveness of Urban Retention Basins." In B. Urbonas and L. Roesner (Eds.) *Urban Runoff Quality – Impact and Quality Enhancement Technology, Proceedings of and Engineering Foundation Conference*, New Hampshire, American Society of Civil Engineers, New York, pp. 338 350.

Zhang, T., Ellis, J.B., Revitt, D.M. and Shutes, R.B. (1990) "Metal Uptake and Associated Pollution Control by *Typha latifolia* in Urban Wetlands." In P.F. Cooper and B.C. Findlater (Eds.), Proceedings of the International Conference on the Use of Constructed Wetlands in Water Pollution Control, Pergamon Press, Oxford, pp. 451-459.

Appendix A

DETAILS FOR FLOW CALCULATIONS

Contents:

Weir coefficients used in flow calculations

Dynamic Stage/discharge rating curve
for North Arm East Monitoring Station

Stage/discharge rating curve for South Road
Connector Station

Table A.1 Values of the ratio y_c/H_1 as a function of z_c and H_1/b_c for trapezoidal control sections (1)

	Side Slopes of channel, ratio of horizontal to vertical (z_c:1)									
H_1/b_c	Vertical	0.25:1	0.50:1	0.75:1	1:1	1.5:1	2:1	2.5:1	3:1	4:1
0.00	0.667	0.667	0.667	0.667	0.667	0.667	0.667	0.667	0.667	0.667
0.01	0.667	0.667	0.667	0.668	0.668	0.669	0.670	0.670	0.671	0.672
0.02	0.667	0.667	0.668	0.669	0.670	0.671	0.672	0.674	0.675	0.678
0.03	0.667	0.668	0.669	0.670	0.671	0.673	0.675	0.677	0.679	0.683
0.04	0.667	0.668	0.670	0.671	0.672	0.675	0.677	0.680	0.683	0.687
0.05	0.667	0.668	0.670	0.672	0.674	0.677	0.680	0.683	0.686	0.692
0.06	0.667	0.669	0.671	0.673	0.675	0.679	0.683	0.686	0.690	0.696
0.07	0.667	0.669	0.672	0.674	0.676	0.681	0.685	0.689	0.693	0.699
0.08	0.667	0.670	0.672	0.675	0.678	0.683	0.687	0.692	0.696	0.703
0.09	0.667	0.670	0.673	0.676	0.679	0.684	0.690	0.695	0.698	0.706
0.10	0.667	0.670	0.674	0.677	0.680	0.686	0.692	0.697	0.701	0.709
0.12	0.667	0.671	0.675	0.679	0.684	0.690	0.696	0.701	0.706	0.715
0.14	0.667	0.672	0.676	0.681	0.686	0.693	0.699	0.705	0.711	0.720
0.16	0.667	0.672	0.678	0.683	0.687	0.696	0.703	0.709	0.715	0.725
0.18	0.667	0.673	0.679	0.684	0.690	0.698	0.706	0.713	0.719	0.729
0.20	0.667	0.674	0.680	0.686	0.692	0.701	0.709	0.717	0.723	0.733
0.22	0.667	0.674	0.681	0.688	0.694	0.704	0.712	0.720	0.726	0.736
0.24	0.667	0.675	0.683	0.689	0.696	0.706	0.715	0.723	0.729	0.739
0.26	0.667	0.676	0.684	0.691	0.698	0.709	0.718	0.725	0.732	0.742
0.28	0.667	0.676	0.685	0.693	0.699	0.711	0.720	0.728	0.734	0.744
0.30	0.667	0.677	0.686	0.694	0.701	0.713	0.723	0.730	0.737	0.747
0.32	0.667	0.678	0.687	0.696	0.703	0.715	0.725	0.733	0.739	0.749
0.34	0.667	0.678	0.689	0.697	0.705	0.717	0.727	0.735	0.741	0.751
0.36	0.667	0.679	0.690	0.699	0.706	0.719	0.729	0.737	0.743	0.752
0.38	0.667	0.680	0.691	0.700	0.708	0.721	0.731	0.738	0.745	0.754
0.40	0.667	0.680	0.692	0.701	0.709	0.723	0.733	0.740	0.747	0.756
0.42	0.667	0.681	0.693	0.703	0.711	0.725	0.734	0.742	0.748	0.757
0.44	0.667	0.681	0.694	0.704	0.712	0.727	0.766	0.744	0.750	0.759
0.46	0.667	0.682	0.695	0.705	0.714	0.728	0.737	0.745	0.751	0.760
0.48	0.667	0.683	0.696	0.706	0.715	0.729	0.739	0.747	0.752	0.761
0.50	0.667	0.683	0.697	0.708	0.717	0.730	0.740	0.748	0.754	0.762
0.60	0.667	0.686	0.701	0.713	0.723	0.737	0.747	0.754	0.759	0.767
0.70	0.667	0.688	0.706	0.718	0.728	0.742	0.752	0.758	0.764	0.771
0.80	0.667	0.692	0.709	0.723	0.732	0.746	0.756	0.762	0.767	0.774
0.90	0.667	0.694	0.713	0.727	0.737	0.750	0.759	0.766	0.770	0.776
1.0	0.667	0.697	0.717	0.730	0.740	0.754	0.762	0.768	0.773	0.778
1.2	0.667	0.701	0.723	0.737	0.747	0.759	0.767	0.772	0.776	0.782
1.4	0.667	0.706	0.729	0.742	0.752	0.764	0.771	0.776	0.779	0.784
1.6	0.667	0.709	0.733	0.747	0.756	0.767	0.774	0.778	0.781	0.786
1.8	0.667	0.713	0.737	0.750	0.759	0.770	0.776	0.781	0.783	0.787
2.0	0.667	0.717	0.740	0.754	0.762	0.773	0.778	0.782	0.785	0.788
3.0	0.667	0.730	0.753	0.766	0.773	0.781	0.785	0.787	0.790	0.792
4.0	0.667	0.740	0.762	0.773	0.778	0.785	0.788	0.790	0.792	0.794
5.0	0.667	0.748	0.768	0.777	0.782	0.788	0.791	0.792	0.794	0.795
10	0.667	0.768	0.782	0.788	0.791	0.794	0.795	0.796	0.797	0.798
∞		0.800	0.800	0.800	0.800	0.800	0.800	0.800	0.800	0.800

(1) *Bos (1989)*

Table A.2 **Dynamic stage/discharge rating tables for North Arm East Drain – AU504101**

Stage Height (m)	Non-backwatered Discharge (m^3/s)	Backwatered Discharge (m^3/s)
0.105	0.0000	0.0000
0.115	0.0003	0.0003
0.125	0.0015	0.0015
0.135	0.0042	0.0041
0.145	0.0085	0.0085
0.155	0.0149	0.0149
0.165	0.0234	
0.168	0.0265	
0.205	0.129	0.0529
0.305	1.06	0.22
0.355	1.72	
0.405	2.47	0.58
0.455	3.32	
0.505	4.24	1.26
0.555	5.24	
0.605	6.32	2.39
0.655	7.46	
0.705	8.66	3.84
0.755	9.93	
0.805	11.26	5.84
0.855	12.64	
0.905	14.09	8.76
0.955	15.58	
1.005	17.13	
1.055	18.74	
1.105	20.39	
1.155	22.10	
1.205	23.85	14.97
1.255	25.66	
1.305	27.51	
1.505		19.13

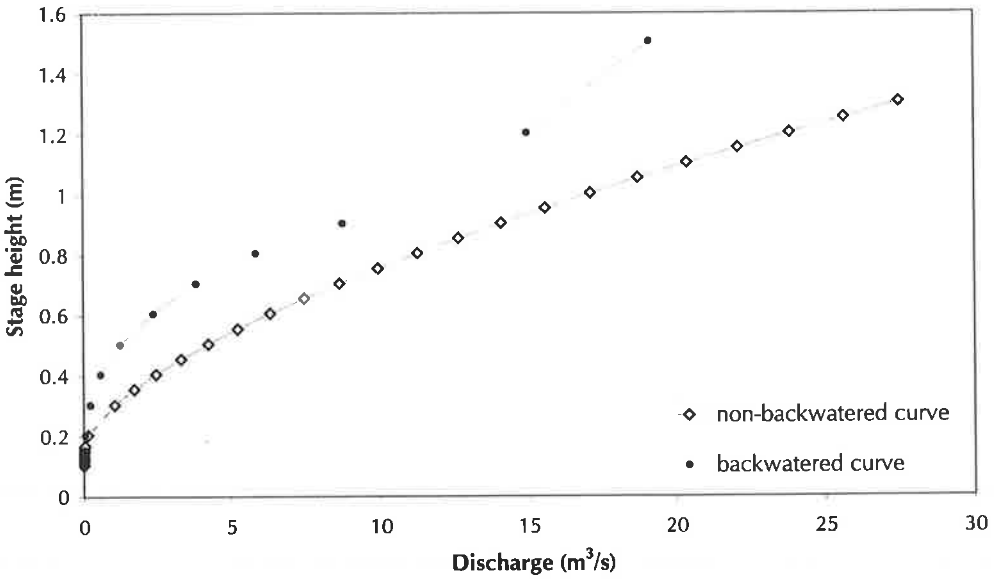


Figure A.1 **Dynamic stage/dischrage rating tables for North Arm East Drain**
(Source: Williams, 1997)

Table A.3 Values for C_e as a function of the ratios b_c/B_1 and h_1/P_1 (1)

b_c/B_1	C_e			
1.0	0.602	+	0.075	h_1/p_1
0.9	0.599	+	0.064	h_1/p_1
0.8	0.597	+	0.045	h_1/p_1
0.7	0.595	+	0.030	h_1/p_1
0.6	0.593	+	0.018	h_1/p_1
0.5	0.592	+	0.011	h_1/p_1
0.4	0.591	+	0.0058	h_1/p_1
0.3	0.590	+	0.0020	h_1/p_1
0.2	0.589	-	0.0018	h_1/p_1
0.1	0.588	-	0.0021	h_1/p_1
0.0	0.587	-	0.0023	h_1/p_1

(1) Bos (1989)

See Figure A.2 for definition of terms.

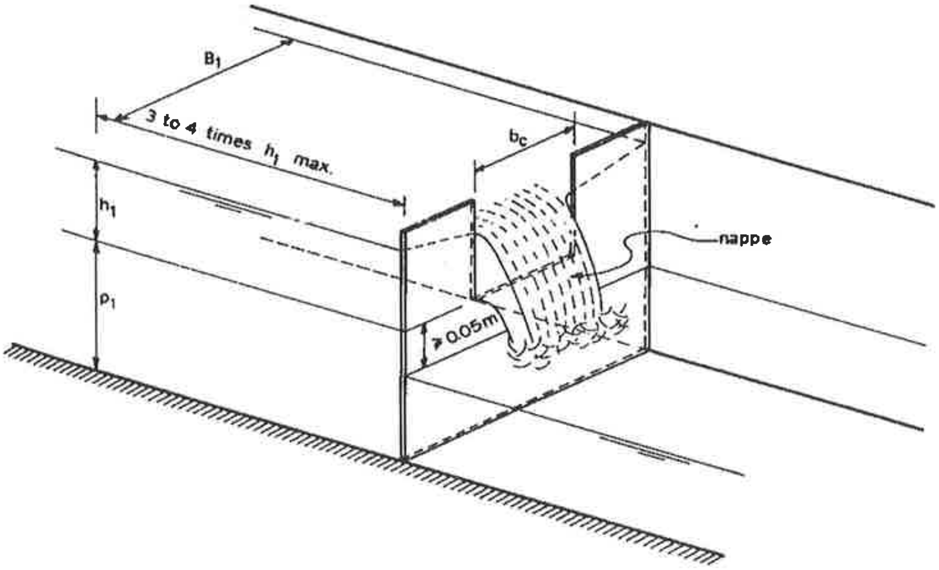
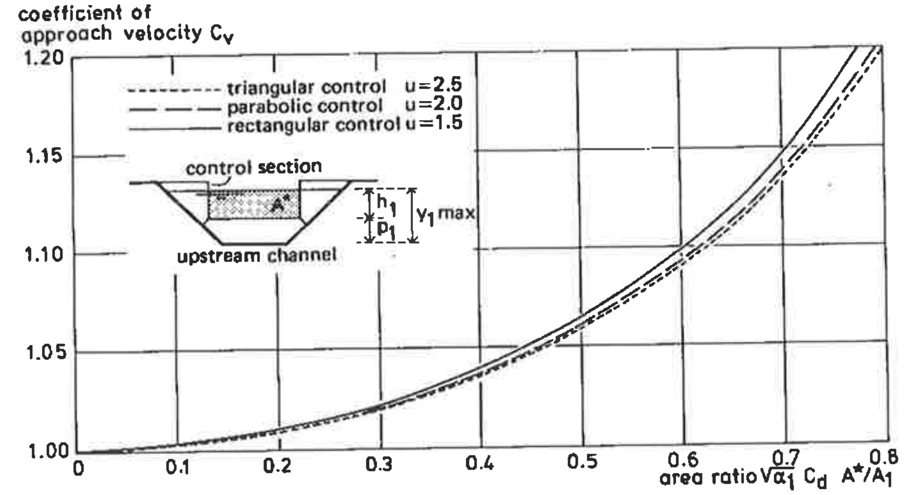


Figure A.2 Definition of terms for sharp crested weir
(Source: Bos, 1989)



A^* = wetted area at control section of water depth equals $y = h_1$

A_1 = wetted area at head measurement station

Figure A.3 C_v-values as a function of area ratio $\sqrt{\alpha_1}C_dA^*/A_1$ (Source: Bos, 1977)

Table A.4 South Road Connector stage/discharge rating table

Depth	Outflow
0.805	0
0.815	0.02
0.825	0.07
0.843	0.22
0.85	0.30
0.855	0.35
0.865	0.40
0.875	0.50
0.895	0.72
0.915	1.05
0.938	1.30
0.96	1.75
1.005	2.52
1.055	3.80
1.141	6.13
1.178	7.37
1.205	8.74
1330	13.60
1.430	18.0
1.535	23.0
1.635	28.0

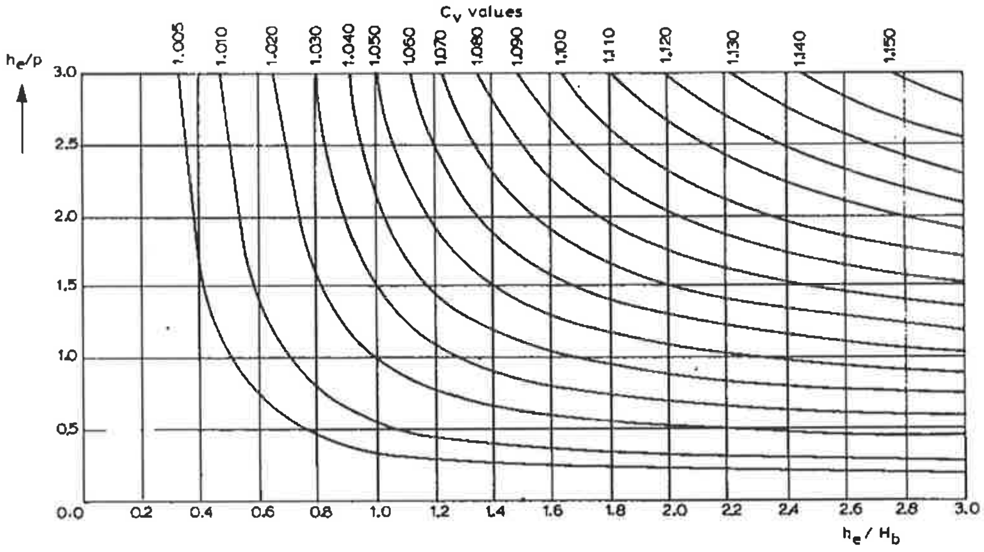


Figure A.4 C_v –values as a function of h_e/p and h_e/H_b for triangular profile flat v-weirs (source: Bos, 1989; after White,1971)

Table A.5 K_h**-values for triangular profile flat V-weirs** (1)

Weir profile	1-to-20 cross slope	1-to-10 cross slope
1-to-2/1-to-2	0.0004 m	0.0006 m
1-to-2/1-to-5	0.0005 m	0.0008 m

(1) Bos (1989), after White, 1971

Appendix B

EVENT MEAN CONCENTRATION AND EVENT LOAD DATA

Contents:

North Arm East event mean concentration data

North Arm East event load data

South Road Connector event mean concentration data

South Road Connector event load data

Table B.1 North Arm East event mean concentraton (EMC) data

		NUTRIENT CONCENTRATIONS (mg/L)							METAL CONCENTRATIONS (mg/L)													
Storm Date	Volume (m^3)	Ammonia (as N)	TKN (as N)	Nitrate + Nitrite (as N)	Total Nitrogen	Filt. React. Phos. (as P)	Particulate P	Phosphorous (Total as P)	Aluminium (Total)	Arsenic (Inorganic)	Cadmium (Total)	Chromium (Total)	Copper (Total)	Iron (Total)	Lead (Total)	Manganese (Total)	Mercury (Total)	Nickel (Total)	Zinc (Total)	TSS (mg/L)	EC (µS/cm)	TDS (mg/L)
14-01-97	11556	0.669	6.07	0.014	6.08	0.297	0.556	0.853	0.51	0.002	0.001	0.006	0.062	1.41	0.145	0.274	0.0001	0.01	0.824			
21-01-97	149446	NN							1.30	0.003	0.0003	0.012	0.076	1.06	0.046	0.045	0.0005	0.004	0.22			
06-02-97	245124	0.050	2.20	0.116	2.32	0.051	0.369	0.420	3.83	0.005	0.0015	0.01	0.051	3.84	0.224	0.117	<0.0001	0.007	0.616	267		
28-02-97	31281	NR																				
02-05-97	7213	0.860	8.05	0.101	8.15	0.342	1.598	1.940	3.37	0.003	0.0015	0.038	0.087	4.24	0.234	0.19	0.0002	0.01	0.852	590.1		
03-05-97	32069	0.120	2.66	0.348	3.01	0.150	0.325	0.475	1.62	0.003	0.0009	0.03	0.055	1.74	0.117	0.056	0.0001	0.005	0.527	136.9		
04-05-97	39438	0.170	2.06	0.342	2.40	0.125	0.307	0.432	1.83	0.004	0.0011	0.036	0.047	2.23	0.143	0.062	0.0001	0.007	0.366	270.5		
06-05-97	9808	0.016	1.26	0.277	1.54	0.082	0.173	0.255	2.25	0.003	0.0006	0.006	0.021	2.03	0.1	0.054	<0.0001	0.005	0.358	119.9		
18-05-97	14827	NR																				
28-05-97	21420	0.390	2.17	0.255	2.43	0.191	0.310	0.501	3.51	0.006	0.0008	< 0.005	0.052	2.91	0.137	0.075	0.0004	0.012	0.655	279.1		
29-05-97	15051	0.035	1.41	0.255	1.67	0.105	0.255	0.360	3.61	0.005	0.0008	0.015	0.041	3.4	0.166	0.095	0.0002	0.006	0.517	73.6		
13-06-97	77648	0.068	2.25	0.366	2.61	0.059	0.498	0.558	7.38	0.007	0.001	0.018	0.079	6.656	0.316	0.178	< 0.0001	0.011	0.860	828.1		
06-08-97	46724	0.200	2.36	0.390	2.75	0.051	0.383	0.434	4.70	0.005	0.0012	0.009	0.055	4.44	0.372	0.098	0.0001	0.007	0.7	243.3		
07-08-97	50902	0.110	1.35	0.314	1.66	0.049	0.273	0.322	6.14	0.007	0.0009	0.015	0.056	5.19	0.483	0.11	0.0001	0.007	0.531	774.1		
08-08-97	37642	0.052	0.98	0.277	1.26	0.050	0.168	0.218	2.61	0.005	0.0005	0.006	0.033	2.41	0.152	0.06	0.0001	0.004	0.401	169.7		
10-08-97	83258	0.089	0.77	0.351	1.12	0.046	0.098	0.144	2.54	0.003	0.0004	< 0.005	0.021	1.78	0.059	0.034	0.0001	0.003	0.261	83.4		
11-08-97	24470	0.029	1.15	0.213	1.36	0.049	0.202	0.251	4.41	0.006	0.0006	0.01	0.036	3.71	0.106	0.082	0.0001	0.005	0.401	225.8		
12-08-97	2942	0.130	1.16	0.366	1.53	0.075	0.105	0.180	3.00	0.003	0.0004	0.006	0.02	2.07	0.048	0.045	0.0001	0.003	0.271	277.2		
30-08-97	91025	0.200	1.84	0.262	2.10	0.069	0.311	0.380	2.72	0.003	0.0014	0.02	0.054	2.8	0.185	0.084	0.0001	0.006	0.534	175.6		
01-09-97	68220	0.100	1.05	0.256	1.31	0.064	0.161	0.225	2.99	0.004	0.0008	0.008	0.037	2.67	0.143	0.064	0.0001	0.005	0.387	150.3		
02-09-97	22493	0.130	0.89	0.389	1.28	0.055	0.104	0.159	1.16	0.003	0.0023	0.006	0.033	1.08	0.084	0.033	0.0001	0.004	0.327	73.5		
04-09-97	28899	0.110	1.39	0.302	1.69	0.051	0.201	0.252	2.72	0.004	0.0008	0.013	0.033	2.55	0.157	0.067	0.0001	0.005	0.459	156.4		
06-09-97	66230	0.090	1.33	0.406	1.73	0.038	0.188	0.227	2.94	0.004	0.0009	0.013	0.042	2.66	0.144	0.066	0.0001	0.006	0.469	124.5		
13-09-97	30067	0.160	1.47	0.302	1.77	0.043	0.229	0.272	2.48	0.003	0.001	0.015	0.048	2.47	0.126	0.062	<0.0001	0.006	0.496	164		
14-09-97	26565	0.120	0.79	0.328	1.12	0.038	0.093	0.131	1.50	0.003	0.0006	0.006	0.025	1.39	0.064	0.029	<0.0001	0.004	0.273	63.2		
18-09-97	79696	0.040	2.32	0.226	2.55	0.033	0.686	0.719	12.50	0.007	0.0032	0.035	0.131	12.4	0.636	0.272	0.0002	0.017	1.26	662.4		
30-10-97	762244	0.426	2.09	0.244	2.33	0.277	0.280	0.557	3.91	0.004	0.0008	0.018	0.058	3.31	0.132	0.078	0.0001	0.006	0.404	185.4		
14-11-97	47443	0.280	1.98	0.158	2.14	0.083	0.368	0.451	4.25	0.004	0.0014	0.025	0.072	4	0.189	0.113	0.0002	0.007	0.54			
08-12-97	24394	0.590	4.26	0.038	4.30	0.087	0.799	0.886	6.51	0.005	0.0024	0.03	0.111	6.46	0.357	0.162	0.0002	0.013	0.83	298.4		
18-12-97	95959	NR																				
23-03-98	7421	NR																				
11-04-98	133062	0.145	1.10	0.289	1.39	0.070	0.154	0.224	1.55	0.002	0.0006	< 0.005	0.036	1.39	0.057	0.049	<0.0001	0.004	0.312	88.2		
12-04-98	33138	0.350	1.40	0.166	1.57	0.063	0.137	0.200	0.63	0.05	0.0006	< 0.005	0.01	0.752	0.02	0.072	<0.0001	0.003	0.081	36.8		
19-04-98	187084	NR																				
21-04-98	20148	0.172	2.04	0.359	2.40	0.112	0.347	0.459	2.33	0.004	0.0009	0.007	0.029	2.99	0.119	0.083	0.0001	0.009	0.476			
24-04-98	49215	0.165	1.40	0.323	1.72	0.082	0.187	0.269	3.56	0.003	0.0005	0.011	0.031	2.98	0.102	0.077	<0.0001	0.005	0.449	200.1		
25-04-98	25508	0.390	1.18	0.427	1.61	0.098	0.084	0.182	1.64	0.003	0.0004	0.007	0.019	1.35	0.033	0.03	<0.0001	0.003	0.209	47.6		
14-05-98	3223	0.048	1.18	< 0.05		0.350	0.322	0.672	0.83	0.002	0.0008	< 0.005	0.016	1.05	0.033	0.087	0.0001	0.005	0.265	62.5		
19-05-98	51503	0.160	2.03	0.307	2.34	0.081	0.350	0.431	3.70	0.004	0.0009	0.012	0.044	3.32	0.165	0.097	<0.0001	0.009	0.478	274.9		
22/05/98 (a)	38475	0.032	1.24	0.350	1.59	0.052	0.220	0.272	3.07	0.003	0.0010	0.015	0.045	2.68	0.111	0.072	< 0.0001	0.006	0.422	160.9		
22/05/98 (b)	5109	0.028	0.95	0.468	1.42	0.066	0.143	0.209	2.11	0.003	0.0005	0.007	0.021	1.63	0.049	0.038	<0.0001	0.004	0.253	78.2		
23-05-98	5920	0.079	0.79	0.389	1.18	0.072	0.096	0.168	0.84	0.003	0.0005	< 0.005	0.02	0.681	0.031	0.028	<0.0001	0.002	0.175	54.8		
05-06-98	1821	0.450		0.150	0.15	0.147			1.02	0.001	0.0009	0.006	0.02	1.28	0.051	0.118	0.0004	0.005	0.376	59.2	430	778
06-06-98	91624	0.165	1.25	0.370	1.62	0.084	0.249	0.333	2.28	0.004	0.0008	0.010	0.029	1.93	0.128	0.063	0.0004	0.005	0.375	124.1	91	167
11/06/98 (a)	13324	0.292	1.91	0.457	2.37	0.152	0.239	0.391	2.28	0.003	0.0009	0.006	0.027	1.94	0.052	0.071	0.0001	0.004	0.427	99.2	170	301
11/06/98 (b)	18832	0.060	1.47	0.437	1.91	0.085	0.241	0.326	2.80	0.005	0.0007	0.008	0.028	2.35	0.09	0.068	0.0002	0.004	0.398	144	90	164
11/06/98 (c)	62190	0.040	1.45	0.280	1.73	0.048	0.283	0.331	4.97	0.005	0.0007	0.008	0.014	3.79	0.134	0.103	0.0001	0.006	0.419	279.3	76	138
11/06/98 (d)	4597	0.445	1.93	0.520	2.45	0.640	0.356	0.996	2.03	0.005	0.0004	0.011	0.033	1.78	0.045	0.028	<0.0001	0.004	0.286	70.3	210	378
14-06-98	4234	0.155	1.32	0.442	1.76	0.310	0.187	0.497	1.58	0.003	0.0006	0.012	0.035	1.48	0.04	0.044	<0.0001	0.004	0.267	165.9	240	431
15-06-98	5833	NR																		33.34		
20-06-98	5723	0.620	2.18	0.583	2.76	0.427	0.253	0.680	0.91	0.002	< 0.0002	0.009	0.027	0.979	0.032	0.051	<0.0001	0.004	0.338	62.5	310	560
21-06-98	61038	0.093	0.89	0.416	1.31	0.076	0.145	0.221	1.31	0.002	< 0.0002	0.006	0.018	1.21	0.052	0.034	<0.0001	0.003	0.343	64.5	94	171
22-06-98	6949	0.112	1.25	0.374	1.62	0.102	0.158	0.260	2.24	0.001	< 0.0002	< 0.005	0.015	1.85	0.045	0.039	<0.0001	0.004	0.292	53.3	130	230
23-06-98	6775	0.030	0.95	0.382	1.33	0.111	0.141	0.252	2.34	0.003	< 0.0002	0.005	0.021	1.98	0.049	0.044	0.0003	0.005	0.307	53.2	210	391
24-06-98	22641	0.016	2.20	0.312	2.51	0.063	0.427	0.490	9.33	0.003	< 0.0002	0.016	0.049	7.94	0.178	0.148	0.0001	0.01	0.771	385.9	140	250
25-06-98	baseflow	0.320	1.55	0.158	1.71	0.145	0.266	0.411	2.13	0.017	0.0009	0.009	0.039	2.13	0.111	0.057	<0.0001	0.005	0.3		170	309
29-06-98	37816	0.065	1.62	0.328	1.95	0.101	0.270	0.371	4.97	0.005	0.0009	0.015	0.039	4.42	0.206	0.107	<0.0001	0.006	0.498	125.2	140	250
05-07-98	34038	0.055	1.59	0.444	2.03	0.079	0.277	0.356	3.50	0.003	0.0007	0.012	0.045	3.04	0.199	0.095	0.0001	0.006	0.552	94	110	197
08-07-98	30268	NR																				
21-07-98	22157	NN							5.16	0.004	0.0013	0.014	0.062	5	0.215	0.156	0.0003	0.008	0.83	259.8	190	338
25-07-98	6676	0.560	2.99	0.478	3.47	0.810	0.430	1.240	1.27	0.003	0.0009	0.014	0.063	1.38	0.067	0.108	<0.0001	0.005	0.603	72.7	240	437
27-07-98	33576	1.520	4.52	0.235	4.76	0.266	1.014	1.280	9.23	0.008	0.0021	0.055	0.141	9.85	0.342	0.341	0.0004	0.015	1.24	126.4	210	381
27-07-98	58648+	NR																				
28-07-98	13624	0.190	3.13	0.229	3.36	0.140	0.526	0.666	7.12	0.007	0.0013	0.035	0.095	6.19	0.192	0.154	0.0003	0.009	0.722	336.7	120	218

Table B.1 North Arm East event mean concentraton (EMC) data

		NUTRIENT CONCENTRATIONS (mg/L)							METAL CONCENTRATIONS (mg/L)													
Storm Date	Volume (m^3)	Ammonia (as N)	TKN (as N)	Nitrate + Nitrite (as N)	Total Nitrogen	Filt. React. Phos. (as P)	Particulate P	Phosphorous (Total as P)	Aluminium (Total)	Arsenic (Inorganic)	Cadmium (Total)	Chromium (Total)	Copper (Total)	Iron (Total)	Lead (Total)	Manganese (Total)	Mercury (Total)	Nickel (Total)	Zinc (Total)	TSS (mg/L)	EC (µS/cm)	TDS (mg/L)
28-07-98	28902	0.120	2.24	0.301	2.54	0.152	0.295	0.447	2.79	0.005	0.0004	0.018	0.045	2.32	0.096	0.064	<0.0001	0.004	0.439	132.5	140	261
29-07-98	69564	0.110	1.19	0.360	1.55	0.127	0.120	0.247	1.12	0.003	0.0004	0.006	0.019	1.02	0.041	0.03	<0.0001	0.002	0.275	39	150	277
30-07-98	10654	0.124	2.50	0.469	2.97	0.177	0.363	0.540	4.35	0.006	0.0007	0.011	0.04	4.09	0.082	0.113	<0.0001	0.006	0.432	200	230	414
03-08-98	29748	NN							0.94	0.003	0.0005	< 0.005	0.033	0.607	0.028	0.02	0.0001	0.003	0.238			
06-08-98	7472		1.77	0.498	2.27		0.366	0.366						1.51	0.07				0.437			
06-08-98	26734		1.76	0.485	2.25		0.363	0.363						4.15	0.224				0.64			
20-08-98	51249	0.035	1.60	0.375	1.98	0.045	0.282	0.327	3.31	0.004	0.0009	0.013	0.049	3.35	0.142	0.094	< 0.0001	0.008	0.567	187	99	180
21-08-98	40343	0.052	1.02	0.320	1.34	0.050	0.154	0.204	3.27	0.003	0.0005	0.014	0.033	2.76	0.068	0.049	< 0.0001	< 0.001	0.393	90	100	190
23-08-98	30088	0.034	0.68	0.281	0.96	0.049	0.219	0.268	4.73	0.003	0.001	0.032	0.072	4.46	0.117	0.097	< 0.0001	0.007	0.548	179	120	212
SUMMARY STATISTICS																						
Mean		0.208	1.862	0.322	2.165	0.133	0.307	0.435	3.258	0.005	0.001	0.015	0.044	2.992	0.137	0.088	0.0002	0.006	0.473	186.8	168.4	304.9
Maximum		1.52	8.05	0.583	8.151	0.81	1.598	1.94	12.6	0.05	0.0032	0.055	0.141	12.4	0.636	0.341	0.0005	0.017	1.26	828.1	430.0	778.0
Minimum		0.016	0.68	< 0.05	0.15	0.033	0.084	0.131	0.827	0.001	<0.0002	< 0.005	0.01	0.607	0.02	0.02	0.0001	< 0.001	0.081	33.3	76.0	138.0
Median		0.12	1.47	0.328	1.772	0.082	0.266	0.36	2.79	0.003	0.0008	0.012	0.039	2.47	0.117	0.072	0.0001	0.005	0.427	140.4	140.0	261.0
Standard Deviation		0.253	1.236	0.115	1.213	0.142	0.238	0.313	2.264	0.006	0.001	0.010	0.027	2.140	0.112	0.061	0.000	0.003	0.225	169.4	80.7	145.7
Count		60	61	61	61	60	61	61	63	63	58	55	63	65	65	63	36	62	65	58.0	25.0	25.0
C_v (=σ/x)		1.217	0.664	0.358	0.560	1.066	0.777	0.718	0.695	1.295	0.582	0.705	0.599	0.715	0.821	0.695	0.656	0.495	0.476	0.9	0.5	0.5

Table B.2 North Arm East event load data

		NUTRIENT LOADS (g)							METAL LOADS (g)												
Storm Date	Volume (m^3)	Ammonia (as N)	TKN (as N)	Nitrate + Nitrite (as N)	Total Nitrogen	Filt. React. Phos. (as P)	Particulate P	Phosphorous (Total as P)	Aluminium (Total)	Arsenic (Inorganic)	Cadmium (Total)	Chromium (Total)	Copper (Total)	Iron (Total)	Lead (Total)	Manganese (Total)	Mercury (Total)	Nickel (Total)	Zinc (Total)	TSS (mg/L)
14-01-97	11556	7731.0	70144.9	161.8	70306.7	3432.1	6425.1	9857.3	10516.0	23.1	11.6	69.3	716.5	16294.0	1675.6	3166.3	1.2	115.6	9522.1	1473040.0
21-01-97	149446	15290.0	238080.0	39910.0		8310.0	8311.0	110610.0	194279.8	448.3	44.8	1793.4	11357.9	158412.8	6874.5	6725.1	74.7	597.8	32878.1	35212290.0
06-02-97	245124	12256.2	539272.8	28434.4	567707.2	12501.3	90450.8	102952.1	938824.9	1225.6	367.7	2451.2	12501.3	941276.2	54907.8	28679.5	28.0	1715.9	150996.4	65448108.0
28-02-97	31281	3680.0	50860.0	8870.0		2730.0	2731.0	23150.0	86920.0	120.0	30.0	420.0	1190.0	82490.0	3680.0	2400.0	4.0	170.0	14530.0	5063860.0
02-05-97	7213	6203.2	58064.7	728.5	58793.2	2466.8	11526.4	13993.2	24307.8	21.6	10.8	274.1	627.5	30583.1	1687.8	1370.5	1.4	72.1	6145.5	4256391.3
03-05-97	32069	3848.3	85303.5	11160.0	96463.6	4810.4	10422.4	15232.8	51951.8	96.2	28.9	962.1	1763.8	55800.1	3752.1	1795.9	3.2	160.3	16900.4	4388860.8
04-05-97	39438	6704.5	81242.3	13487.8	94730.1	4929.8	12107.5	17037.2	74143.4	157.8	43.4	1419.8	1853.6	87946.7	5639.6	2445.2	3.9	276.1	14434.3	10667303.8
06-05-97	9808	156.9	12358.1	2716.8	15074.9	804.3	1696.8	2501.0	22068.0	29.4	5.9	58.8	206.0	19910.2	980.8	529.6	2.0	49.0	3511.3	1175979.2
08-05-97																				
18-05-97	14827	1870.0	24340.0	4320.0		1600.0	1601.0	10970.0	34700.0	50.0	10.0	180.0	500.0	33430.0	1390.0	1080.0	2.0	80.0	6440.0	2006500.0
21-05-97																				
23-05-97																				
26-05-97																				
28-05-97	21420	8353.8	46481.4	5462.1	51943.5	4091.2	6640.2	10731.4	75184.2	128.5	17.1	270.0	1113.8	62332.2	2934.5	1606.5	8.6	257.0	14030.1	5978322.0
29-05-97	15051	526.8	21221.9	3838.0	25059.9	1580.4	3838.0	5418.4	54334.1	75.3	12.0	225.8	617.1	51173.4	2498.5	1429.8	3.0	90.3	7781.4	1107753.6
13-06-97	77648	5252.1	174418.2	28436.8	202854.9	4601.9	38692.5	43294.5	573083.8	519.7	110.6	1416.3	6112.5	516814.9	24502.6	13815.1	2.0	839.6	66748.3	64300308.8
06-08-97	46724	9344.8	110268.6	18222.4	128491.0	2382.9	17895.3	20278.2	219602.8	233.6	56.1	420.5	2569.8	207454.6	17381.3	4579.0	4.7	327.1	32706.8	11367949.2
07-08-97	50902	5599.2	68717.7	15983.2	84700.9	2494.2	13896.2	16390.4	312538.3	356.3	45.8	763.5	2850.5	264181.4	24585.7	5599.2	5.1	356.3	27029.0	39403238.2
08-08-97	37642	1957.4	36889.2	10426.8	47316.0	1882.1	6323.9	8206.0	105774.0	188.2	18.8	225.9	1242.2	90717.2	5721.6	2258.5	3.8	150.6	15094.4	6387847.4
10-08-97	83258	7410.0	64108.7	29223.6	93332.2	3829.9	8159.3	11989.2	211475.3	249.8	33.3	1300.0	1748.4	148199.2	4912.2	2830.8	8.3	249.8	21730.3	6943717.2
11-08-97	24470	709.6	28140.5	5212.1	33352.6	1199.0	4942.9	6142.0	107912.7	146.8	14.7	244.7	880.9	90783.7	2593.8	2006.5	2.4	122.4	9812.5	5525326.0
12-08-97	2942	382.5	3412.7	1076.8	4489.5	220.7	308.9	529.6	8826.0	8.8	1.2	17.7	58.8	6089.9	141.2	132.4	0.3	8.8	797.3	815522.4
30-08-97	91025	18205.0	167486.0	23848.6	191334.6	6280.7	28308.8	34589.5	247588.0	273.1	127.4	1820.5	4915.4	254870.0	16839.6	7646.1	9.1	546.2	48607.4	15983990.0
01-09-97	68220	6822.0	71631.0	17464.3	89095.3	4366.1	10983.4	15349.5	203977.8	272.9	54.6	545.8	2524.1	182147.4	9755.5	4366.1	6.8	341.1	26401.1	10253466.0
02-09-97	22493	2924.1	20018.8	8749.8	28768.5	1237.1	2339.3	3576.4	26091.9	67.5	51.7	135.0	742.3	24292.4	1889.4	742.3	2.2	90.0	7355.2	1653235.5
04-09-97	28899	3178.9	40169.6	8727.5	48897.1	1473.8	5808.7	7282.5	78605.3	115.6	23.1	375.7	953.7	73692.5	4537.1	1936.2	2.9	144.5	13264.6	4519803.6
06-09-97	66230	5945.1	87823.8	26882.3	114706.1	2525.9	12481.7	15007.6	194847.1	234.1	57.0	844.0	2796.9	176072.5	9537.1	4351.0	6.6	402.0	31088.4	8245635.0
13-09-97	30067	4810.7	44198.5	9080.2	53278.7	1292.9	6885.3	8178.2	74566.2	90.2	30.1	451.0	1443.2	74265.5	3788.4	1864.2	4.0	180.4	14913.2	4930988.0
14-09-97	26565	3187.8	20986.4	8713.3	29699.7	1009.5	2470.5	3480.0	39847.5	79.7	15.9	159.4	664.1	36925.4	1700.2	770.4	4.0	106.3	7252.2	1678908.0
18-09-97	79696	3187.8	184894.7	18011.3	202906.0	2630.0	54671.5	57301.4	1004169.6	557.9	255.0	2789.4	10440.2	988230.4	50686.7	21677.3	15.9	1354.8	100417.0	52790630.4
30-10-97	762244	324937.7	1592569.3	185655.3	1778224.6	211203.7	213142.2	424346.0	2979282.3	3208.5	577.9	14004.0	44387.1	2519447.3	100738.5	59454.2	92.2	4413.9	307908.5	141320037.6
14-11-97	47443	13284.0	93937.1	7496.0	101433.1	3937.8	17459.0	21396.8	201632.8	189.8	66.4	1186.1	3415.9	189772.0	8966.7	5361.1	9.5	332.1	25619.2	8487650.0
08-12-97	24394	14392.5	103918.4	927.0	104845.4	2122.3	19490.8	21613.1	158804.9	122.0	58.5	731.8	2707.7	157585.2	8708.7	3951.8	4.9	317.1	20247.0	7279169.6
18-12-97	95959	10220.0	153760.0	26060.0		6060.0	6061.0	17020.0	345060.0	450.0	90.0	1530.0	4310.0	320220.0	16000.0	7950.0	12.0	580.0	49310.0	20329190.0
23-03-98	7421	994.9	12293.0	2221.4		977.4	5492.5	5492.5	14811.6	23.3	5.5	80.5	226.9	14467.1	559.6	513.8	1.2	36.6	3028.7	850564.4
11-04-98	133062	19294.0	146368.2	38454.9	184823.1	9314.3	20491.5	29805.9	206246.1	266.1	79.8		4790.2	184956.2	7584.5	6520.0		532.2	41515.3	11736068.4
12-04-98	33138	11598.3	46393.2	5500.9	51894.1	2087.7	4539.9	6627.5	27405.1	1656.9	19.9		331.4	24919.8	662.8	2385.9		99.4	2684.2	1219478.4
19-04-98	187084	18760.0	297180.0	49540.0		9760.0	9761.0	138470.0	784400.0	980.0	180.0	3290.0	9280.0	718270.0	38370.0	16240.0	22.0	1200.0	102090.0	46522090.0
21-04-98	20148	3472.2	41184.6	7225.3	48409.9	2251.7	6988.5	9240.2	46924.2	80.6	18.0	133.5	586.8	60195.1	2392.5	1672.9	2.0	175.5	9594.9	2934820.0
24-04-98	49215	8120.5	68901.0	15896.4	84797.4	4035.6	9203.2	13238.8	175205.4	147.6	24.6	541.4	1525.7	146660.7	5019.9	3789.6		246.1	22097.5	9847921.5
25-04-98	25508	9948.1	30099.4	10891.9	40991.4	2499.8	2142.7	4642.5	41833.1	76.5	10.2	178.6	484.7	34435.8	841.8	765.2		76.5	5331.2	1214180.8
14-05-98	3223	154.7	3803.1	1000.0	1001.0	1128.1	1037.8	2165.9	2684.8	6.4	2.6		51.6	3384.2	106.4	280.4	0.3	16.1	854.1	201437.5
19-05-98	51503	8240.5	104551.1	15811.4	120362.5	4171.7	18026.1	22197.8	190561.1	206.0	46.4	618.0	2266.1	170990.0	8498.0	4995.8		463.5	24618.4	14158174.7
22/05/98 (a)	38475	1236.3	47837.0	13466.3	61303.3	2011.0	8451.6	10462.5	118195.7	118.0	37.4	561.7	1746.7	102958.5	4255.2	2772.7		228.3	16220.7	6190627.5
22/05/98 (b)	5109	143.1	4853.6	2391.0	7244.6	337.2	730.6	1067.8	10780.0	15.3	2.6	35.8	107.3	8327.7	250.3	194.1		20.4	1292.6	399523.8
23-05-98	5920	467.7	4676.8	2302.9	6979.7	426.2	568.3	994.6	4955.0	17.8	3.0		118.4	4031.5	183.5	165.8		11.8	1036.0	324416.0
05-06-98	1821	819.5	3070.0	273.2	273.2	267.7	0.0	1350.0	1857.4	1.8	1.6	10.9	36.4	2330.9	92.9	214.9	0.7	9.1	684.7	107803.2
06-06-98	91624	15073.3	114141.7	33904.4	148046.1	7667.1	22860.0	30527.1	209018.9	366.5	73.5	873.9	2664.1	176927.0	11734.8	5818.1	36.6	413.5	34370.5	11370538.4
11/06/98 (a)	13324	3890.6	25448.8	6089.1	31537.9	2025.2	3184.4	5209.7	30378.7	40.0	12.0	79.9	359.7	25848.6	692.8	946.0	1.3	53.3	5689.3	1321740.8
11/06/98 (b)	18832	1129.9	27683.0	8229.6	35912.6	1600.7	4538.5	6139.2	52729.6	94.2	13.2	150.7	527.3	44255.2	1694.9	1280.6	3.8	75.3	7495.1	2711808.0
11/06/98 (c)	62190	2487.6	90175.5	17413.2	107588.7	2985.1	17599.8	20584.9	309084.3	311.0	43.5	497.5	870.7	235700.1	8333.5	6405.6	6.2	373.1	26057.6	17369667.0
11/06/98 (d)	4597	2045.7	8872.2	2390.4	11262.7	2942.1	1636.5	4578.6	9331.9	23.0	1.8	50.6	151.7	8182.7	206.9	128.7		18.4	1314.7	323169.1
14-06-98	4234	656.3	5588.9	1871.4	7460.3	1312.5	791.8	2104.3	6689.7	12.7	2.5	50.8	148.2	6266.3	169.4	186.3		16.9	1130.5	702420.6
15-06-98	5833	800.0	9690.0	1760.0		820.0	821.0	4320.0	11010.0	20.0		60.0	170.0	10810.0	410.0	400.0	1.0	30.0	2330.0	194472.2
20-06-98	5723	3548.3	12476.1	3336.5	15812.6	2443.7	1447.9	3891.6	5179.3	11.4		51.5	154.5	5602.8	183.1	291.9		22.9	1934.4	357687.5
21-06-98	61038	5676.5	54323.8	25391.8	79715.6	4638.9	8850.5	13489.4	79959.8	122.1		366.2	1098.7	73856.0	3174.0	2075.3		183.1	20936.0	3936951.0
22-06-98	6949	778.3	8686.3	2598.9	11285.2	708.8	1097.9	1806.7	15565.8	6.9			104.2	12855.7	312.7	271.0		27.8	2029.1	370381.7
23-06-98	6775	203.3	6436.5	2588.2	9024.7	752.1	955.3	1707.4	15854.2	20.3		33.9	142.3	13415.1	332.0	298.1	2.0	33.9	2080.0	360446.2
24-06-98	22641	362.3	49810.2	7064.0	56874.2	1426.4	9667.7	11094.1	211240.5	67.9		362.3	1109.4	179769.5	4030.1	3350.9	2.3	226.4	17456.2	8737161.9
29-06-98	37816	2458.1	61262.5	12403.8	73666.3	3819.5	10210.4	14029.9	187947.4	189.1	34.0	567.2	1474.8	167148.4	7790.2	4046.4		226.9	18832.6	4734610.8

Table B.2 North Arm East event load data

		NUTRIENT LOADS (g)							METAL LOADS (g)											
Storm Date	Volume (m^3)	Ammonia (as N)	TKN (as N)	Nitrate + Nitrite (as N)	Total Nitrogen	Filt. React. Phos. (as P)	Particulate P	Phosphorous (Total as P)	Aluminium (Total)	Arsenic (Inorganic)	Cadmium (Total)	Chromium (Total)	Copper (Total)	Iron (Total)	Lead (Total)	Manganese (Total)	Mercury (Total)	Nickel (Total)	Zinc (Total)	TSS (mg/L)
05-07-98	**34038**	1872.1	54120.8	15113.0	69233.8	2689.0	9428.6	12117.6	129345.3	102.1	23.8	408.5	1531.7	103476.2	6773.6	3233.6	3.4	204.2	18789.1	3199594.3
08-07-98	**30268**	3580.0	49230.0	8590.0		2660.0	2660.0	22400.0	83470.0	120.0	20.0	410.0	1140.0	79270.0	3530.0	2310.0	4.0	170.0	14020.0	4861310.0
21-07-98	**22157**	2690.0	36190.0	6360.0		2130.0	2130.0	16400.0	114330.1	88.6	28.8	310.2	1373.7	110785.0	4763.8	3456.5	6.6	177.3	18390.3	5756388.6
25-07-98	**6676**	3738.6	19961.2	3191.1	23152.4	5407.6	2870.7	8278.2	8478.5	20.0	6.0	93.5	420.6	9212.9	447.3	721.0		33.4	4025.6	485345.2
27-07-98	**33576**	51035.5	151763.5	7890.4	159653.9	8931.2	34046.1	42977.3	309906.5	268.6	70.5	1846.7	4734.2	330723.6	11483.0	11449.4	13.4	503.6	41634.2	4244006.4
27-07-98	**58648**																			
28-07-98	**13624**	2588.6	42643.1	3119.9	45763.0	1907.4	7166.2	9073.6	97002.9	95.4	17.7	476.8	1294.3	84332.6	2615.8	2098.1	4.1	122.6	9836.5	4587200.8
28-07-98	**28902**	3468.2	64740.5	8699.5	73440.0	4393.1	8526.1	12919.2	80636.6	144.5	11.6	520.2	1300.6	67052.6	2774.6	1849.7		115.6	12688.0	3829515.0
29-07-98	**69564**	7652.0	82781.2	25043.0	107824.2	8834.6	8347.7	17182.3	77911.7	208.7	27.8	417.4	1321.7	70955.3	2852.1	2086.9		139.1	19130.1	2712996.0
30-07-98	**10654**	1321.1	26635.0	4996.7	31631.7	1885.8	3867.4	5753.2	46344.9	63.9	7.5	117.2	426.2	43574.9	873.6	1203.9		63.9	4602.5	2130800.0
03-08-98	**29748**	3520.0	48400.0	8450.0		2630.0	2630.0	22020.0	27903.6	89.2	14.9		981.7	18057.0	832.9	595.0	3.0	89.2	7080.0	4757970.0
06-08-98	**7472**	1000.0	13225.4	3721.1	16946.5	980.0	980.0	2734.8	14940.0	20.0	10.0	80.0	230.0	11282.7	523.0	520.0	1.0	40.0	3265.3	857820.0
06-08-98	**26734**	3190.0	47051.8	12966.0	60017.8	2440.0	2440.0	9704.4	17650.0	100.0	20.0	350.0	990.0	110946.1	5988.4	2020.0	4.0	150.0	17109.8	4167670.0
20-08-98	**51249**	1793.7	81998.4	19218.4	101216.8	2306.2	14452.2	16758.4	195258.7	205.0	46.1	666.2	2511.2	171684.2	7277.4	4817.4		410.0	29058.2	9583563.0
21-08-98	**40343**	2097.8	41149.9	12909.8	54059.6	2017.2	6212.8	8230.0	131921.6	121.0	20.2	564.8	1331.3	111346.7	2743.3	1976.8			15854.8	3630870.0
23-08-98	**30088**	1023.0	20459.8	8454.7	28914.6	1474.3	6589.3	8063.6	142316.2	90.3	30.1	962.8	2166.3	134192.5	3520.3	2918.5		210.6	16488.2	5385752.0

SUMMARY STATISTICS

Mean		**10120.0**	**87719.6**	**14028.3**	**103593.7**	**6077.7**	**12967.8**	**21997.2**	**184107.1**	**230.3**	**50.5**	**810.2**	**2426.4**	**167828.4**	**8048.4**	**4346.3**	**7.8**	**308.5**	**24141.3**	**10444908**
Maximum		**324937.7**	**1592569.3**	**185655.3**	**1778224.6**	**211203.7**	**213142.2**	**424346.0**	**2979282.3**	**3208.5**	**577.9**	**14004.0**	**44387.1**	**2519447.3**	**100738.5**	**59454.2**	**92.2**	**4413.9**	**307908.5**	**141320038**
Minimum		**143.052**	**3070**	**273.15**	**273.15**	**220.65**	**0**	**529.56**	**1857.42**	**1.821**	**1.1768**	**10.926**	**36.42**	**2330.88**	**92.871**	**128.716**	**0.2942**	**8.826**	**684.696**	**107803**
Median		**3329.1**	**46766.6**	**8644.8**	**54059.6**	**2411.5**	**6456.6**	**10850.7**	**79282.5**	**108.9**	**23.5**	**409.2**	**1126.9**	**76767.7**	**3347.1**	**2047.6**	**3.9**	**160.3**	**14482.2**	**4454332**
Standard Deviation		**39143.4**	**199920.2**	**23263.4**	**234934.2**	**25179.4**	**28054.8**	**53595.8**	**393766.4**	**454.0**	**90.2**	**1819.8**	**5634.0**	**343427.4**	**15403.1**	**8318.4**	**14.3**	**591.7**	**42930.9**	**21074256**
Count		**68**	**68**	**68**	**59**	**68**	**68**	**68**	**68**	**68**	**62**	**62**	**68**	**68**	**68**	**68**	**47**	**67**	**68**	**68**
C_v (=σ/x)		**3.9**	**2.3**	**1.7**	**2.3**	**4.1**	**2.2**	**2.4**	**2.1**	**2.0**	**1.8**	**2.2**	**2.3**	**2.0**	**1.9**	**1.9**	**1.8**	**1.9**	**1.8**	**2**

Table B.3 South Road Connector event mean concentraton (EMC) data

		NUTRIENT CONCENTRATIONS (mg/L)								METAL CONCENTRATIONS (mg/L)													
Storm Date	Volume (m^3)	Ammonia (as N)	TKN (as N)	Nitrate + Nitrite (as N)	Total N	Filt. React. Phos. (as P)	Phosphorous (Total as P)	Particulate P		Aluminium (Total)	Arsenic (Inorganic)	Cadmium (Total)	Chromium (Total)	Copper (Total)	Iron (Total)	Lead (Total)	Manganese (Total)	Mercury (Total)	Nickel (Total)	Zinc (Total)	TSS (mg/L)	EC (μS/cm)	TDS (mg/L)
28-05-97	31743	0.096	0.99	0.03	1.02	0.143	0.254	0.111		0.348	0.006	< 0.0002	< 0.005	< 0.005	0.509	0.006	0.015	< 0.0001	0.002	0.068			
13-06-97	70215.96	0.244	2.45	0.054	2.50	0.045	0.171	0.126		0.328	0.004	0.0002	0.012	0.008	0.528	0.019	0.017		0.008	0.087	9		
9/07/97 (a)		0.12	21.4	< 0.005	21.4	0.0147	0.36	0.3453		0.132	0.001	0.0007	< 0.005	< 0.005	1.24	0.005	0.099	< 0.0001	0.006	0.069			
9/07/97 (b)		0.028	1.42	< 0.005	1.42	0.215	0.289	0.074		0.047	0.007	0.0002	< 0.005	< 0.005	0.106	< 0.001	0.012	< 0.0001	0.004	0.1			
06-08-97	53006	0.5	11.4	< 0.005	11.4	0.024	0.445	0.421		0.104	0.005	0.0064	< 0.005	0.007	0.448	0.002	0.066	0.0001	0.007	0.206	7.5		
07-08-97	68313	0.1545	1.4	0.1945	1.59	0.312	0.4565	0.1445		0.7235	0.0065	0.0002	<0.005	0.0125	0.5695	0.0145	0.0225	< 0.0001	0.0025	0.152	26.7		
08-08-97	58819	0.285	0.975	0.254	1.23	0.1435	0.2325	0.089		0.8925	0.005	0.0002	< 0.005	0.0115	0.63	0.016	0.0215	0.0001	0.002	0.1235	35.1		
10-08-97	135544																				9.5		
30-08-97	85985																						
01-09-98	85675																						
04-09-97	24214																						
06-09-97	87483	0.203	1.00	0.246	1.25	0.339	0.409	0.070		0.283	0.006	< 0.0002	0.007	0.011	0.359	0.008	0.028	0.0001	0.004	0.105	7.45		
13-09-97	88163	0.09	0.95	0.051	1.00	0.27	0.351	0.081		0.185	0.005	< 0.0002	< 0.005	0.01	0.229	0.002	0.019	0.0001	0.002	0.057	21.3		
15-09-97	21894	0.11	1.02	0.036	1.06	0.174	0.276	0.102		0.143	0.003	0.0003	< 0.005	0.015	0.252	0.005	0.03	< 0.0001	0.008	0.099			
18-09-97	113419	0.065	0.98	0.076	1.06	0.23	0.336	0.106		0.443	0.004	0.0002	< 0.005	0.017	0.549	0.013	0.027	< 0.0001	0.002	0.093	19.5		
30-10-97	734972	0.248	1.38	0.132	1.51	0.276	0.415	0.139		2.62	0.005	0.0003	0.011	0.026	1.77	0.042	0.046	0.0001	0.003	0.164	99		
14-11-97	60970																						
08-12-97	60667	0.26	1.99	0.016	2.01	0.345	0.485	0.14		0.314	0.01	< 0.0002	< 0.005	0.011	0.34	< 0.001	0.055	0.0001	0.003	0.05	9.23		
18-12-97	98556																						
12-04-98	151694	0.37	2.505	< 0.005	2.51	0.005	0.266	0.261		0.7845	0.0045	0.00045	< 0.005	0.009	0.7	0.0165	0.097	0.0001	0.0045	0.14	33.8		
19-04-98	279837	0.135	1.24	0.172	1.41	0.014	0.212	0.198		1.36	0.003	0.0003	0.005	0.012	1.39	0.03	0.068	0.0001	0.01	0.14	56.2		
21-04-98	20387																						
24-04-98	81058																						
19-05-98	48195	0.066	1.47	0.014	1.48	0.019	0.240	0.221		0.351	0.002	< 0.0002	< 0.005	0.005	1.20	0.011	0.139	< 0.0001	0.003	0.059	22.8		
22-05-98	47635	0.17	1.31	0.102	1.41	0.025	0.212	0.187		0.614	0.004	< 0.0002	< 0.005	0.009	1.28	0.015	0.132	< 0.0001	0.003	0.086	31.3		
05-06-98	114346	0.226	1.10	0.173	1.28	0.047	0.216	0.169		1.18	0.003	0.0003	0.006	0.013	1.26	0.035	0.095	0.0002	0.002	0.143	46.5		
11-06-98	109209	0.259	1.14	0.287	1.43	0.054	0.186	0.132		0.959	0.005	< 0.0002	0.005	0.011	1.14	0.027	0.074	0.0001	0.003	0.134	60.5	730	1320
21-06-98	84825	0.318	1.1	0.348	1.45	0.034	0.107	0.073		0.331	0.004	< 0.0002	0.007	0.012	0.534	< 0.001	0.083	< 0.0001	0.003	0.131	21	1500	2690
24-06-98	27150	0.286	1.05	0.357	1.41	0.036	0.101	0.065		0.322	0.004	< 0.0002	< 0.005	0.009	0.488	< 0.001	0.096	< 0.0001	0.002	0.133	13.2	1700	3070
29-06-98	35750	0.2	1.05	0.341	1.39	0.037	0.097	0.06		0.32	0.002	< 0.0002	0.005	0.007	0.378	< 0.001	0.088	< 0.0001	0.002	0.114	7.6	1800	3190
05-07-98	34891	0.311	1.23	0.323	1.56	0.032	0.125	0.092		0.630	0.002	< 0.0002	0.009	0.014	0.675	< 0.001	0.128	< 0.0001	0.004	0.115	14.9		
08-07-98	26331	0.39	1.52	0.365	1.89	0.026	0.125	0.099		0.456	0.003	< 0.0002	< 0.005	< 0.005	0.596	< 0.001	0.121	< 0.0001	0.012	0.106	14.2	2000	3680
21-07-98	11075									0.257	0.002	< 0.0002	0.017	0.033	0.327	< 0.001	0.118	0.0003	0.003	0.138	11.2	1900	3500
27-07-98	21764	0.21	1.44	0.333	1.77	0.005	0.129	0.124		0.93	0.003	< 0.0002	0.015	0.031	1.27	< 0.001	0.213	< 0.0001	0.003	0.167	56.9	2000	3540
27-07-98	49104	0.29	1.39	0.371	1.76	0.03	0.17	0.14		1.29	0.004	< 0.0002	0.015	0.032	1.61	0.005	0.128	< 0.0001	0.003	0.196	55.3	1400	2450
28-07-98	10249	0.034	1.2	0.247	1.45	0.034	0.211	0.177		2.09	0.004	0.0002	0.019	0.047	1.93	0.047	0.071	-	0.003	0.239	77	210	391
28-07-98	54102	0.06	1.04	0.288	1.33	0.052	0.203	0.151		1.87	0.005	0.0002	0.017	0.04	1.72	0.039	0.05	< 0.0001	0.003	0.231	65.8	220	403
29-07-98	21898	0.144	1.07	0.29	1.36	0.07	0.204	0.134		1.57	0.005	< 0.0002	< 0.005	0.018	1.57	0.035	0.038	< 0.0001	0.003	0.139	52.4	300	539
30-07-98	14395	0.159	1.1	0.293	1.39	0.072	0.21	0.138		1.54	0.004	< 0.0002	0.005	0.017	1.58	0.034	0.039	< 0.0001	0.003	0.145	64.9	420	768
02-08-98	22325																						
06-08-98	31060		1.04	0.355	1.40		0.127	0.127							0.811	0.012				0.109			
20-08-98	61872	0.076	1.11	0.065	1.18	0.03	0.115	0.085		0.93	0.004	< 0.0002	0.006	0.012	0.958	0.006	0.071	< 0.0001	0.01	0.13	23	1800	3330
21-08-98	18461	0.216	1.1	0.248	1.35	0.048	0.163	0.115		2.1	0.003	< 0.0002	0.01	0.02	2.06	0.019	0.069	< 0.0001	< 0.001	0.214	39	850	1550
23-08-98	12184	0.041	0.81	0.173	0.98	0.018	0.108	0.09		1.3	0.003	< 0.0002	0.007	0.014	1.22	0.01	0.066	< 0.0001	< 0.001	0.09	24	870	1570

SUMMARY STATISTICS

Mean		0.193	2.158	0.208	2.342	0.098	0.235	0.141		0.816	0.004	0.0007	0.010	0.016	0.921	0.018	0.072	0.000	0.004	0.128	33.41	1180	2133
Maximum		0.5	21.4	0.371	21.4	0.345	0.485	0.421		2.62	0.010	0.0064	0.019	0.047	2.060	0.047	0.213	0.000	0.012	0.239	99.00	2000	3680
Minimum		0.028	0.81	< 0.005	0.983	0.005	0.097	0.06		0.047	0.001	< 0.0002	< 0.005	< 0.005	0.106	< 0.001	0.012	< 0.0001	< 0.005	0.05	7.45	210	391
Median		0.2	1.125	0.246	1.412	0.045	0.212	0.126		0.622	0.004	0.0003	0.008	0.012	0.700	0.015	0.069	0.0001	0.003	0.13	24.00	1400	2450
Standard Deviation		0.114	3.837	0.123	3.790	0.107	0.112	0.077		0.667	0.002	0.0016	0.005	0.010	0.549	0.013	0.046	0.000	0.003	0.048	24.03	691	1249
Count		33	34	30	34	33	34	34	0	34	34	14	18	30	35	26	34	11	32	35	31	15	15
C_v (=s/x)		0.592	1.778	0.591	1.619	1.102	0.476	0.548		0.818	0.405	2.254	0.491	0.630	0.596	0.741	0.635	0.508	0.645	0.375	0.719	0.585	0.585

Table B.4 South Road Connector event load data

		NUTRIENT LOADS (g)							METAL LOADS (g)											
Storm Date	Volume (m^3)	Ammonia (as N)	TKN (as N)	Nitrate + Nitrite (as N)	Total N (as N)	Filt. React. Phos. (as P)	Phosphorus (Total as P)	Particulate P (as P)	Aluminium (Total)	Arsenic (Inorganic)	Cadmium (Total)	Chromium (Total)	Copper (Total)	Iron (Total)	Lead (Total)	Manganese (Total)	Mercury (Total)	Nickel (Total)	Zinc (Total)	SEDIMENT (TSS)
28-05-97	31743		31425.451																	
13-06-97	70216	17141.4	172044.6	3767.4	175812.0	3159.7	11996.3	8836.6	23011.5	280.9	14.4	837.3	565.1	37092.2	1334.1	1200.0		563.2	6086.7	631943.6
9/07/97 (a)																				
9/07/97 (b)																				
06-08-97	53006	26503.0	604268.4		604268.4	1272.1	23587.7	22315.5	5512.6	265.0	339.2		371.0	23746.7	106.0	3498.4	5.3	371.0	10919.2	396993.2
07-08-97	68313	10554.4	95638.2	13286.9	108925.1	21313.7	31184.9	9871.2	49424.5	444.0	13.7		853.9	38904.3	990.5	1537.0		170.8	10383.6	1825221.5
08-08-97	58819	16763.4	57348.5	14940.0	72288.6	8440.5	13675.4	5234.9	52496.0	294.1	11.8		676.4	37056.0	941.1	1264.6	5.9	117.6	7264.1	2064509.3
10-08-97	135544	27475.4	187584.7	18416.4		9234.4	35331.5		94738.1	587.6	46.6	1001.9	1669.3	108201.1	2007.8	7567.4	14.4	485.1	16388.3	3899967.5
30-08-97	85985	15957.3	115555.2	13150.9		5014.0	19920.0		59869.5	349.2	29.7	679.3	1137.7	69922.6	1266.9	5184.8	10.1	299.0	10607.0	2376235.6
01-09-98	85675	15888.6	115111.5	13115.8		4989.8	19829.6		59651.7	347.8	29.6	677.2	1134.2	69680.6	1262.3	5169.2	10.1	297.9	10570.4	2366904.6
04-09-97	24214	3514.8	29987.5	5149.3		915.6	4039.6		16681.4	82.1	8.5	230.2	391.2	20733.2	351.5	1809.3	3.7	77.8	3158.7	598104.7
06-09-97	87483	17725.9	87627.4	21504.7	109132.0	29650.7	35796.7	6145.9	24749.2	502.0		570.8	966.6	31446.4	658.2	2489.5	8.7	308.3	9190.0	651748.4
13-09-97	88163	7934.7	83754.9	4496.3	88251.2	23804.0	30945.2	7141.2	16310.2	440.8			881.6	20189.3	176.3	1675.1	8.8	176.3	5025.3	1877871.9
15-09-97	21894	3116.4	26938.0	4779.5	23119.6	799.9	3558.4	2233.1	15069.9	73.1	7.7	211.2	359.4	18823.2	317.5	1664.0	3.4	69.9	2868.7	535977.7
18-09-97	113419	7372.2	111150.6	8619.8	119770.5	26086.4	38108.8	12022.4	50244.6	453.7	22.7		1928.1	62267.0	1474.4	3062.3		226.8	10548.0	2211670.5
30-10-97	734972	182134.7	1013650.5	96959.5	1110610.0	202730.1	304707.9	101977.8	1927845.9	4027.1	220.5	7732.5	18787.6	1298905.6	31029.7	33747.6	73.5	2396.3	120388.9	72762228.0
14-11-97	60970	10585.3	80140.9	10197.3		3161.2	12921.1		42329.8	235.8	21.2	506.5	851.6	50279.3	894.7	3896.6	7.7	207.5	7636.1	1634390.6
08-12-97	60667	15773.4	120727.3	970.7	121698.0	20930.1	29423.5	8493.4	19049.4	606.7			667.3	20626.8		3336.7	6.1	182.0	3033.4	559956.4
18-12-97	98556																			
13-03-98																				
23-03-98	2859.1	274.2	3085.0	1059.8	4037.0	52.1	274.2	566.1	1934.6	7.1	1.0	37.1	64.7	2670.5	40.5	306.7	0.7	8.0	409.8	58442.5
12-04-98	151694	56126.8	379993.5			758.5	40350.6		119003.9	682.6	68.3		1365.2	106185.8	2503.0	14714.3	15.2	682.6	21237.2	5127257.2
19-04-98	279837	37778.0	346997.9	48132.0		3917.7	59325.4		380578.3	839.5	84.0	1399.2	3358.0	388973.4	8395.1	19028.9	28.0	2798.4	39177.2	15726839.4
21-04-98	20387	2862.1	24969.2	4533.9	30225.1	726.9	3252.9	4499.6	14024.5	67.4	7.2	198.7	338.4	17578.8	295.4	1568.3	3.2	64.8	2679.7	495952.9
24-04-98	81058	14871.9	108520.4	12589.1	114453.9	4632.4	18493.7	15157.8	56411.1	326.5	28.1	645.9	1082.5	66074.7	1193.5	4936.7	9.6	280.9	10025.3	2228402.2
19-05-98	48195	3158.0	70788.5	663.9	61525.7	919.4	11556.4	8136.7	16916.5	113.8			241.0	57796.7	512.3	6700.4		135.9	2860.1	1098846.0
22-05-98	47635	8098.0	62401.9	4858.8	67975.1	1190.9	10098.6	6287.8	29247.9	190.5			428.7	60972.8	714.5	6287.8		142.9	4096.6	1490975.5
05-06-98	114346	25858.6	126242.7	19731.3	165573.0	5384.0	24688.9	8347.3	134899.7	395.3	36.9	702.5	1483.4	143840.5	3957.7	10900.5	22.9	281.0	16380.0	5317089.0
11-06-98	109209	28285.1	124498.3	31343.0	153657.1	5897.3	20312.9	7098.6	104731.4	546.0		546.0	1201.3	124498.3	2948.6	8081.5	10.9	327.6	14634.0	6607144.5
21-06-98	84825	26974.4	93307.5	29519.1	117991.6	2884.1	9076.3	5089.5	28077.1	339.3		593.8	1017.9	45296.6		7040.5		254.5	11112.1	1781325.0
24-06-98	27150	7764.9	28507.5	9692.6	42277.1	977.4	2742.2	2506.1	8742.3	108.6			244.4	13249.2		2606.4		54.3	3611.0	358380.0
29-06-98	35750	7150.0	37537.5	12190.8	67388.8	1322.8	3467.8	3539.3	11440.0	71.5		178.8	250.3	13513.5		3146.0		71.5	4075.5	271700.0
05-07-98	34891	10844.8	43072.7	11258.4		1127.9	4348.6		21986.2	78.3		301.2	489.2	23536.4		4458.9		135.3	4021.7	519875.9
08-07-98	26331	10269.1	40023.1	9610.8	46684.9	684.6	3291.4	3265.0	12006.9	79.0				15693.3		3186.1		316.0	2791.1	373900.2
21-07-98	11075	1381.3	13040.7	2886.6	19503.1	320.5	1508.7	1550.5	2846.3	22.2		188.3	365.5	3621.5		1306.9	3.3	33.2	1528.4	124040.0
27-07-98	21764	4570.4	31340.2	7247.4	31492.5	108.8	2807.6	3852.2	20240.5	65.3		326.5	674.7	27640.3		4635.7		65.3	3634.6	1238371.6
27-07-98	49104	14240.2	68254.6	18217.6	65210.1	1473.1	8347.7	7414.7	63344.2	196.4		736.6	1571.3	79057.4	245.5	6285.3		147.3	9624.4	2715451.2
28-07-98	10249	348.5	12298.8	2531.5	13938.6	348.5	2162.5	1373.4	21420.4	41.0	2.0	194.7	481.7	19780.6	481.7	727.7		30.7	2449.5	789173.0
28-07-98	54102	3246.1	56266.1	15581.4	75364.1	2813.3	10982.7	7466.1	101170.7	270.5	10.8	919.7	2164.1	93055.4	2110.0	2705.1		162.3	12497.6	3559911.6
29-07-98	21898	3153.3	23430.9	6350.4		1532.9	4467.2		34379.9	109.5			394.2	34379.9	766.4	832.1		65.7	3043.8	1147455.2
30-07-98	14395	2288.8	15834.5	4217.7	20081.0	1036.4	3023.0	1828.2	22168.3	57.6		72.0	244.7	22744.1	489.4	561.4		43.2	2087.3	934235.5
02-08-98	22325	3189.9	27503.4	4849.0	26231.9	821.1	3646.9	1897.6	15369.4	74.8	7.9	214.7	365.3	19178.9	323.8	1691.2	3.5	71.3	2922.7	547485.7
06-08-98	31060	4731.4	32302.4	11026.3	41868.9	1278.8	3944.6	3571.9	21442.3	109.1	10.9	284.7	482.5	25189.7	372.7	2225.0	4.5	101.3	3385.5	784312.0
20-08-98	61872	4702.3	68677.9	4021.7	60820.2	1856.2	7115.3	5568.5	57541.0	247.5		371.2	742.5	59273.4	371.2	4392.9		618.7	8043.4	1423056.0
21-08-98	18461	3987.6	20307.1	4578.3		886.1	3009.1		38768.1	55.4		184.6	369.2	38029.7	350.8	1273.8			3950.7	719979.0
23-08-98	12184	499.5	9869.0	2107.8		219.3	1315.9		15839.2	36.6		85.3	170.6	14864.5	121.8	804.1			1096.6	292416.0

SUMMARY STATISTICS

Mean		16221.9	114333.9	13029.6	125339.2	9870.1	21430.2	9443.0	92963.8	344.4	46.5	711.3	1270.8	83526.1	2091.1	4817.2	11.8	329.3	10376.7	3661603.4
Maximum		182134.7	1013650.5	96959.5	1110610.0	202730.1	304707.9	101977.8	1927845.9	4027.1	339.2	7732.5	18787.6	1298905.6	31029.7	33747.6	73.5	2798.4	120388.9	72762228.0
Minimum		274.2	3085.0	663.9	4037.0	52.1	274.2	566.1	1934.6	7.1	1.0	37.1	64.7	2670.5	40.5	306.7	0.7	8.0	409.8	58442.5
Median		8098.0	65328.2	9610.8	67681.9	1473.1	10098.6	5857.2	24749.2	235.8	17.8	371.2	671.0	37056.0	714.5	3146.0	8.2	170.8	5025.3	1147455.2
Standard Deviation		28976.0	181253.9	16781.7	214847.0	31816.1	47403.3	18049.6	300294.5	624.7	80.5	1388.5	2914.4	204577.6	5427.3	5995.9	15.3	560.6	18946.6	11377228.6
C_v (=s/$\bar{x}$)		1.8	1.6	1.3	1.7	3.2	2.2	1.9	3.2	1.8	1.7	2.0	2.3	2.4	2.6	1.2	1.3	1.7	1.8	3.1

Appendix C

IFD INFORMATION FOR ENFIELD, SOUTH AUSTRALIA

Contents:

IFD information and chart for ENFIELD, South Australia

Design Intensity Frequency Duration Rainfall

Design rainfalls are calculated given the 9 input variables
Method based upon the procedure outlines in ARR (1987) pp24-25
Spreadsheet based upon work of Daniell (1993)
NB Only works for skew G not equal to 0
Sarah Murphy 19/4/99

Enfield, Corporation of the City of

Step 1 **Enter Input Data (from Maps in ARR Vol 2)**

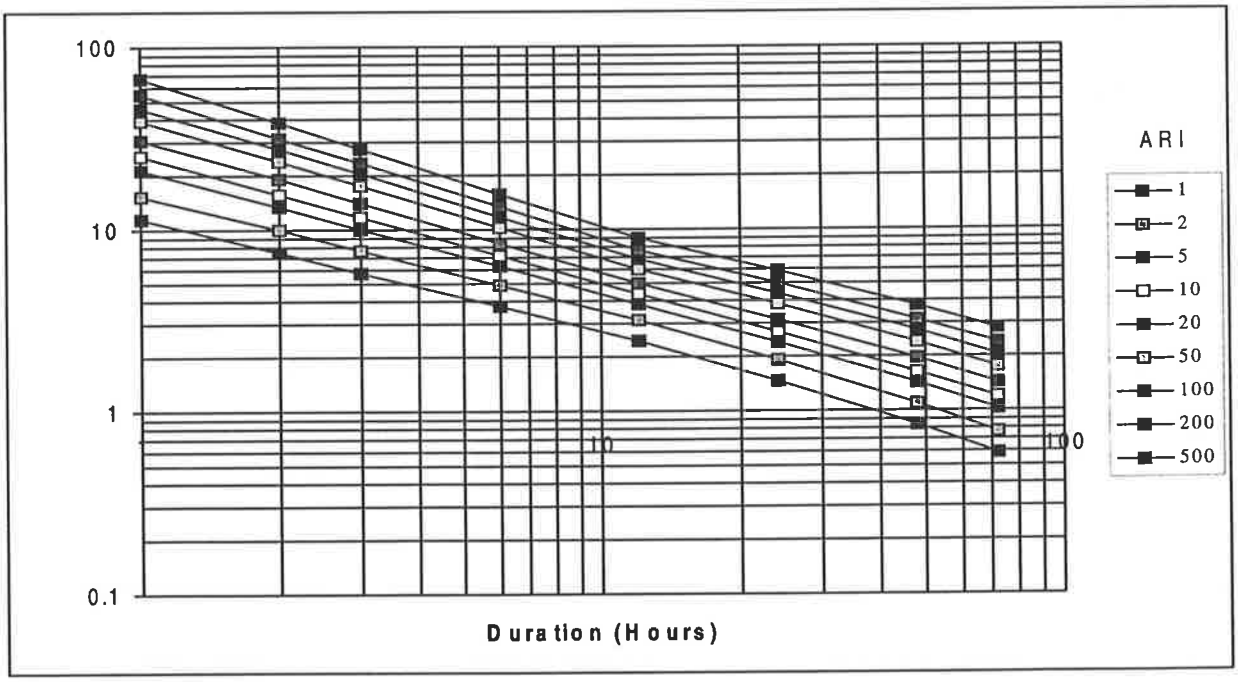
$^{2}i_{1}$	15.9 mm/hr	$^{50}i_{1}$	34.9 mm/hr	F2	4.47
$^{2}i_{12}$	3.2 mm/hr	$^{50}i_{12}$	5.6 mm/hr	F50	15.0
$^{2}i_{72}$	0.80 mm/hr	$^{50}i_{72}$	1.6 mm/hr	G	0.52

Step 2 **Intensities for Durations less than 1 hour**

$^{2}i_{6m}$ 53.9 mm/hr A(3.1) $^{50}i_{6m}$ 126.2 mm/hr A(3.2)

Step 3 **LPIII Design Rainfalls**

	Duration											
ARI	6m	10m	20m	30m	1	2	3	6	12	24	48	72
1	38.2	30.7	21.7	17.3	11.41	7.48	5.81	3.76	2.45	1.45	0.84	0.59
2	51.7	41.5	29.3	23.3	15.3	9.92	7.65	4.89	3.14	1.88	1.10	0.77
5	73.3	58.5	41.0	32.5	21.1	13.28	10.06	6.25	3.89	2.39	1.42	1.02
10	89.2	71.0	49.5	39.1	25.3	15.63	11.73	7.16	4.38	2.72	1.64	1.19
20	110.7	87.9	61.0	48.0	30.9	18.84	14.01	8.42	5.08	3.19	1.95	1.42
50	143.2	113.3	78.3	61.5	39.3	23.53	17.31	10.22	6.05	3.85	2.39	1.75
100	171.5	135.4	93.2	73.0	46.5	27.51	20.09	11.70	6.84	4.40	2.75	2.04
200	203.7	160.5	110.1	86.1	54.6	31.91	23.14	13.32	7.69	4.99	3.15	2.35
500	252.8	198.6	135.8	105.8	66.8	38.46	27.65	15.67	8.92	5.84	3.74	2.80



Appendix D

CONTINUOUS REAL TIME MONITORING DATA SET

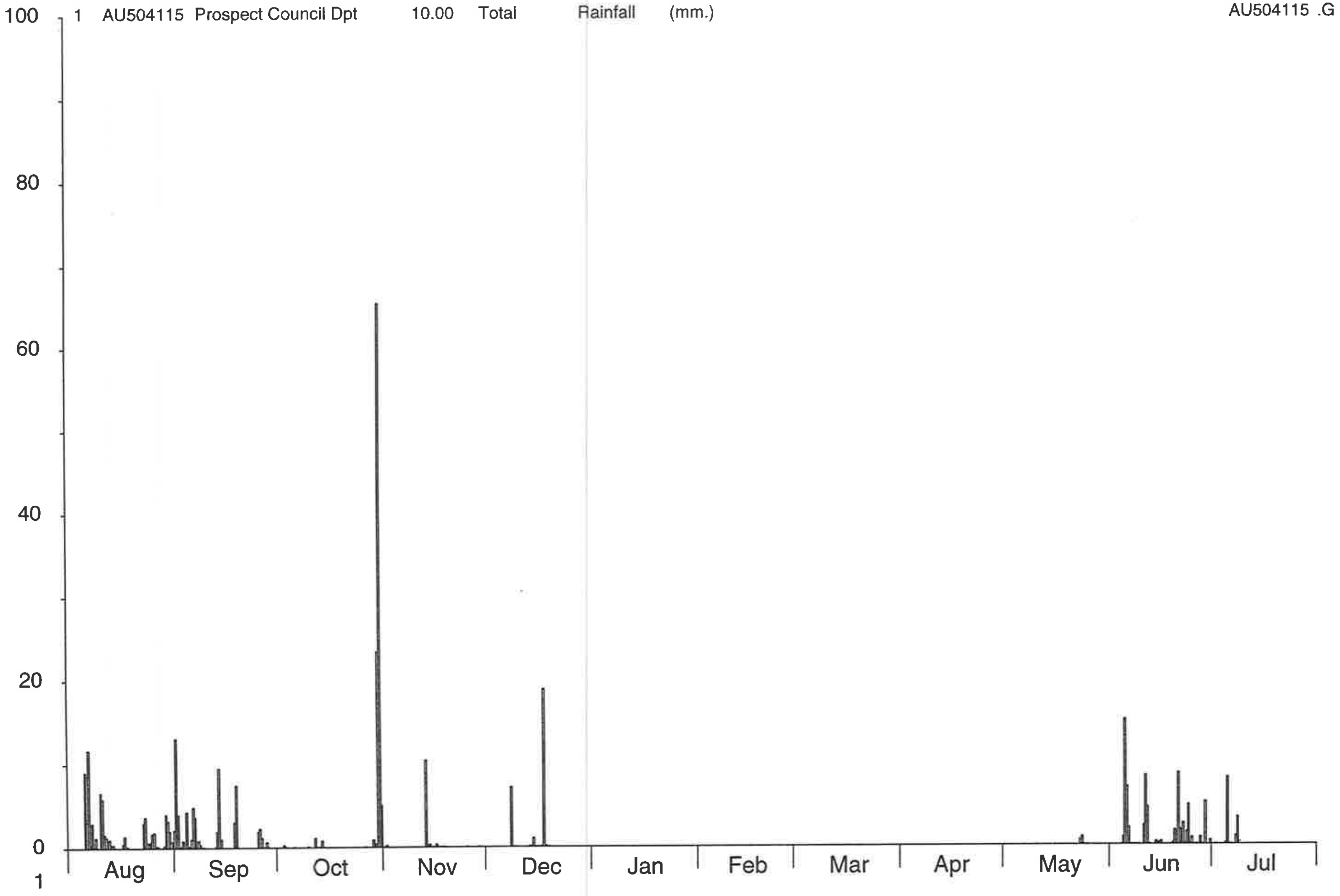
Contents:

University of Adelaide

HYPLOT V89 Output 07/11/1999

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

1997

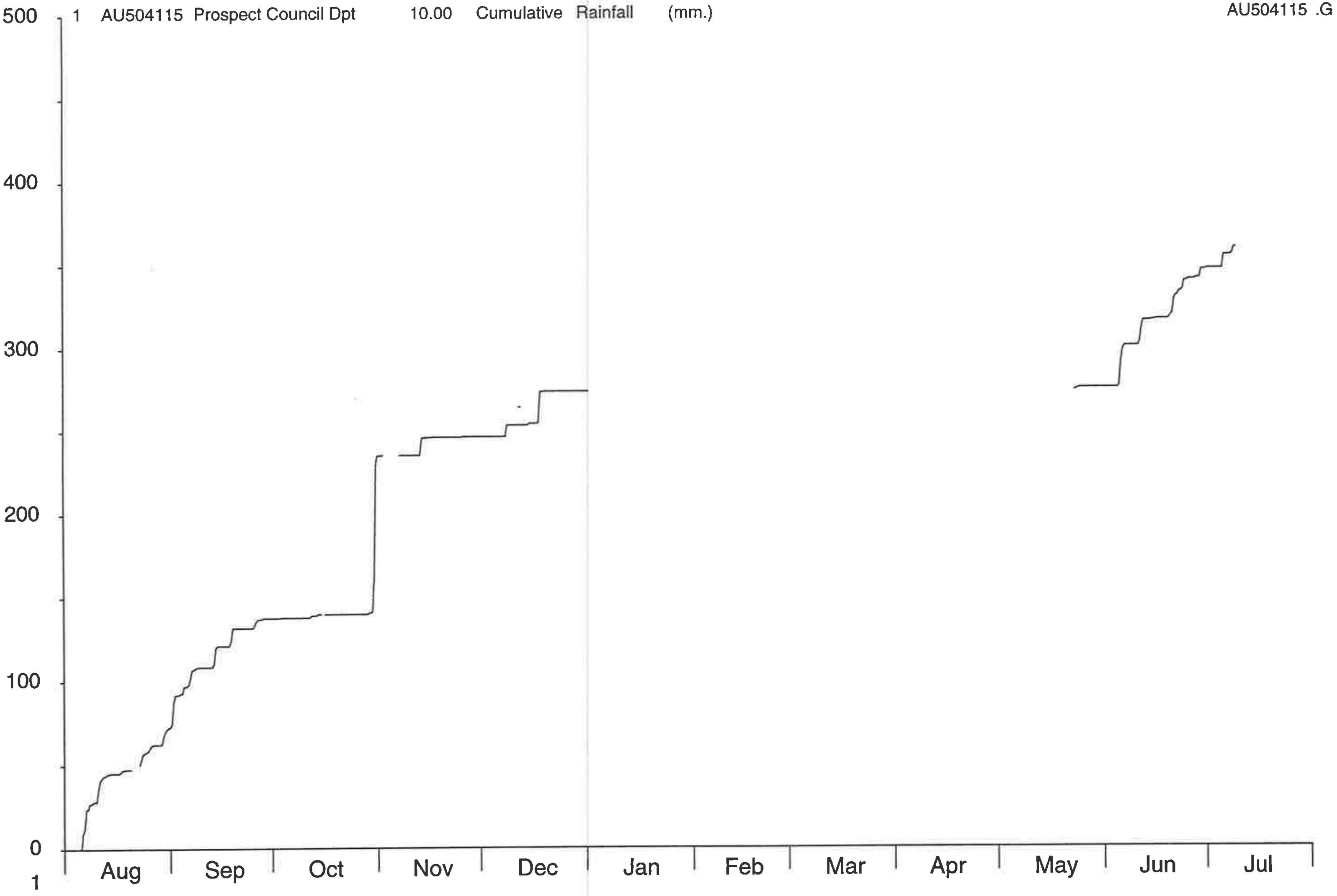


University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period	12 Month	Plot Start	00:00_01/08/1997
Interval	12 Hour	Plot End	00:00_01/08/1998



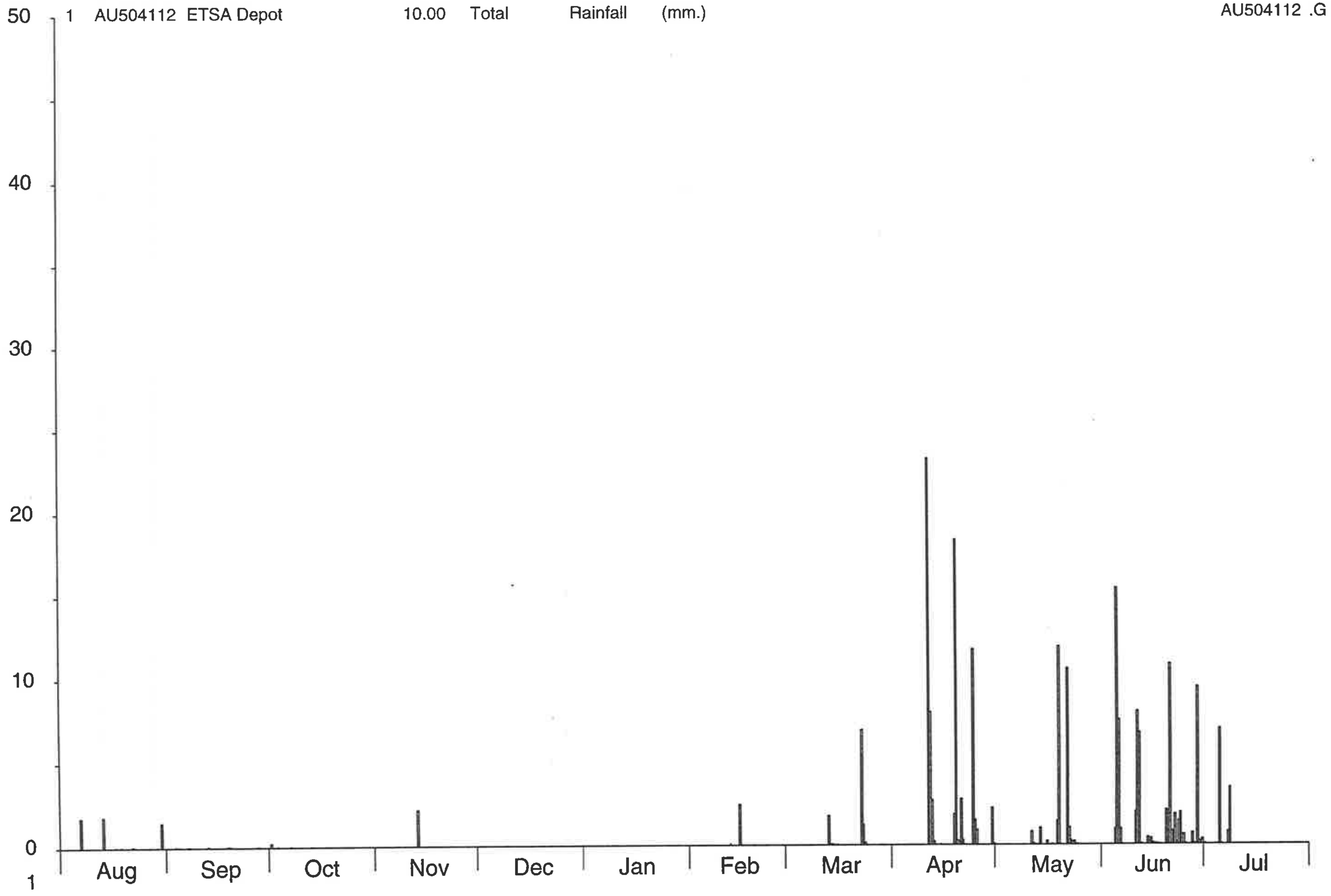
University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

AU504112 .G



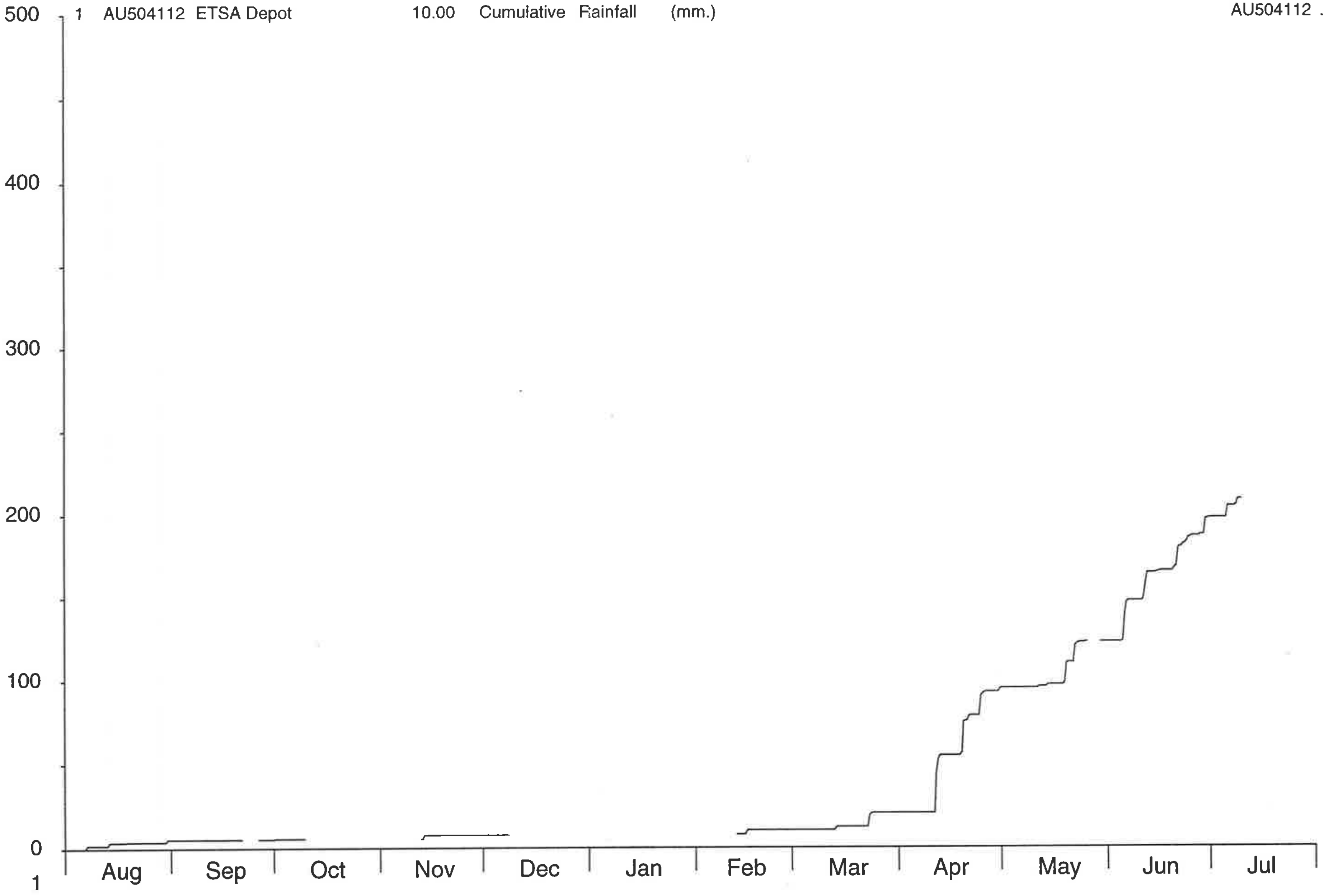
University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

AU504112 .G

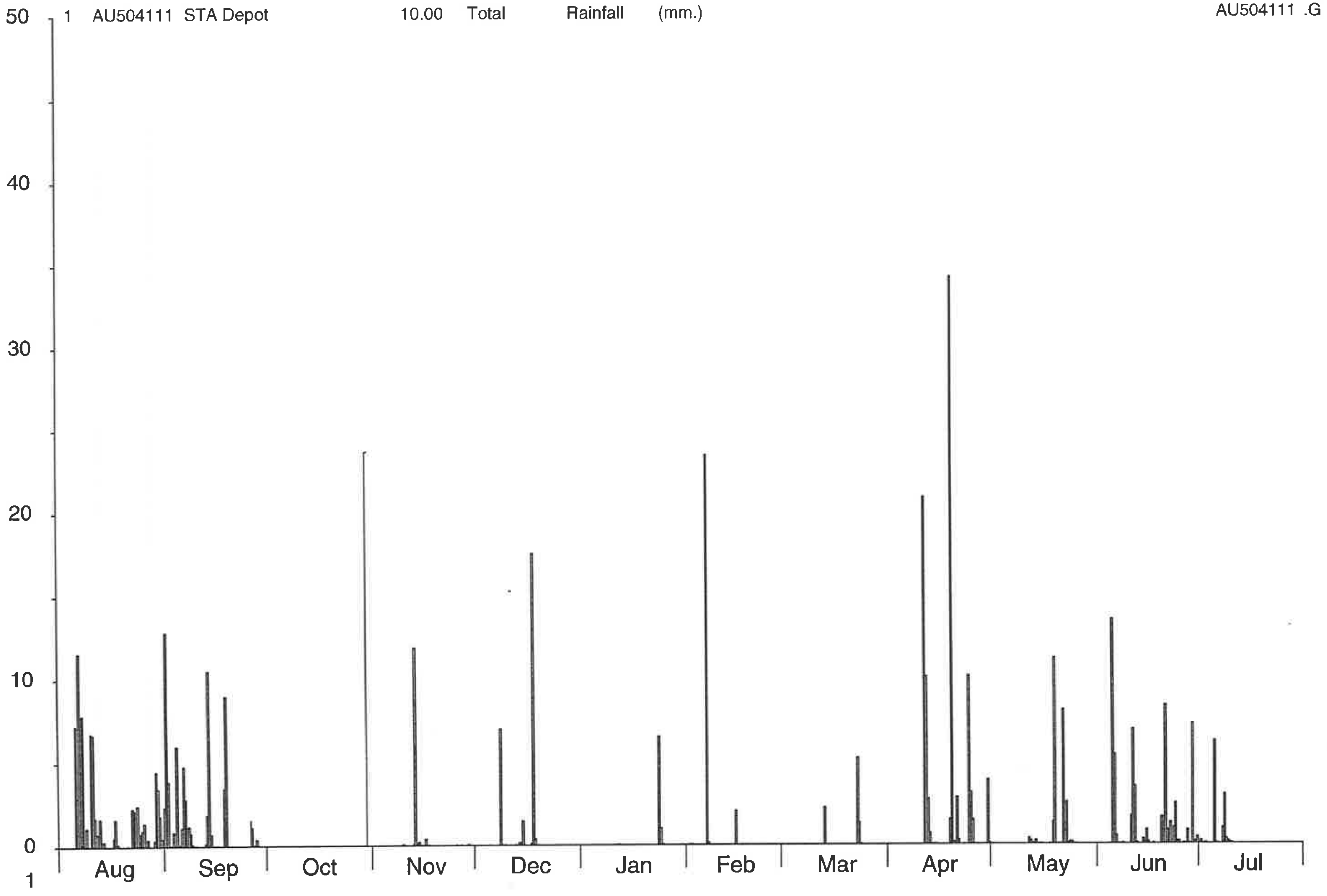


University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

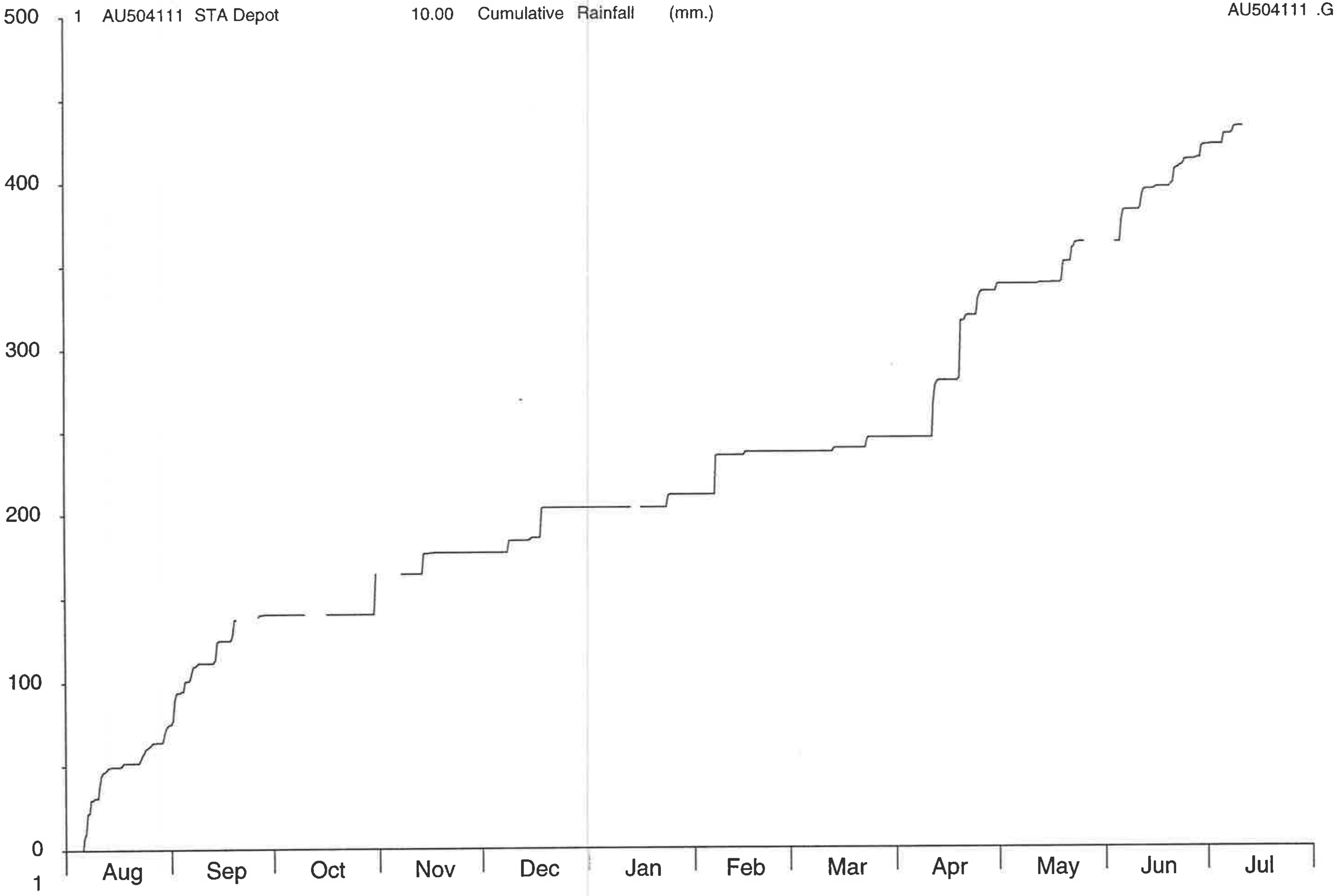


University of Adelaide

HYPLOT V89 Output 07/11/1999

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

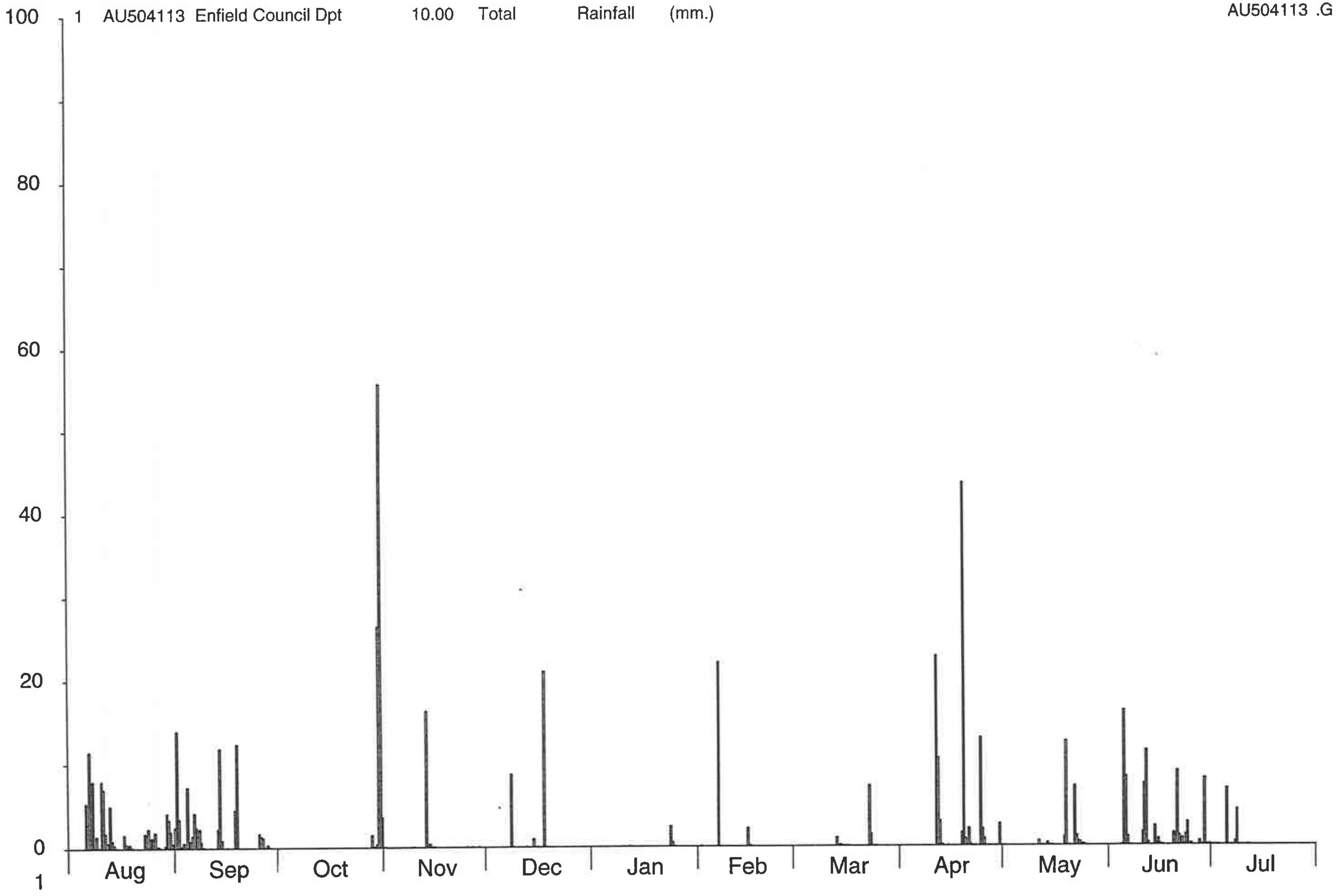
1997



University of Adelaide

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

HYPLOT V89 Output 07/11/1999
1997

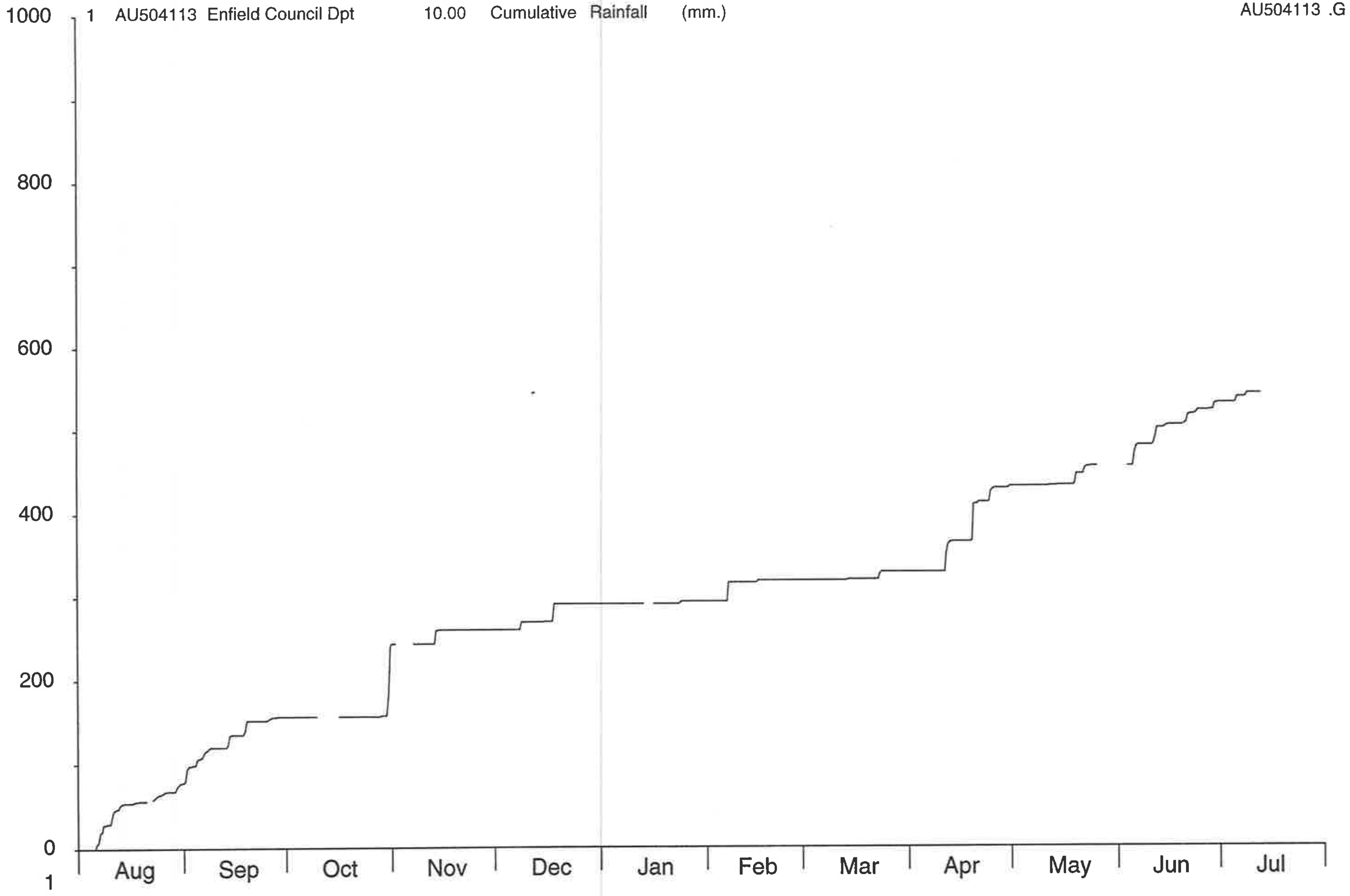


University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998



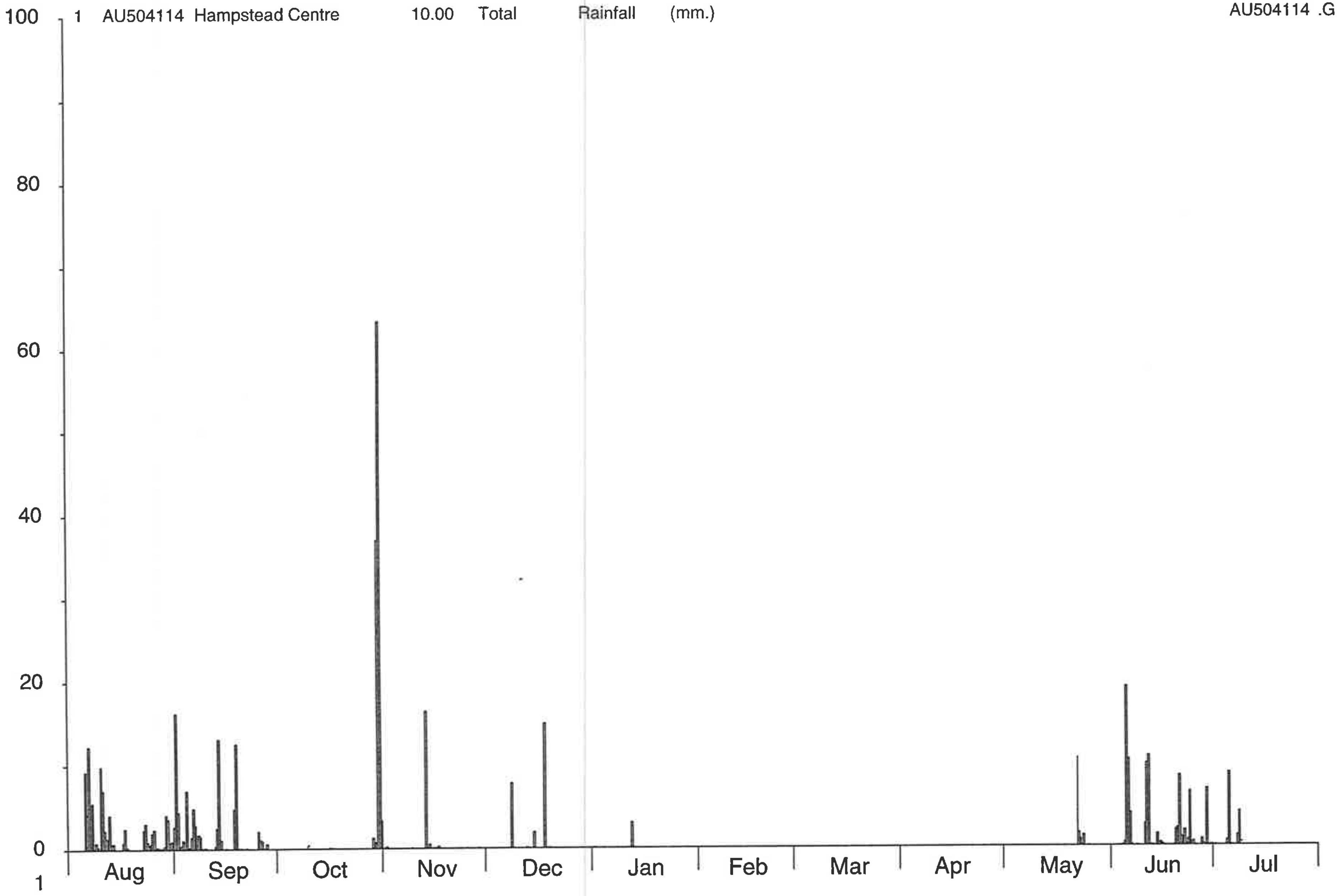
University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

AU504114 .G

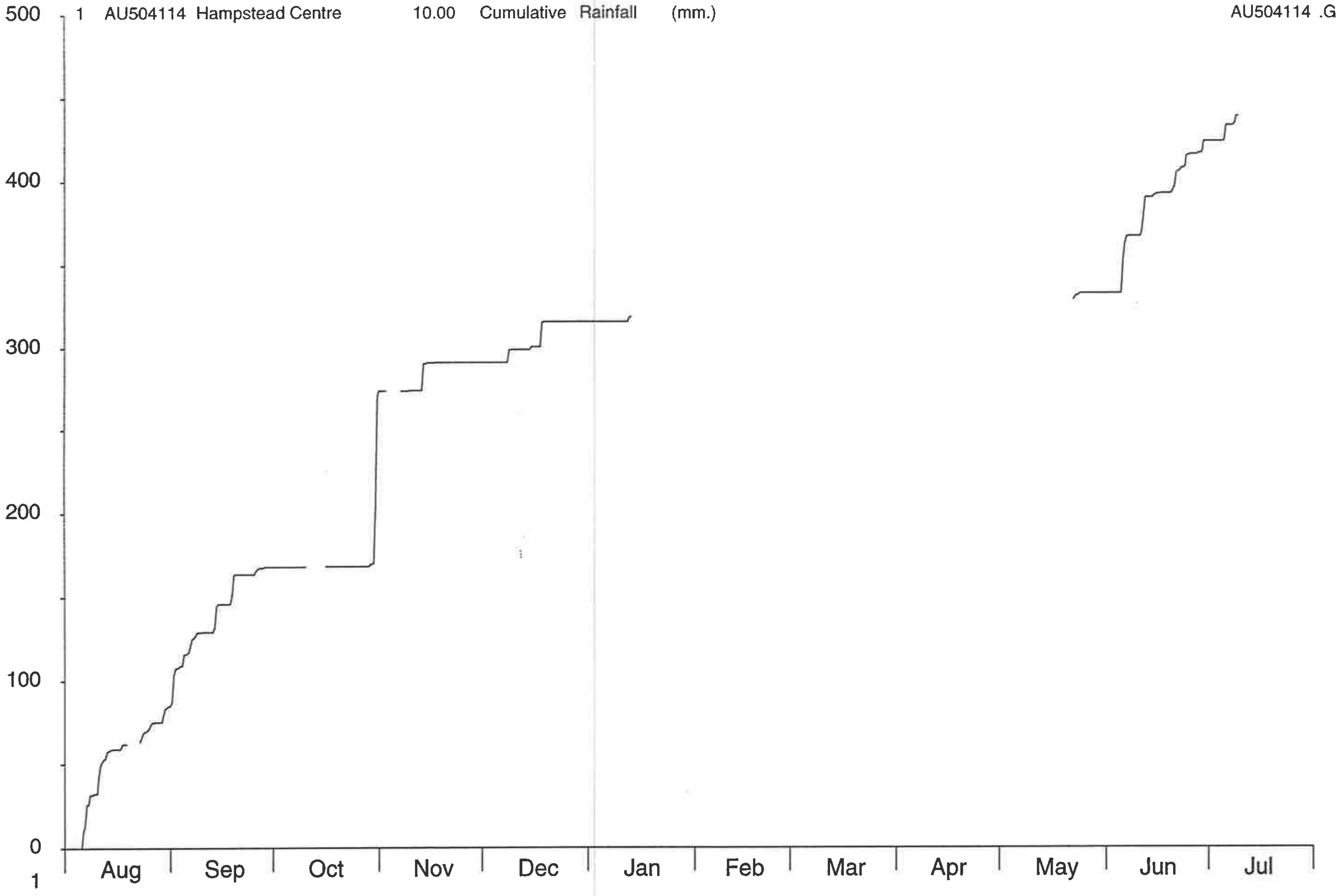


University of Adelaide

HYPLOT V89 Output 07/11/1999

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

1997

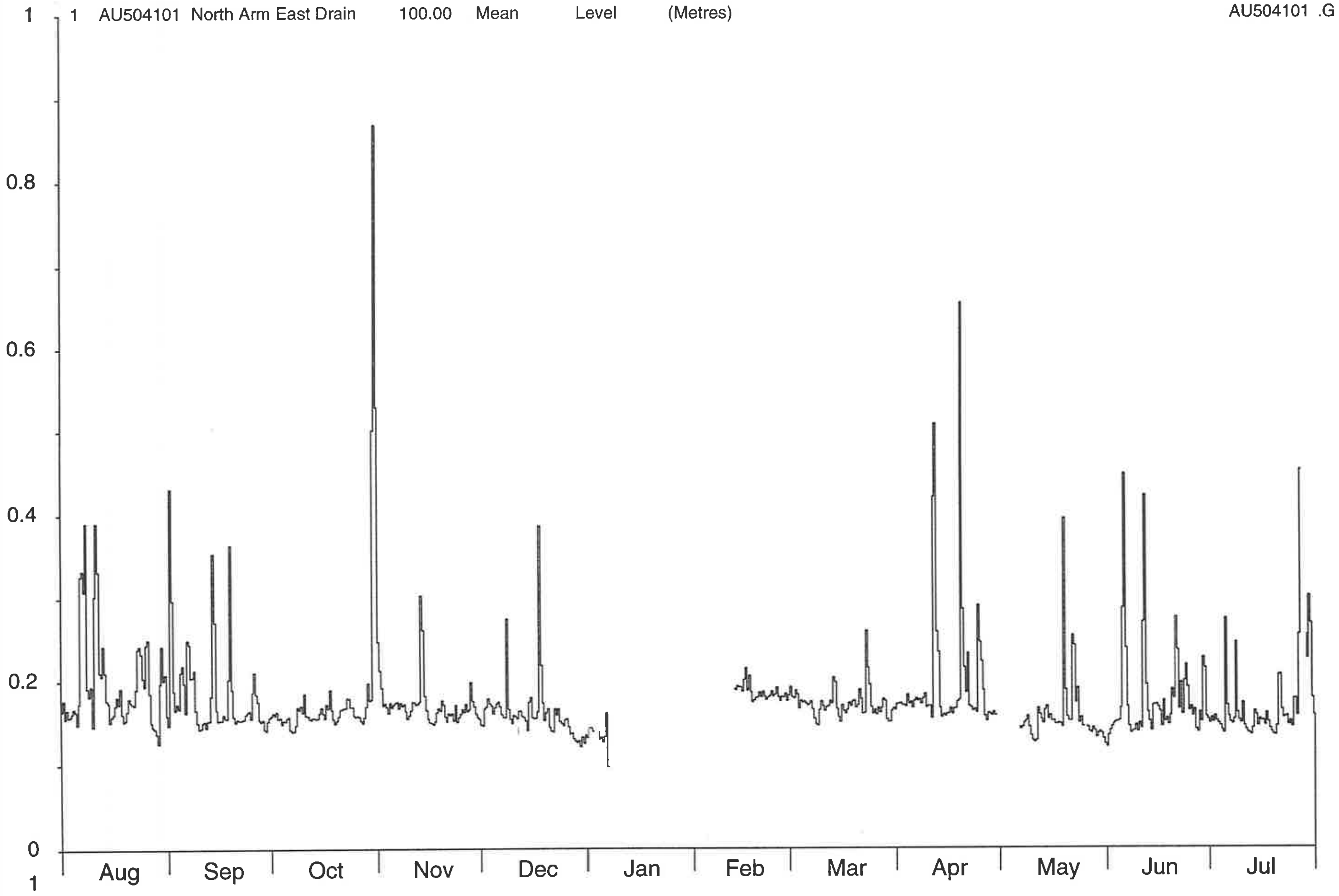


University of Adelaide

HYPLOT V89 Output 07/11/1999

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

1997

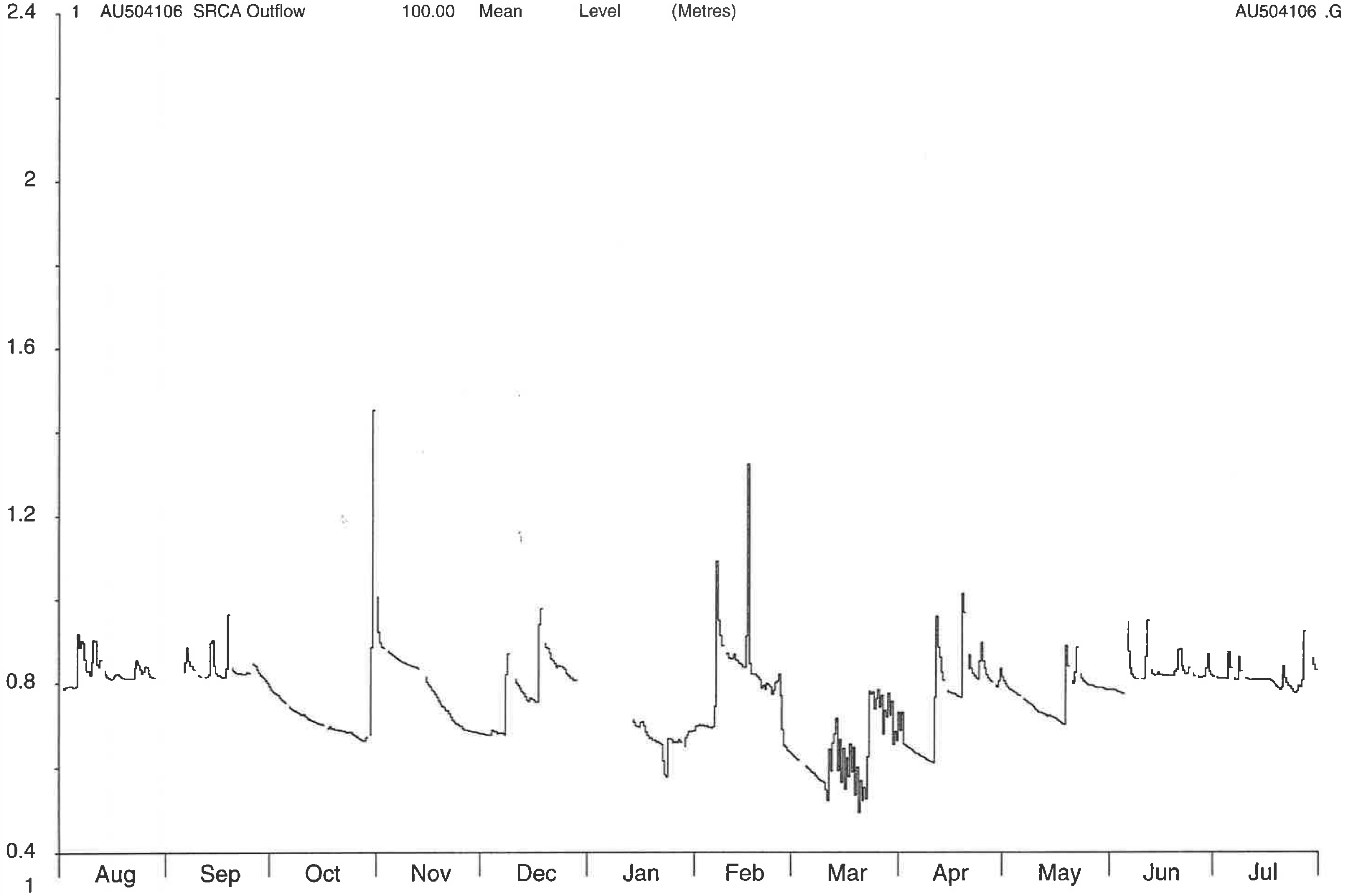


University of Adelaide

HYPLOT V89 Output 07/11/1999

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

1997

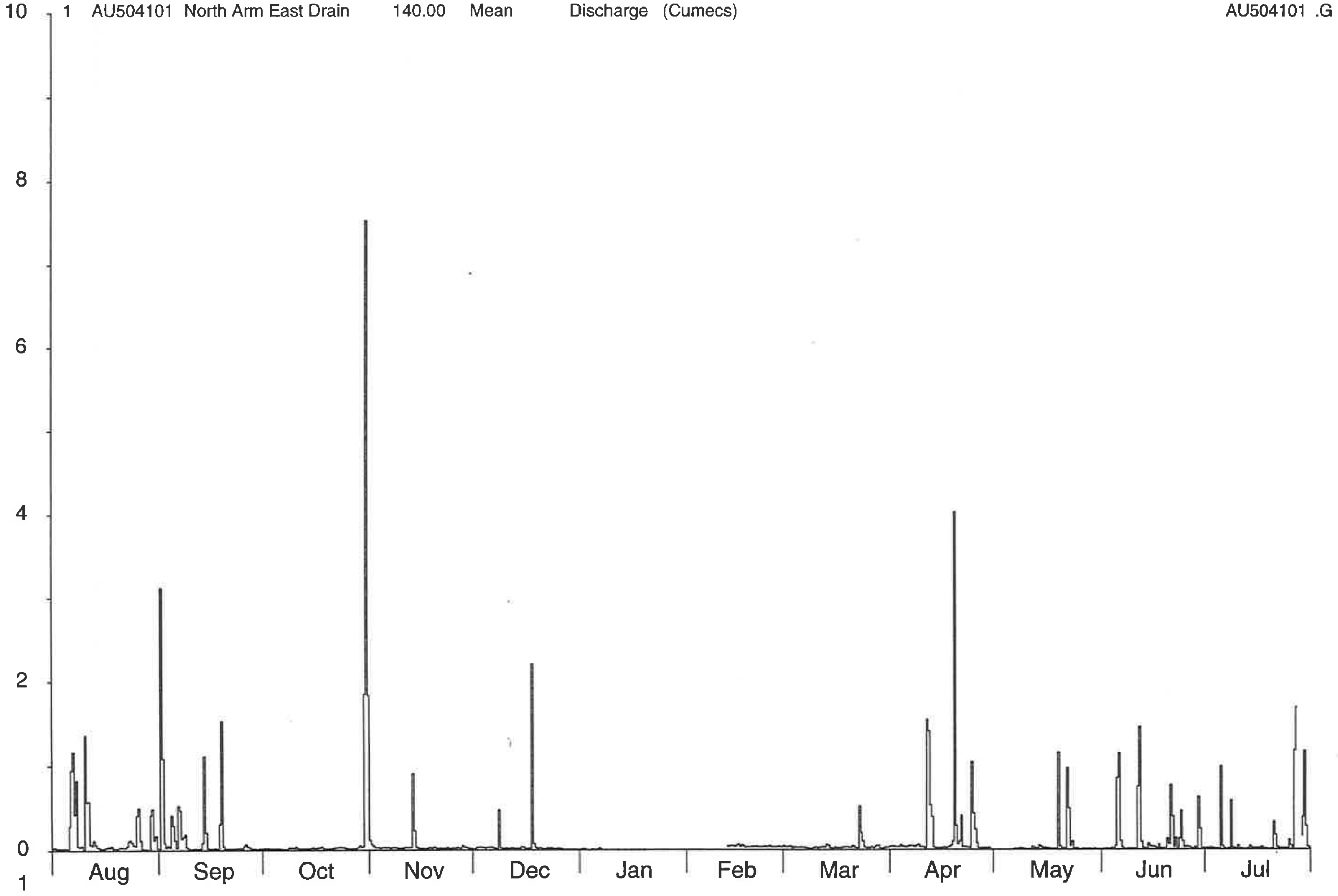


University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

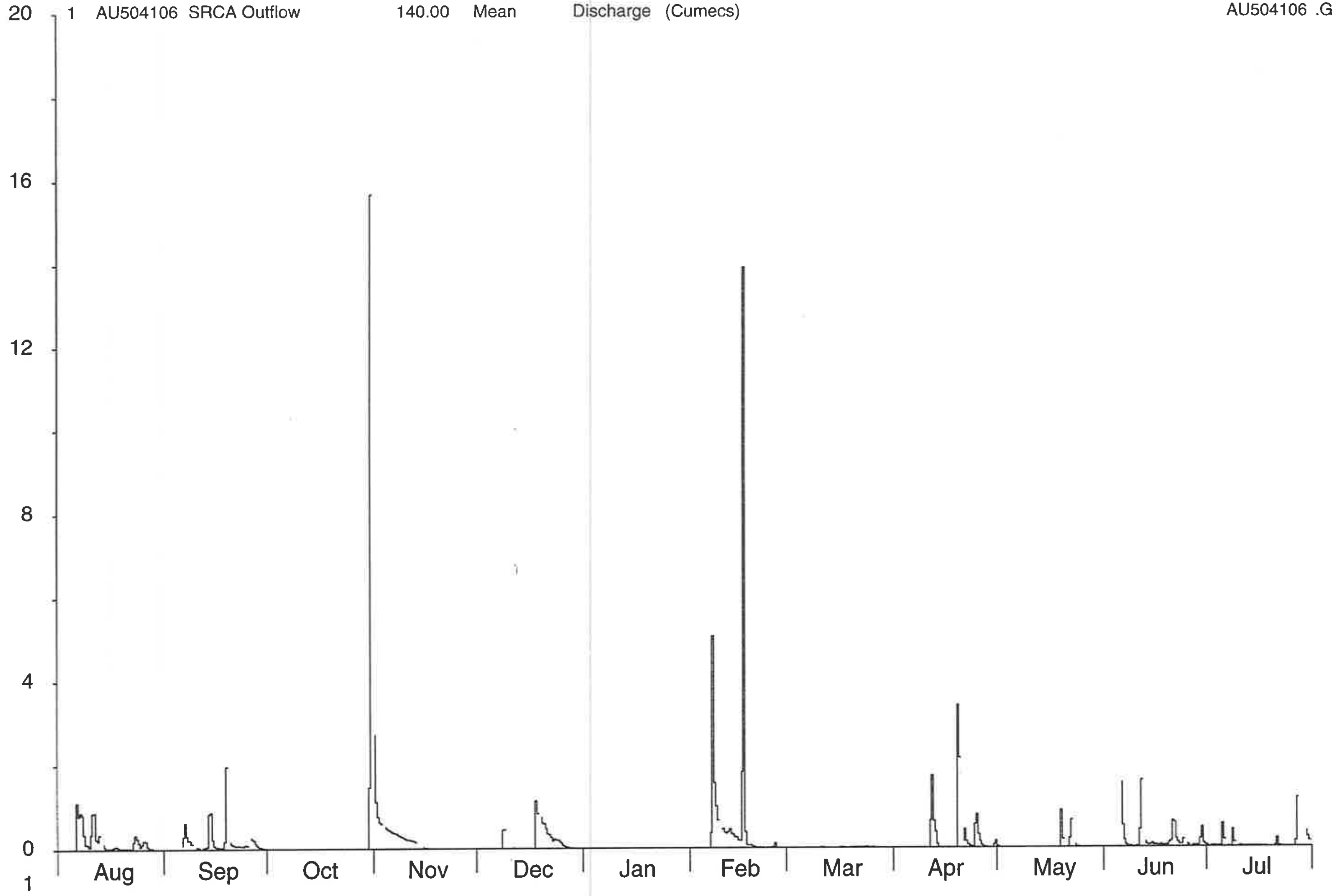


University of Adelaide

HYPLOT V89 Output 07/11/1999

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

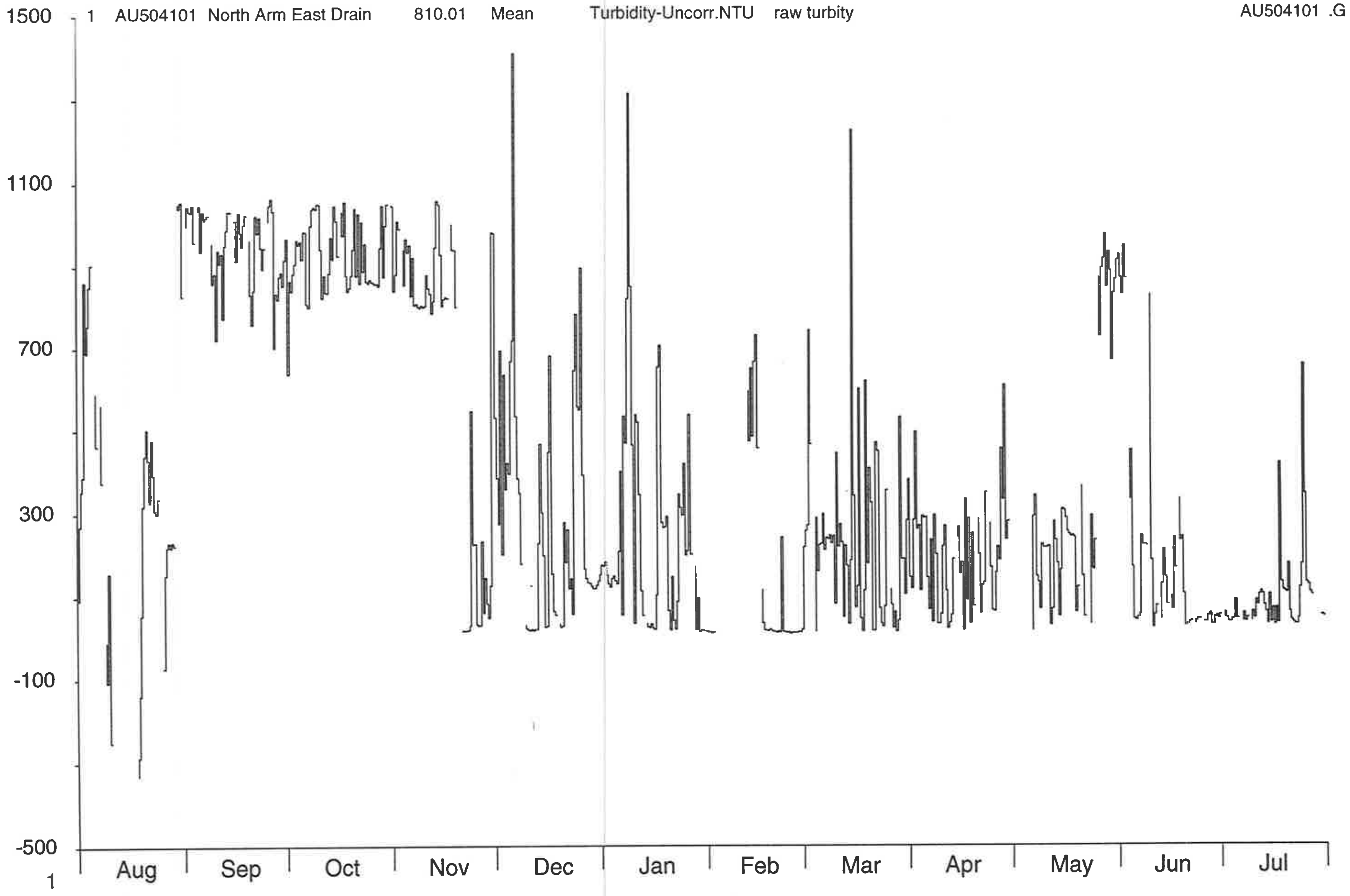
1997



University of Adelaide

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

HYPLOT V89 Output 07/11/1999
1997

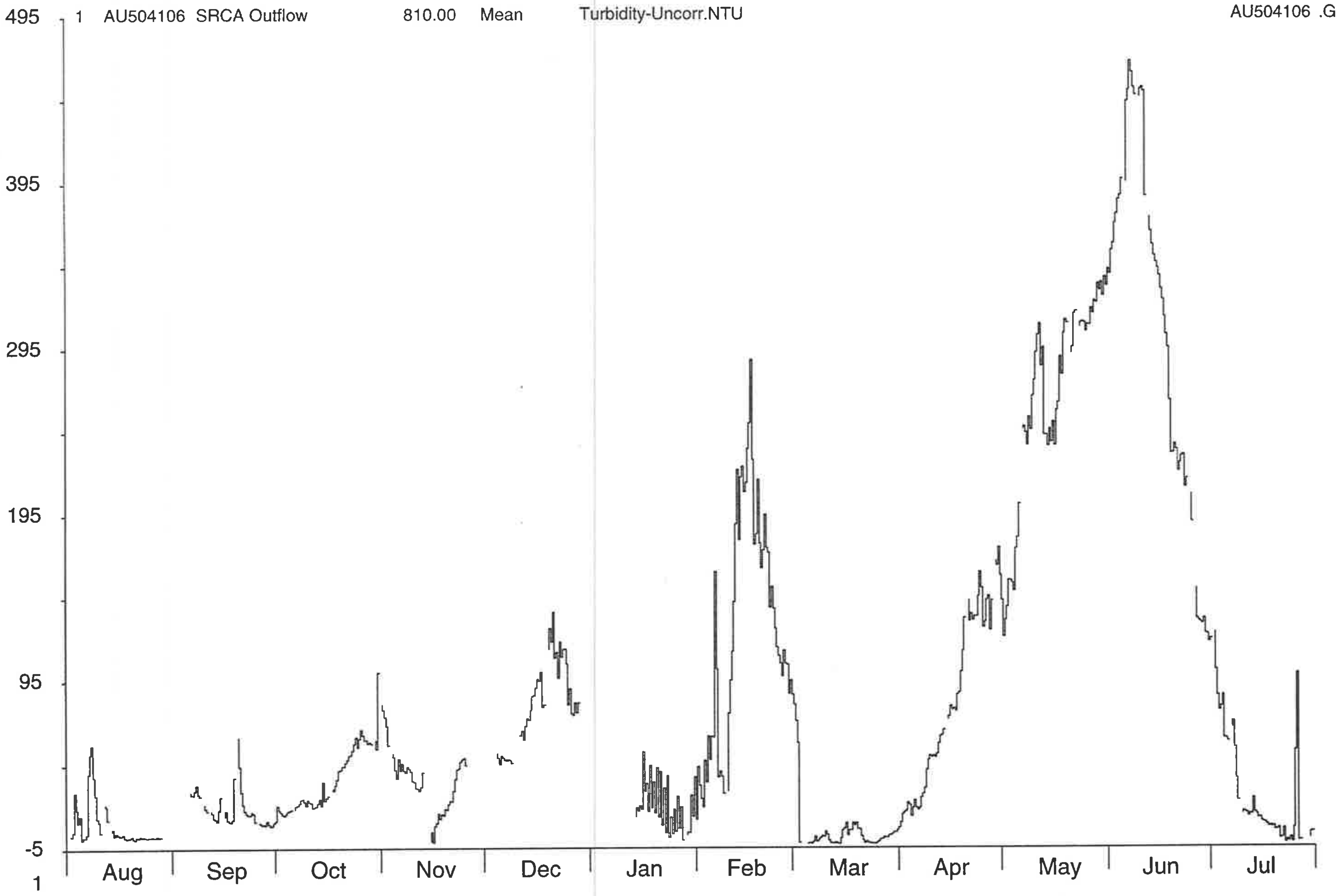


University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998



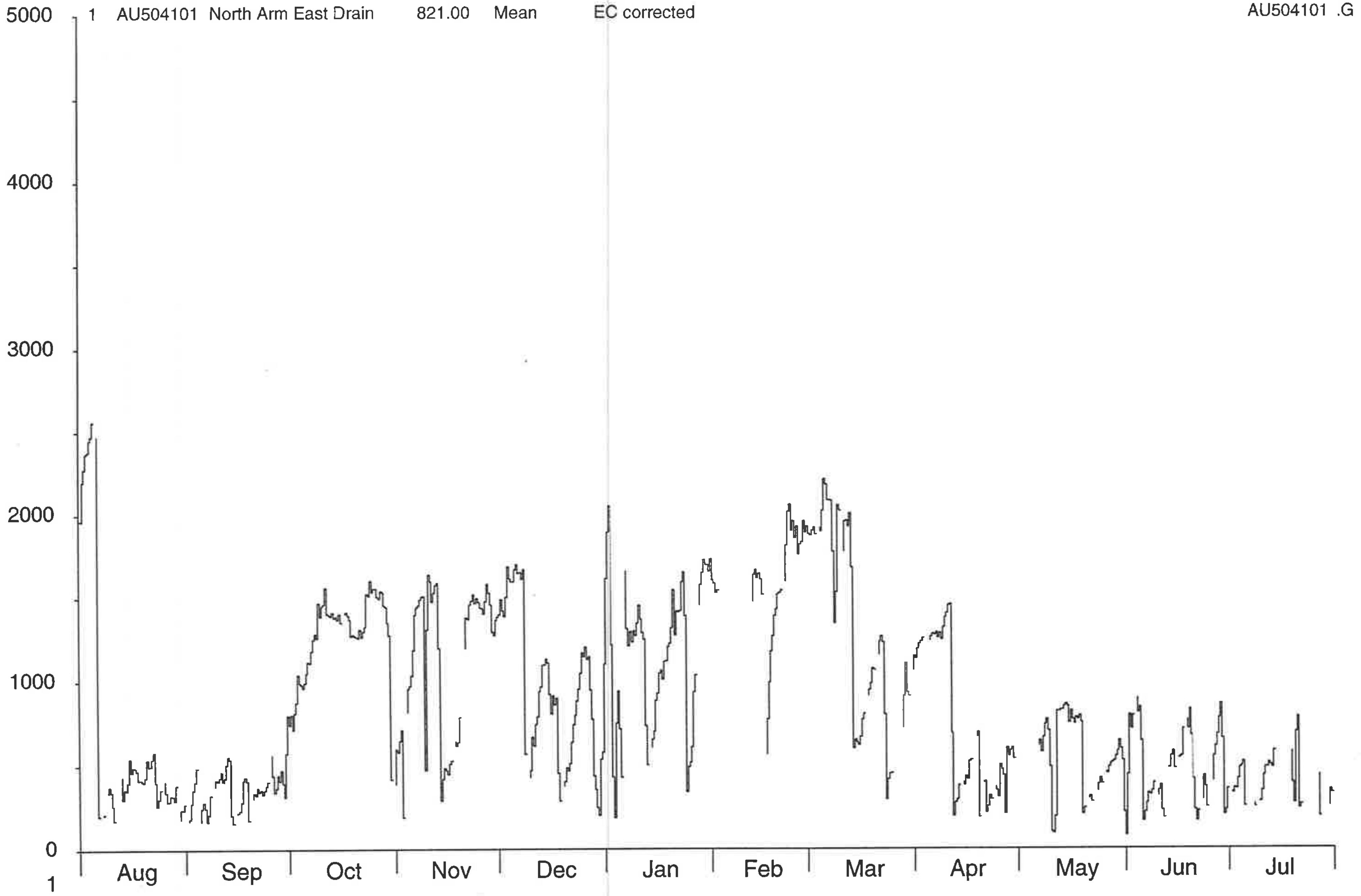
University of Adelaide

HYPLOT V89 Output 07/11/1999

Period 12 Month Plot Start 00:00_01/08/1997

Interval 12 Hour Plot End 00:00_01/08/1998

1997

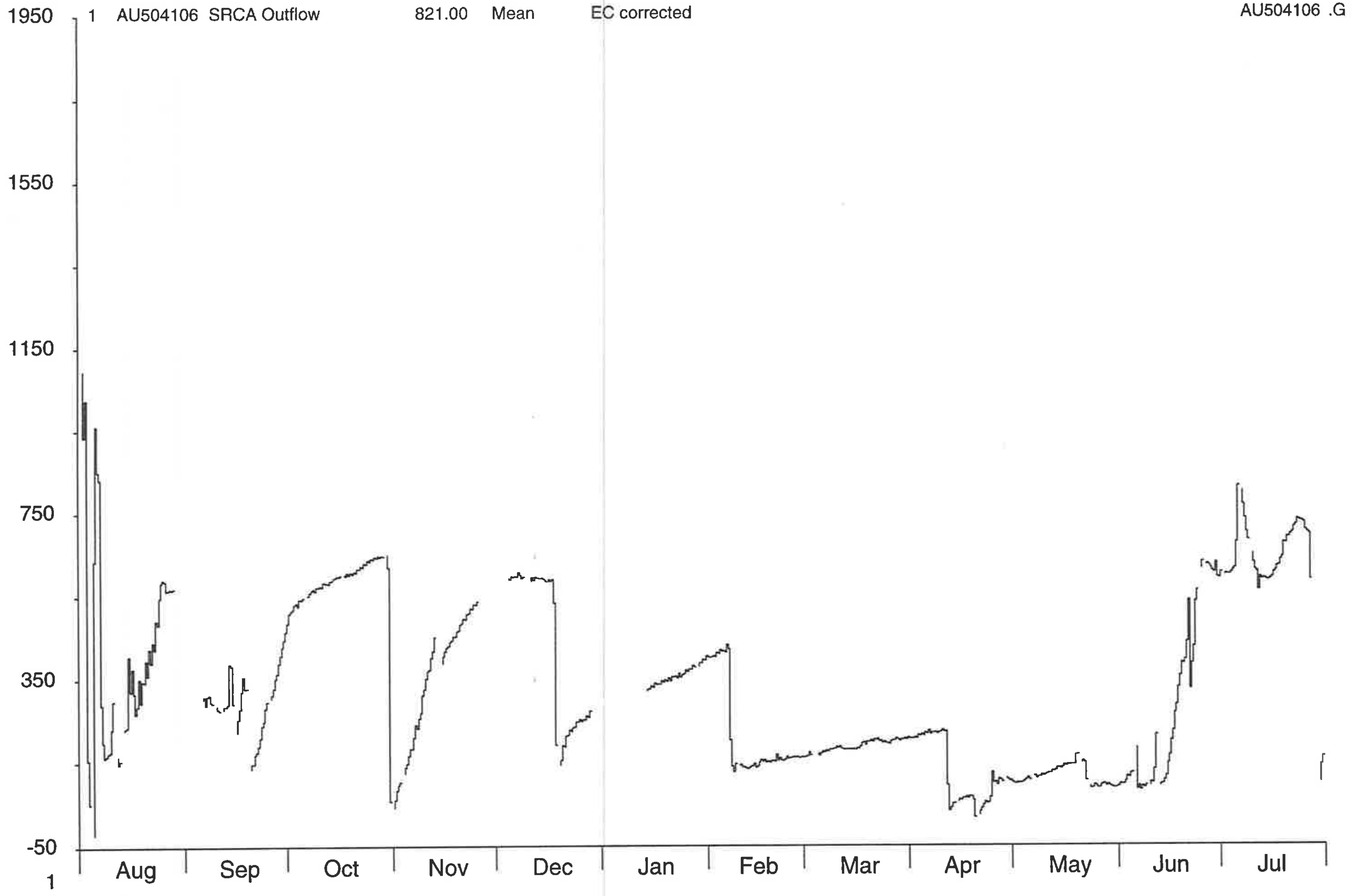


University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

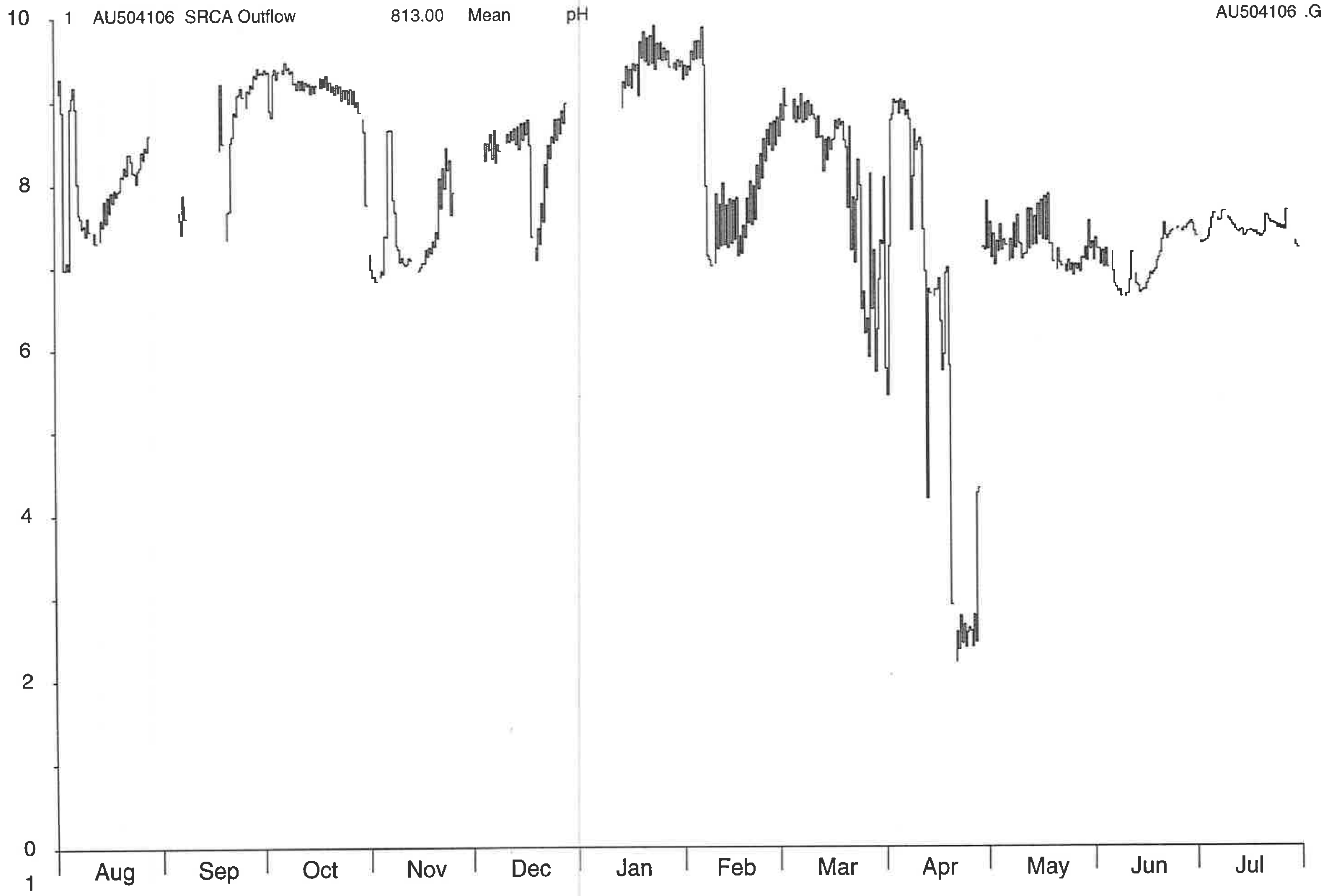


University of Adelaide

HYPLOT V89 Output 07/11/1999

1997

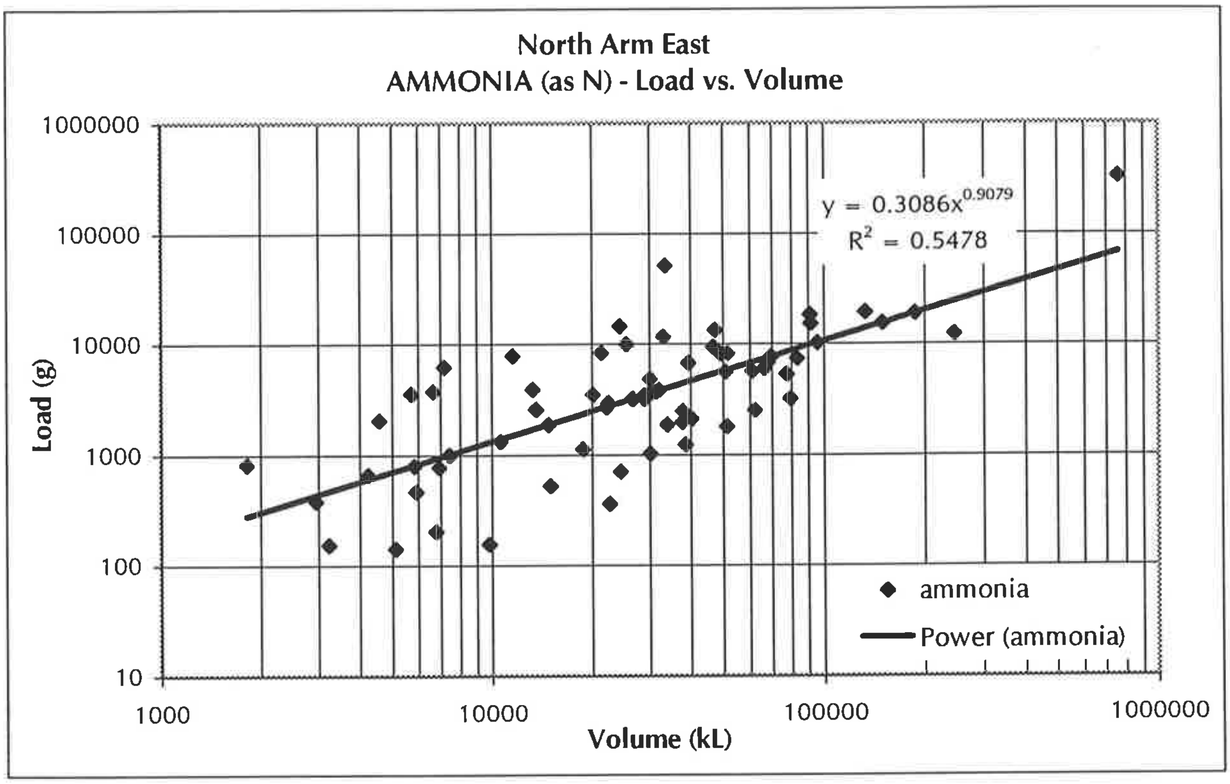
Period 12 Month Plot Start 00:00_01/08/1997
Interval 12 Hour Plot End 00:00_01/08/1998

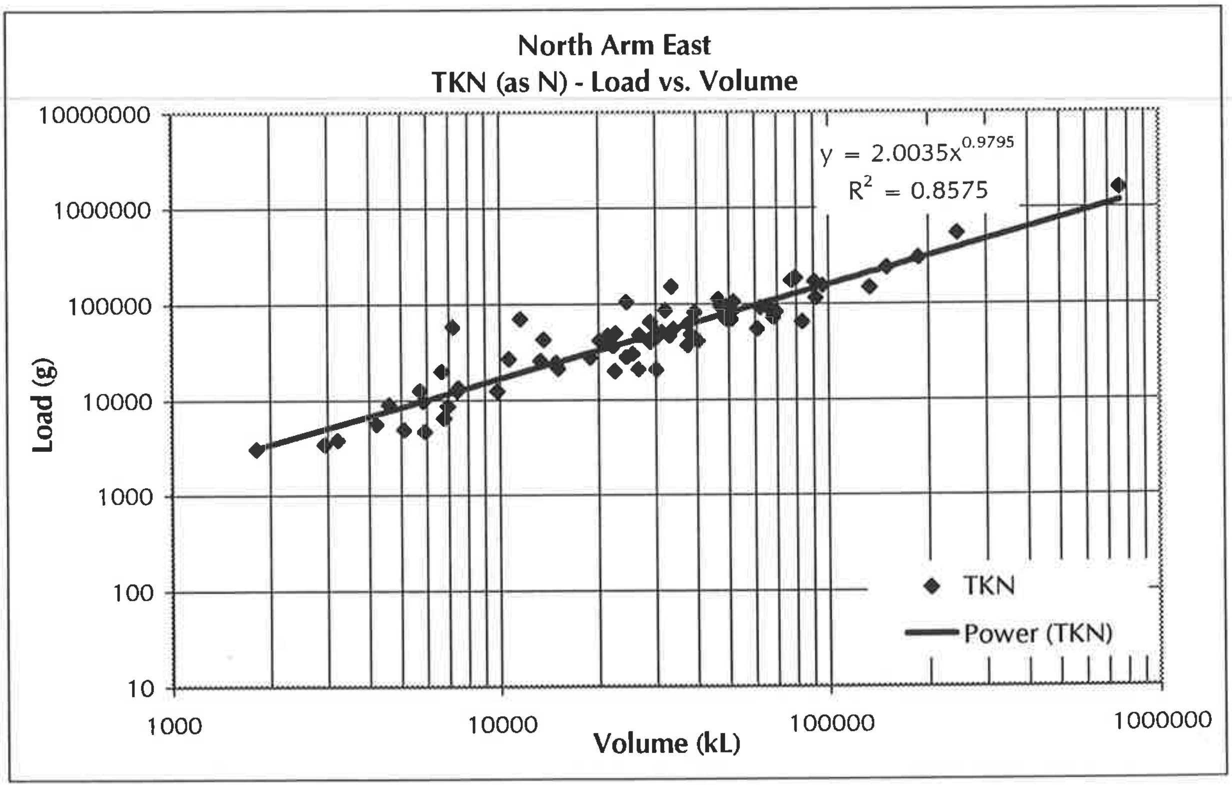


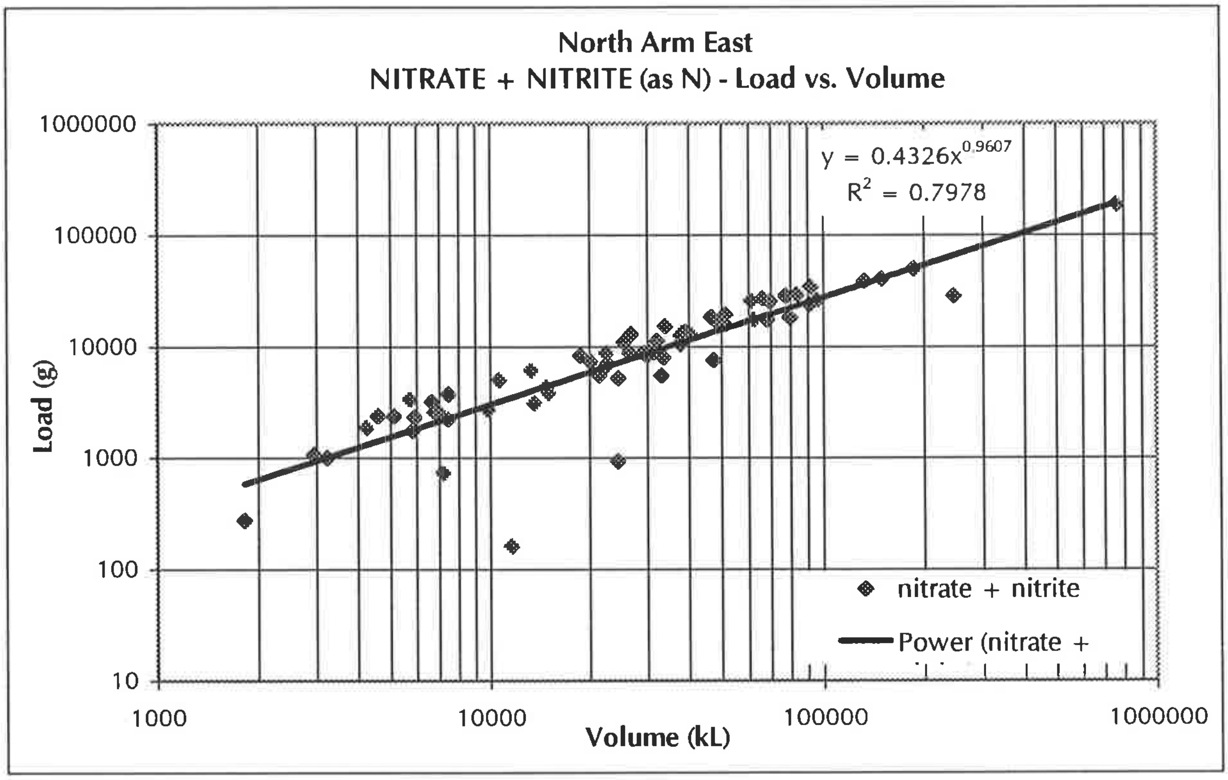
Appendix E

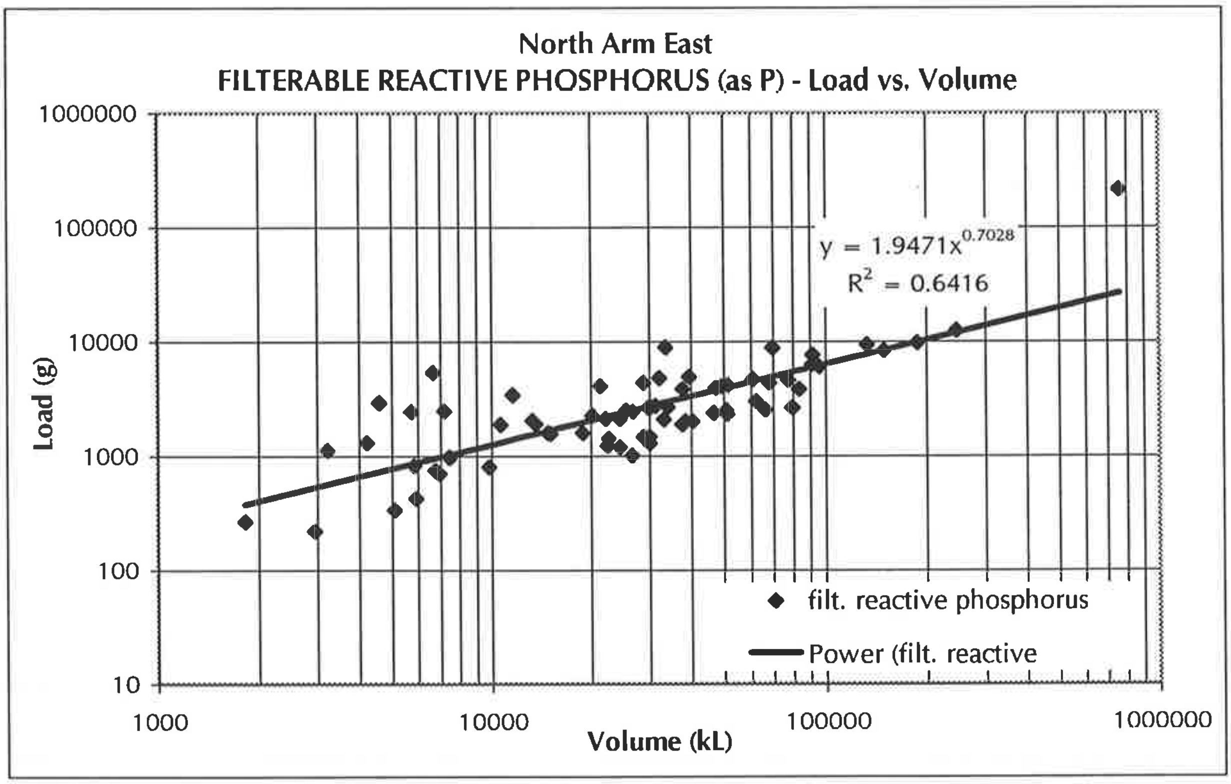
EVENT VOLUME / EVENT LOAD RELATIONSHIPS FOR NORTH ARM EAST & SOUTH ROAD CONNECTOR STATIONS

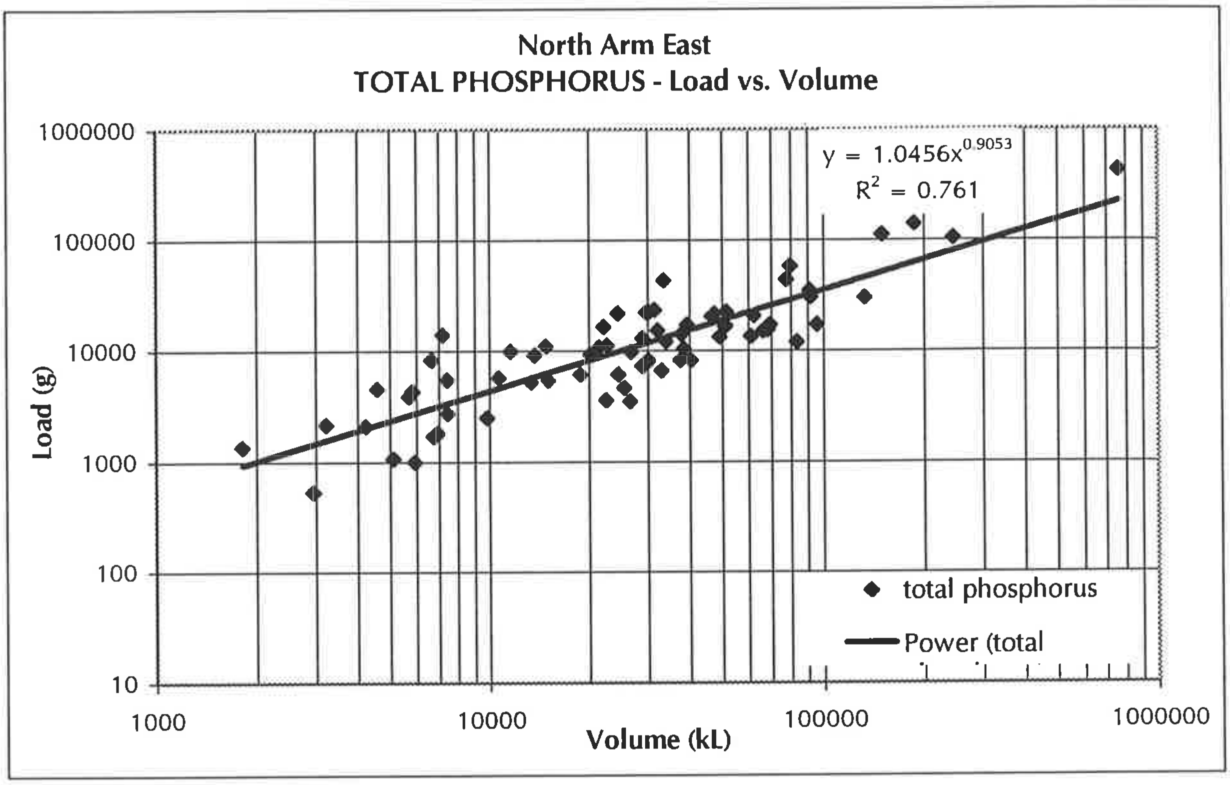
Contents:

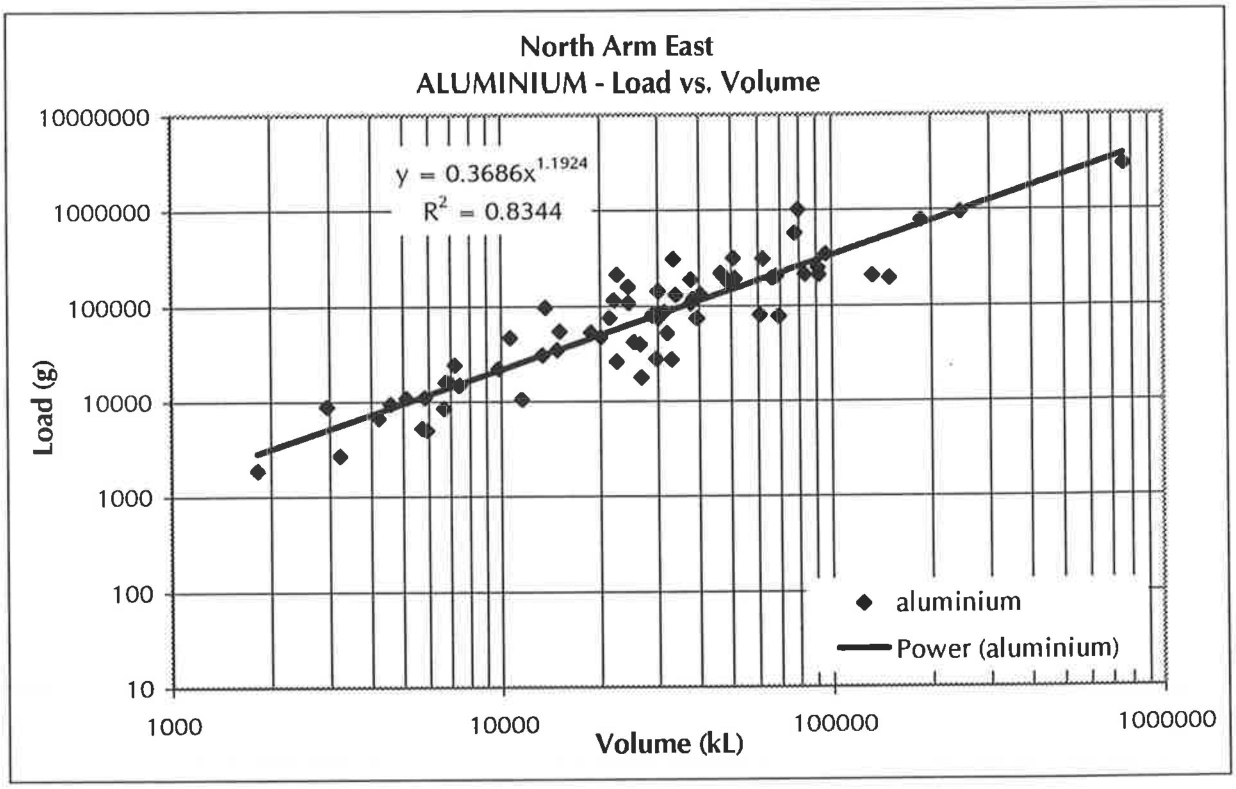

North Arm East
AMMONIA (as N) - Load vs. Volume
1000000
100000
10000
1000
100
10
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.3086x^0.9079
R^2 = 0.5478
◆ ammonia
Power (ammonia)

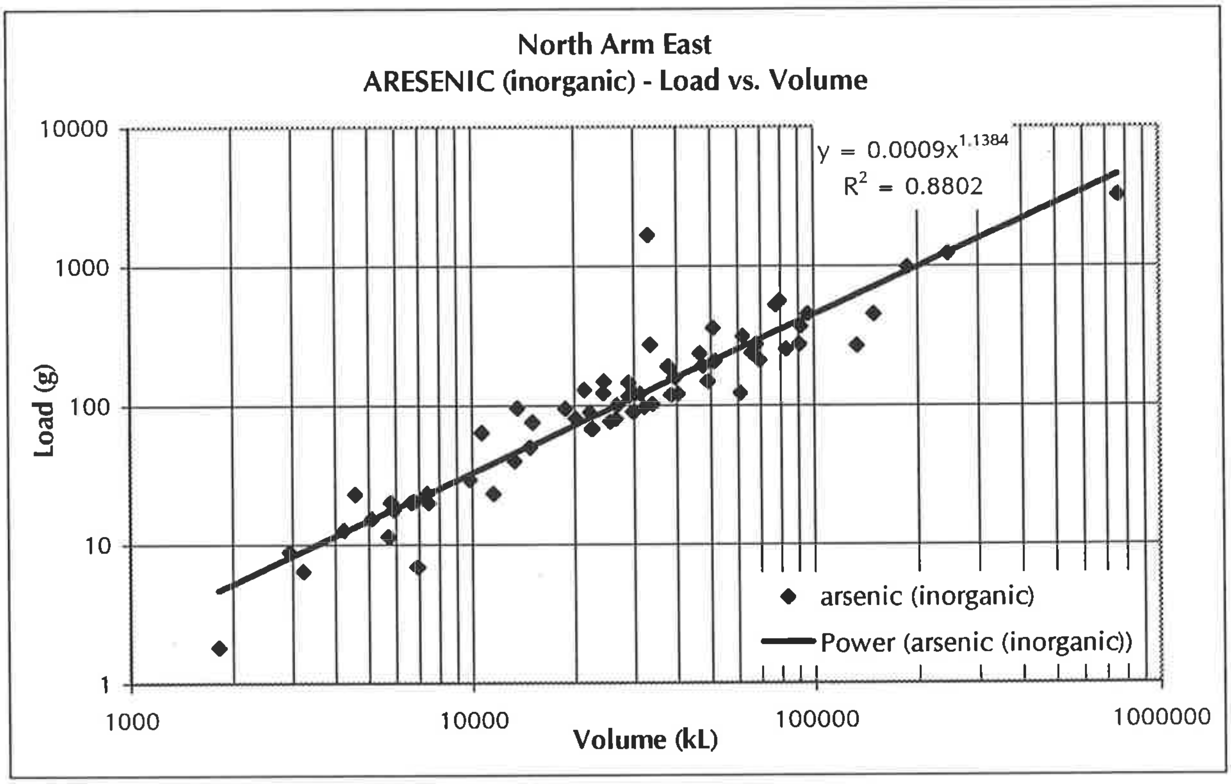

North Arm East
TKN (as N) - Load vs. Volume
10000000
1000000
100000
10000
1000
100
10
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 2.0035x^0.9795
R^2 = 0.8575
◆ TKN
Power (TKN)

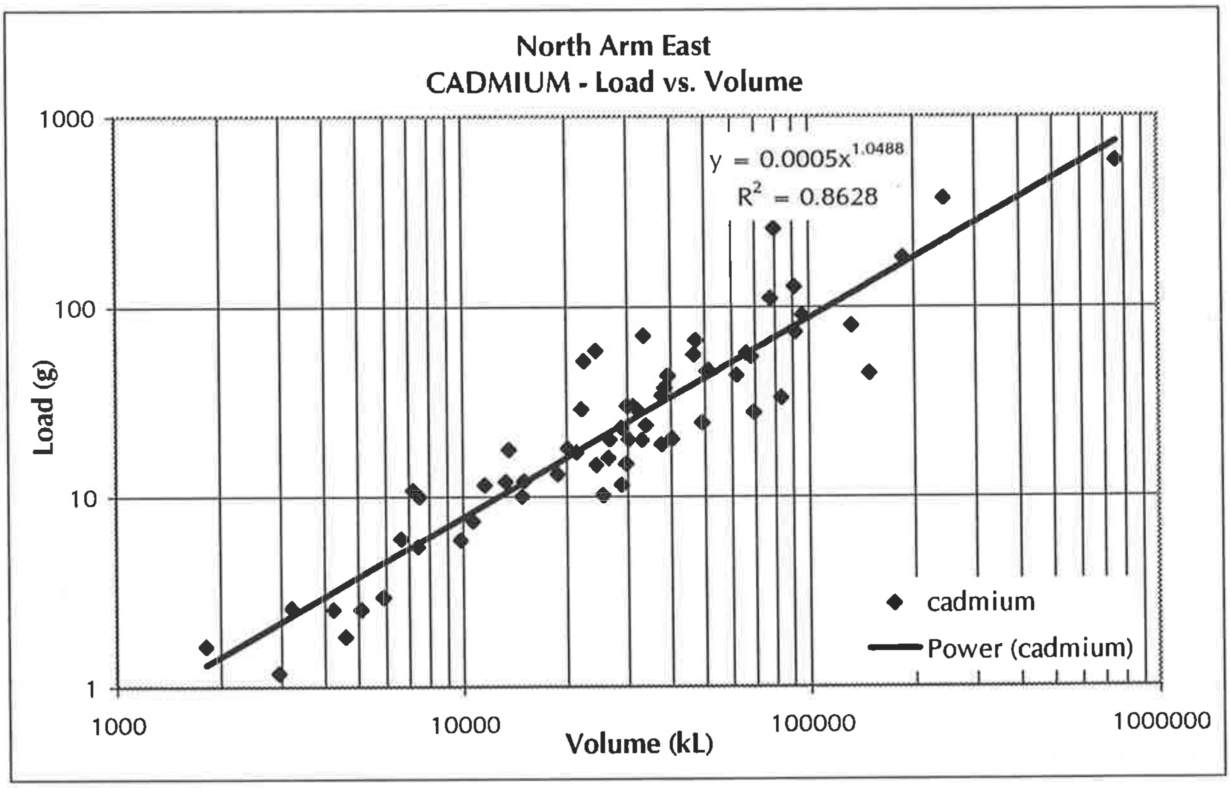

North Arm East
NITRATE + NITRITE (as N) - Load vs. Volume
1000000
100000
10000
1000
100
10
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.4326x^0.9607
R^2 = 0.7978
nitrate + nitrite
Power (nitrate +

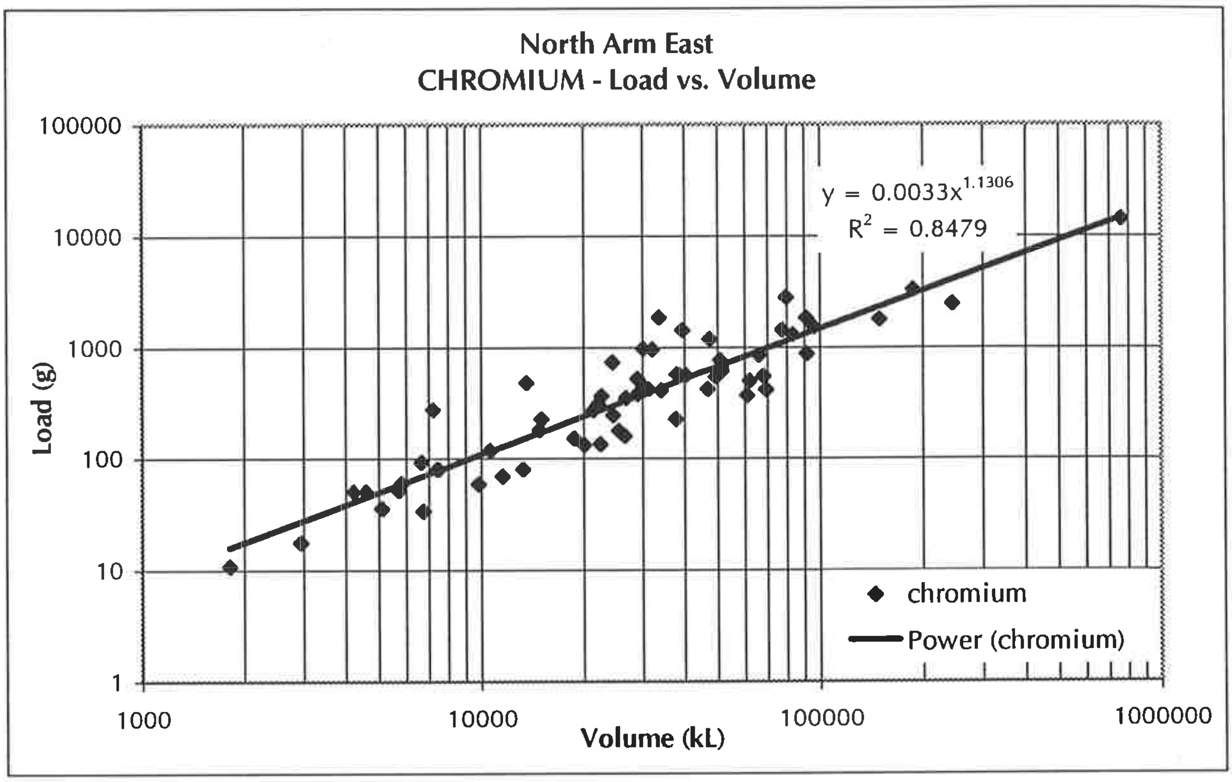

North Arm East
FILTERABLE REACTIVE PHOSPHORUS (as P) - Load vs. Volume
1000000
100000
10000
1000
100
10
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 1.9471x^0.7028
R^2 = 0.6416
filt. reactive phosphorus
Power (filt. reactive

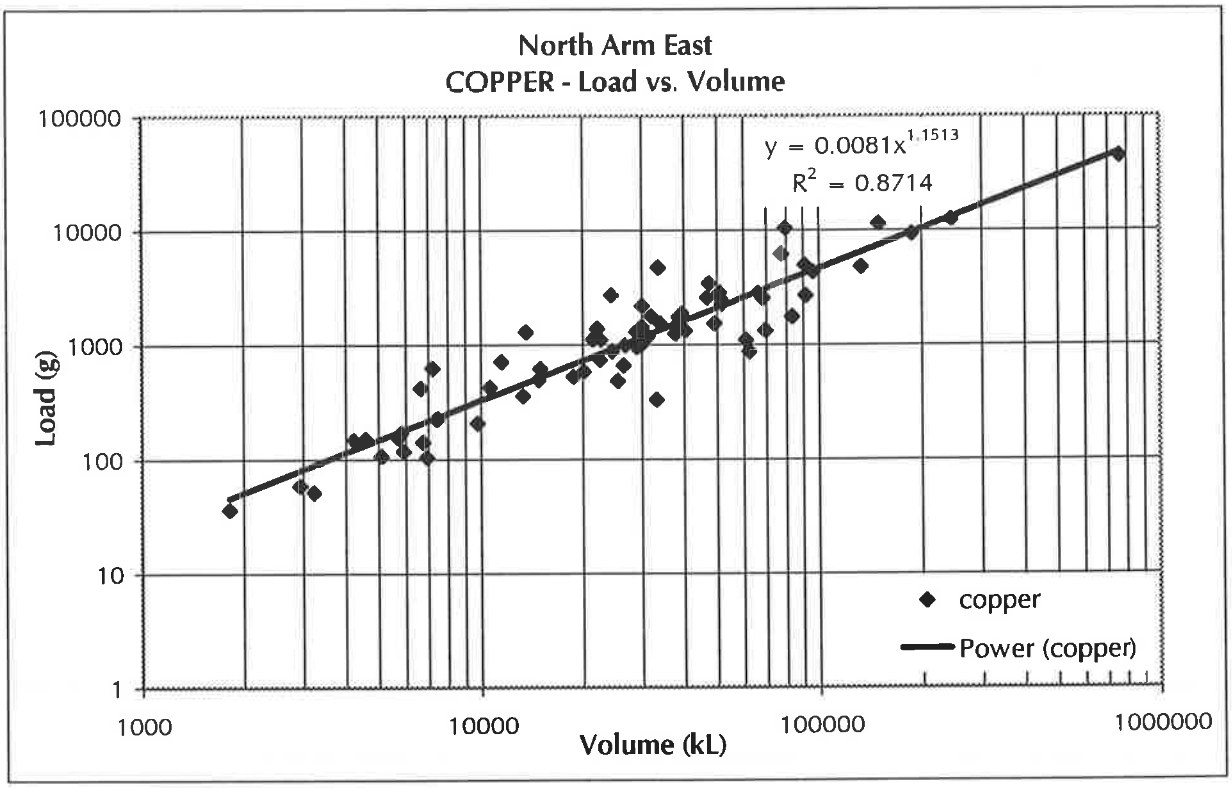

North Arm East
TOTAL PHOSPHORUS - Load vs. Volume
y = 1.0456x^0.9053
R² = 0.761
Load (g)
1000000
100000
10000
1000
100
10
1000
10000
100000
1000000
Volume (kL)
total phosphorus
Power (total

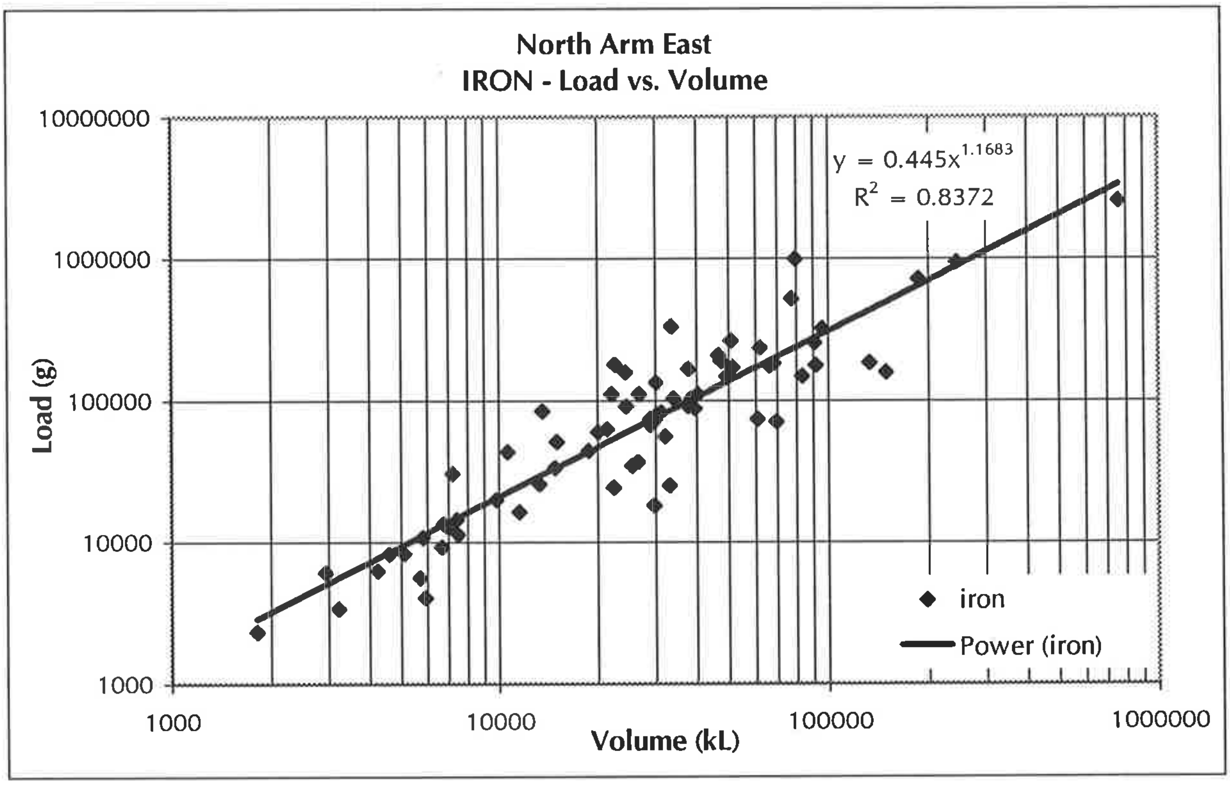

North Arm East
ALUMINIUM - Load vs. Volume
y = 0.3686x^1.1924
R² = 0.8344
Load (g)
10000000
1000000
100000
10000
1000
100
10
1000
10000
100000
1000000
Volume (kL)
aluminium
Power (aluminium)

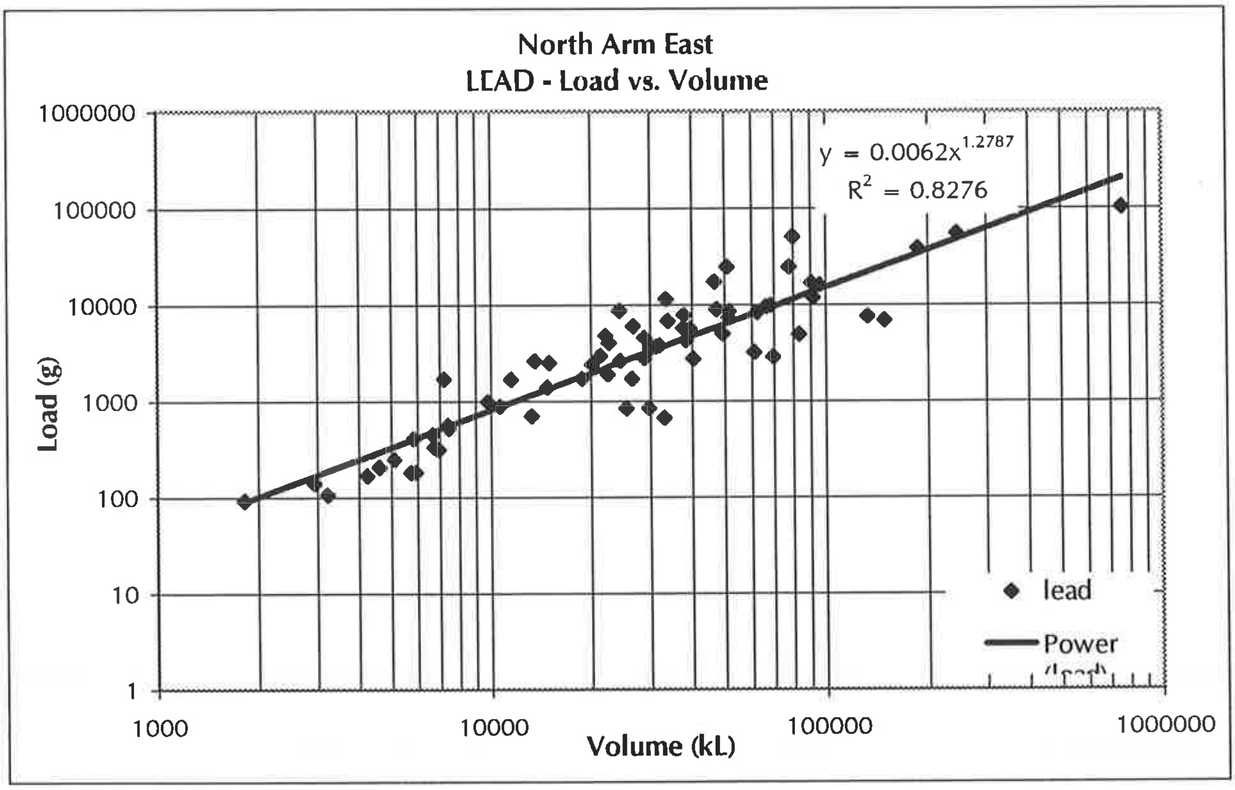

North Arm East
ARESENIC (inorganic) - Load vs. Volume
$y = 0.0009x^{1.1384}$
$R^2 = 0.8802$
Load (g)
10000
1000
100
10
1
Volume (kL)
1000
10000
100000
1000000
◆ arsenic (inorganic)
Power (arsenic (inorganic))

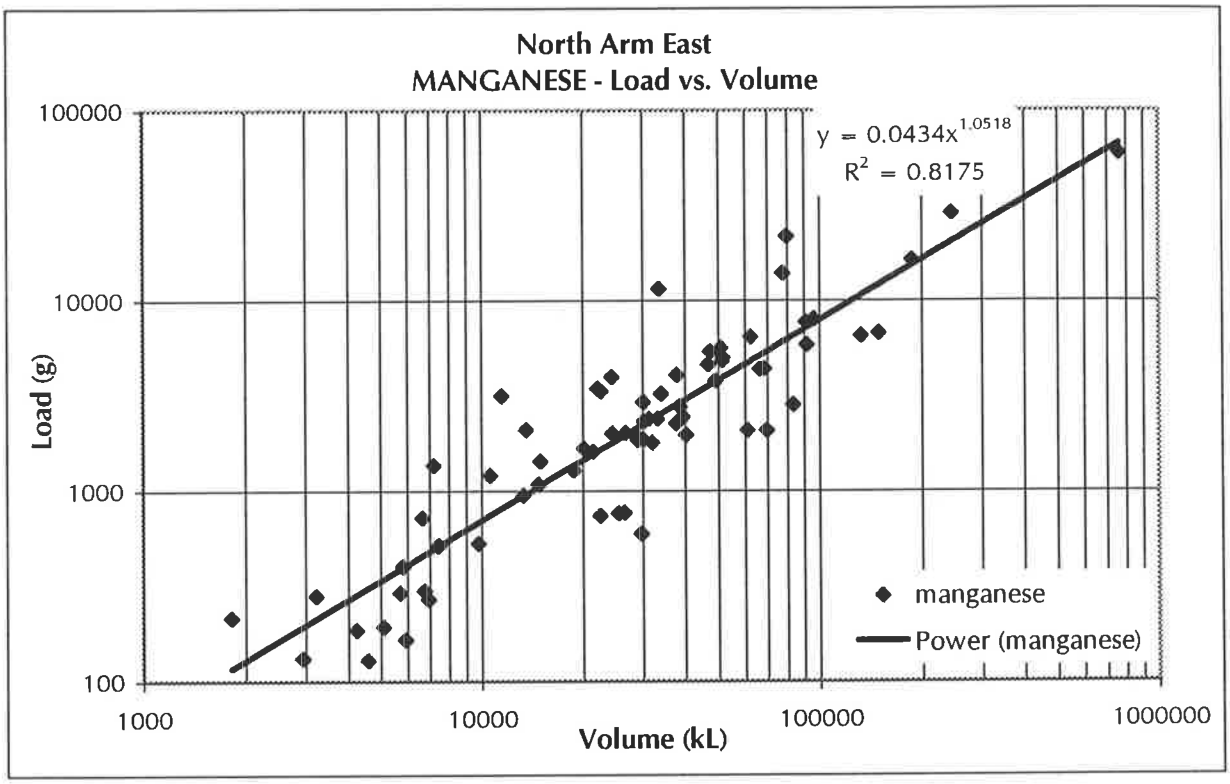

North Arm East
CADMIUM - Load vs. Volume
$y = 0.0005x^{1.0488}$
$R^2 = 0.8628$
Load (g)
1000
100
10
1
Volume (kL)
1000
10000
100000
1000000
◆ cadmium
Power (cadmium)

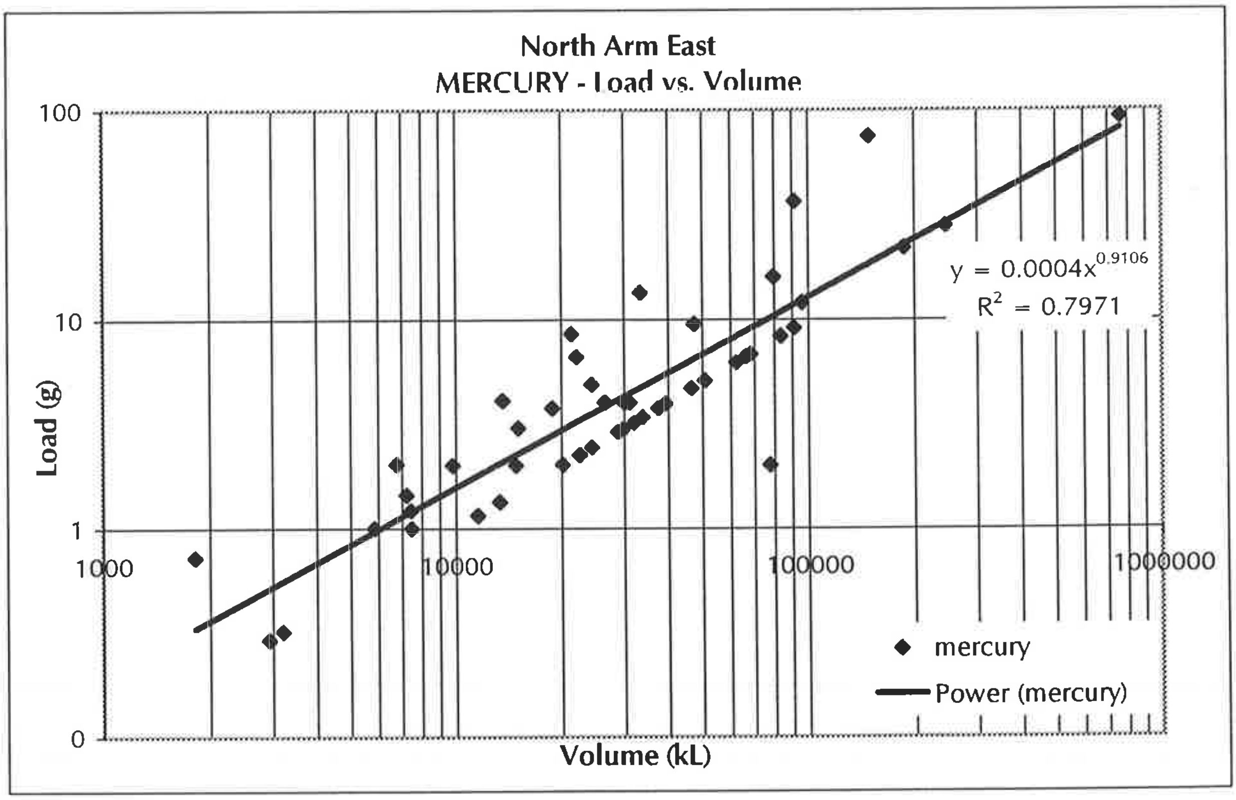

North Arm East
CHROMIUM - Load vs. Volume
y = 0.0033x^1.1306
R^2 = 0.8479
Load (g)
100000
10000
1000
100
10
1
Volume (kL)
1000
10000
100000
1000000
chromium
Power (chromium)

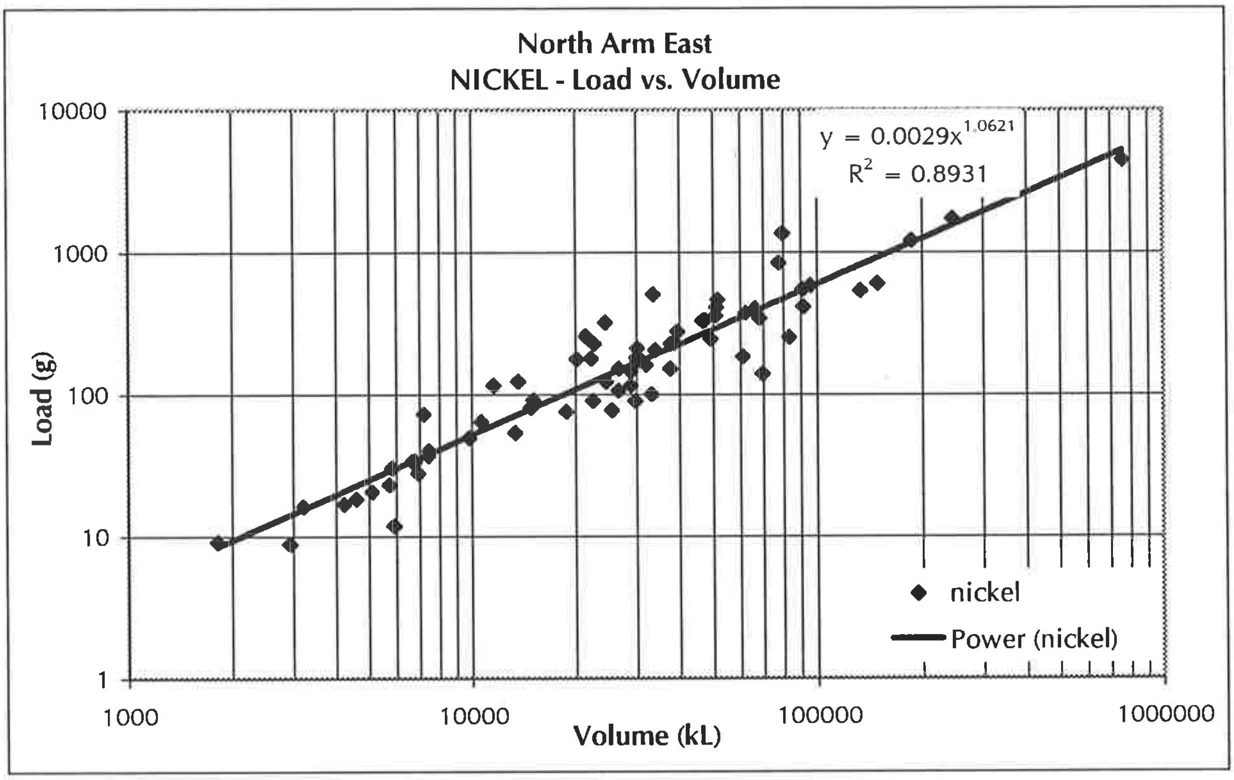

North Arm East
COPPER - Load vs. Volume
y = 0.0081x^1.1513
R^2 = 0.8714
Load (g)
100000
10000
1000
100
10
1
Volume (kL)
1000
10000
100000
1000000
copper
Power (copper)

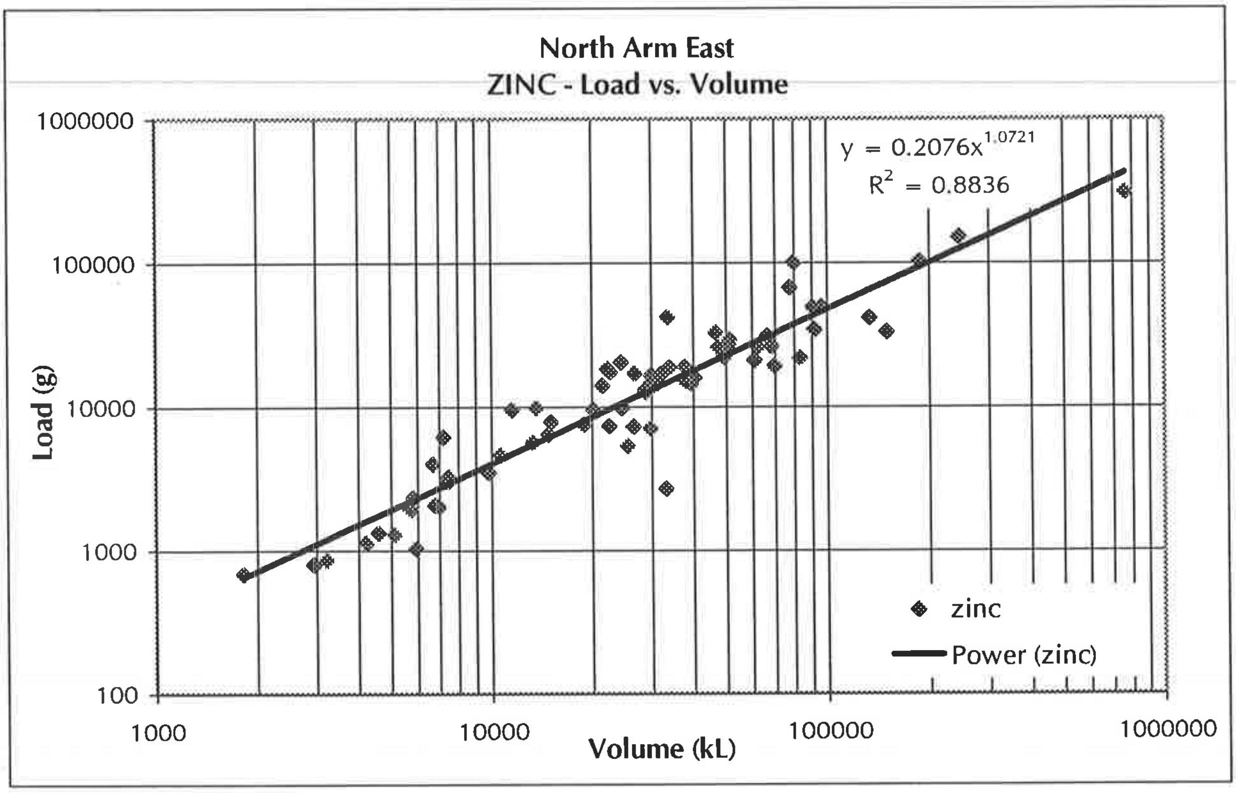

North Arm East
IRON - Load vs. Volume
y = 0.445x^1.1683
R^2 = 0.8372
Load (g)
10000000
1000000
100000
10000
1000
1000
10000
100000
1000000
Volume (kL)
◆ iron
Power (iron)

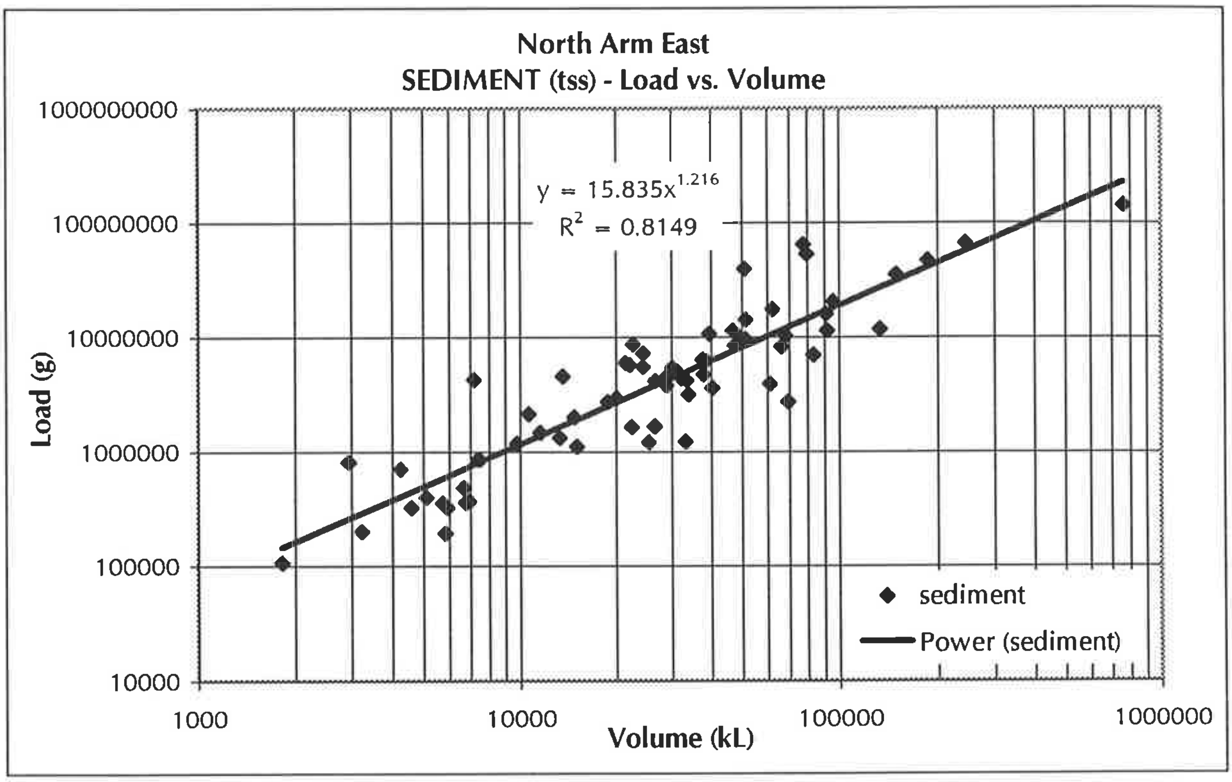

North Arm East
LEAD - Load vs. Volume
y = 0.0062x^1.2787
R^2 = 0.8276
Load (g)
1000000
100000
10000
1000
100
10
1
1000
10000
100000
1000000
Volume (kL)
◆ lead
Power (lead)

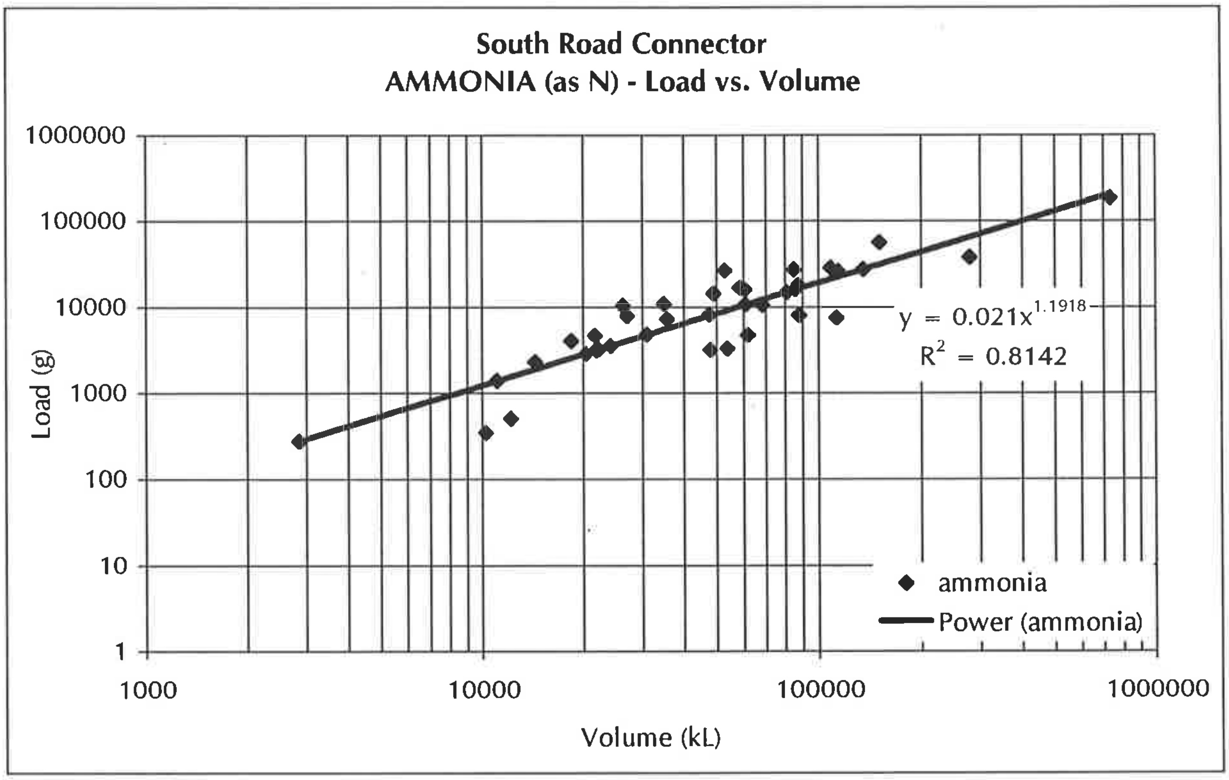

North Arm East
MANGANESE - Load vs. Volume
y = 0.0434x^1.0518
R² = 0.8175
Load (g)
100000
10000
1000
100
1000
10000
100000
1000000
Volume (kL)
manganese
Power (manganese)

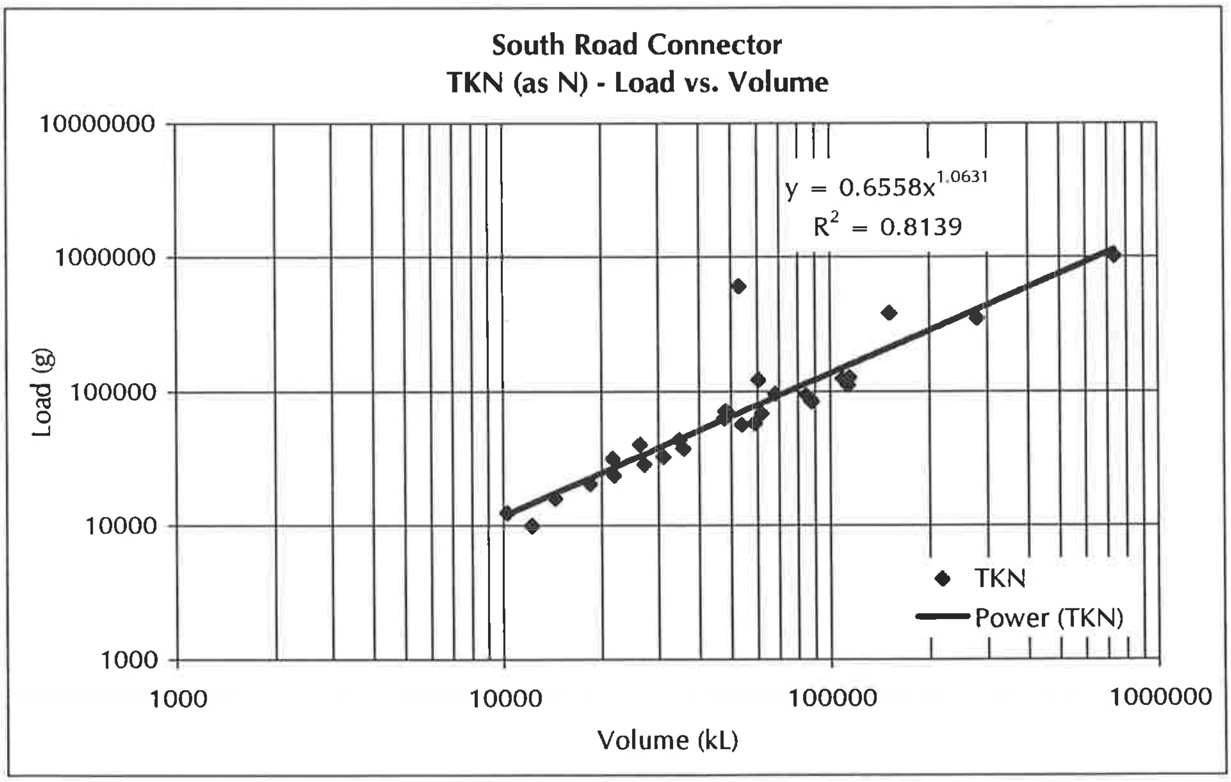

North Arm East
MERCURY - Load vs. Volume
y = 0.0004x^0.9106
R² = 0.7971
Load (g)
100
10
1
0
1000
10000
100000
1000000
Volume (kL)
mercury
Power (mercury)

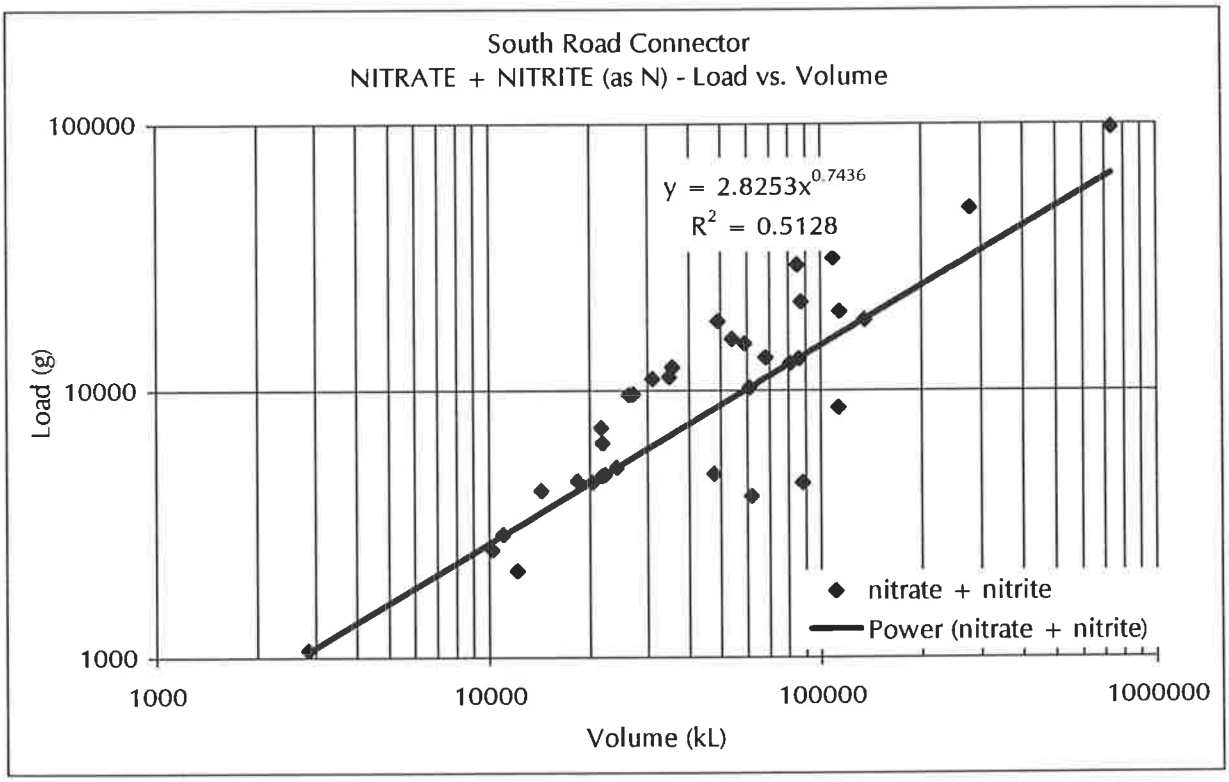

North Arm East
NICKEL - Load vs. Volume
y = 0.0029x^1.0621
R^2 = 0.8931
Load (g)
10000
1000
100
10
1
1000
10000
100000
1000000
Volume (kL)
nickel
Power (nickel)

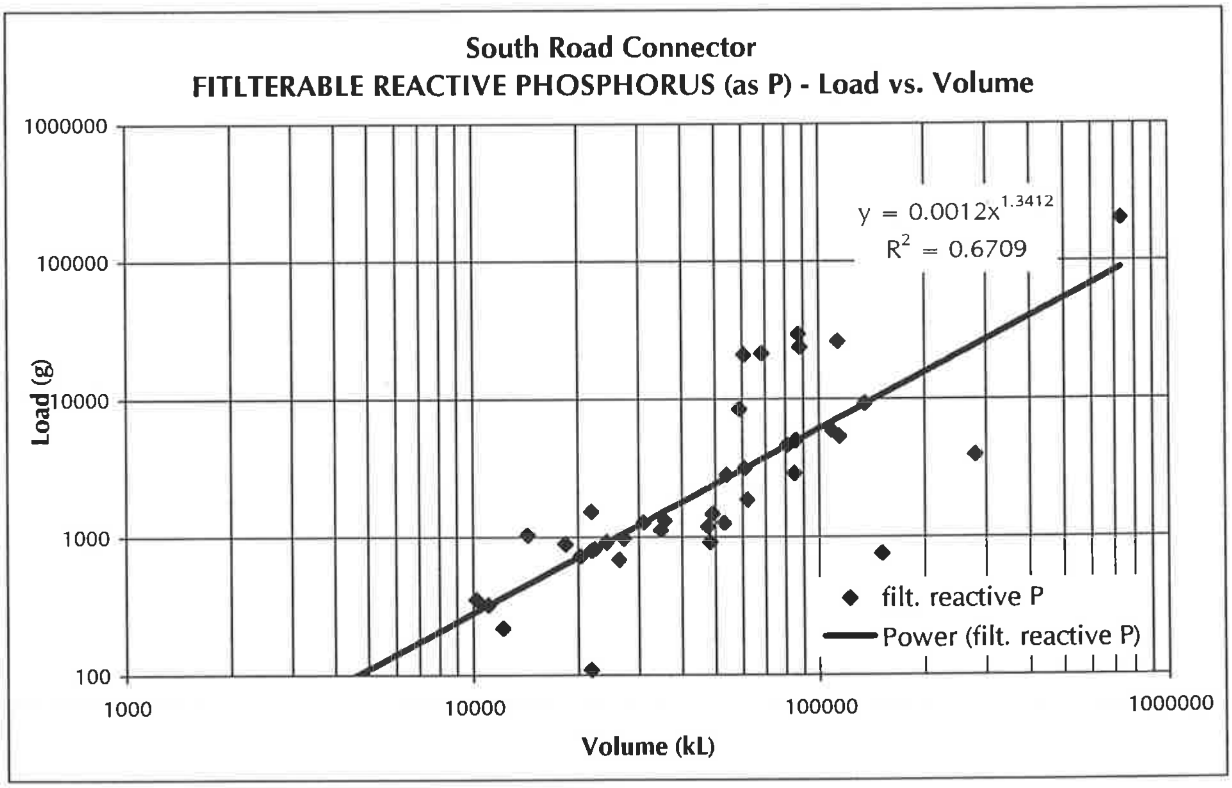

North Arm East
ZINC - Load vs. Volume
y = 0.2076x^1.0721
R^2 = 0.8836
Load (g)
1000000
100000
10000
1000
100
1000
10000
100000
1000000
Volume (kL)
zinc
Power (zinc)

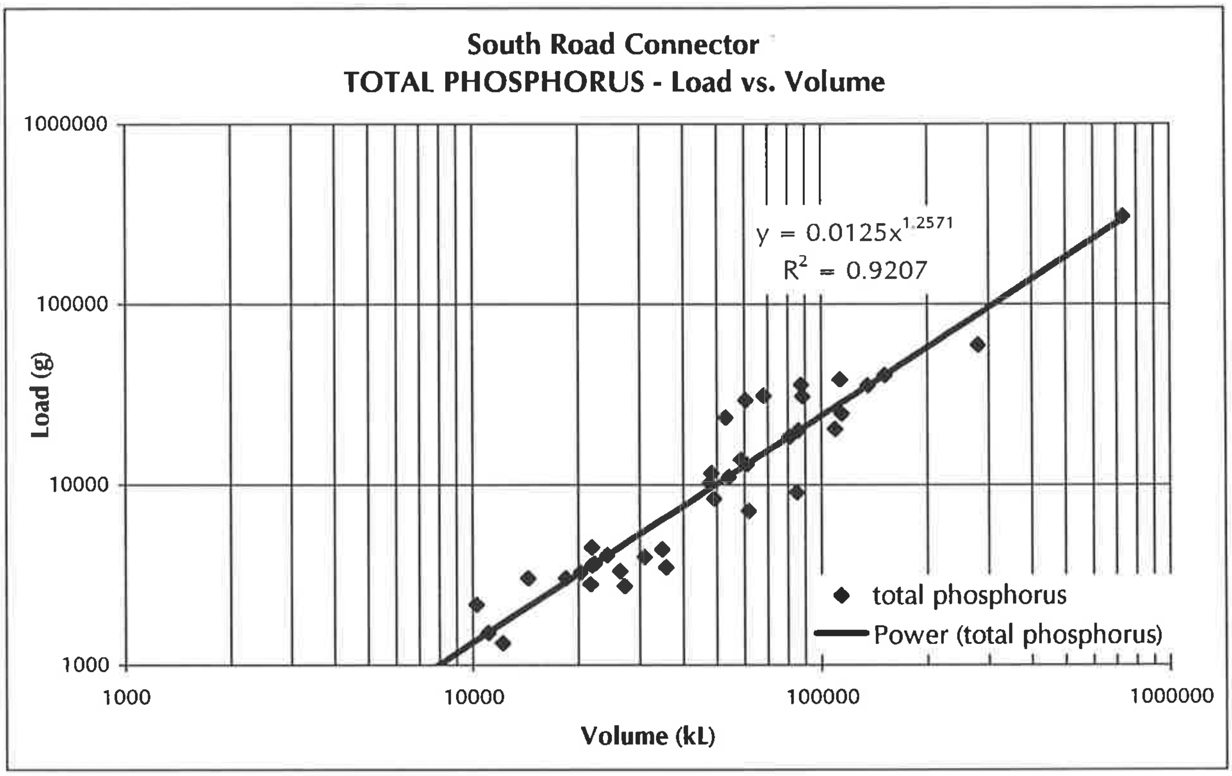

North Arm East
SEDIMENT (tss) - Load vs. Volume
Load (g)
1000000000
100000000
10000000
1000000
100000
10000
Volume (kL)
1000
10000
100000
1000000
y = 15.835x^1.216
R^2 = 0.8149
sediment
Power (sediment)

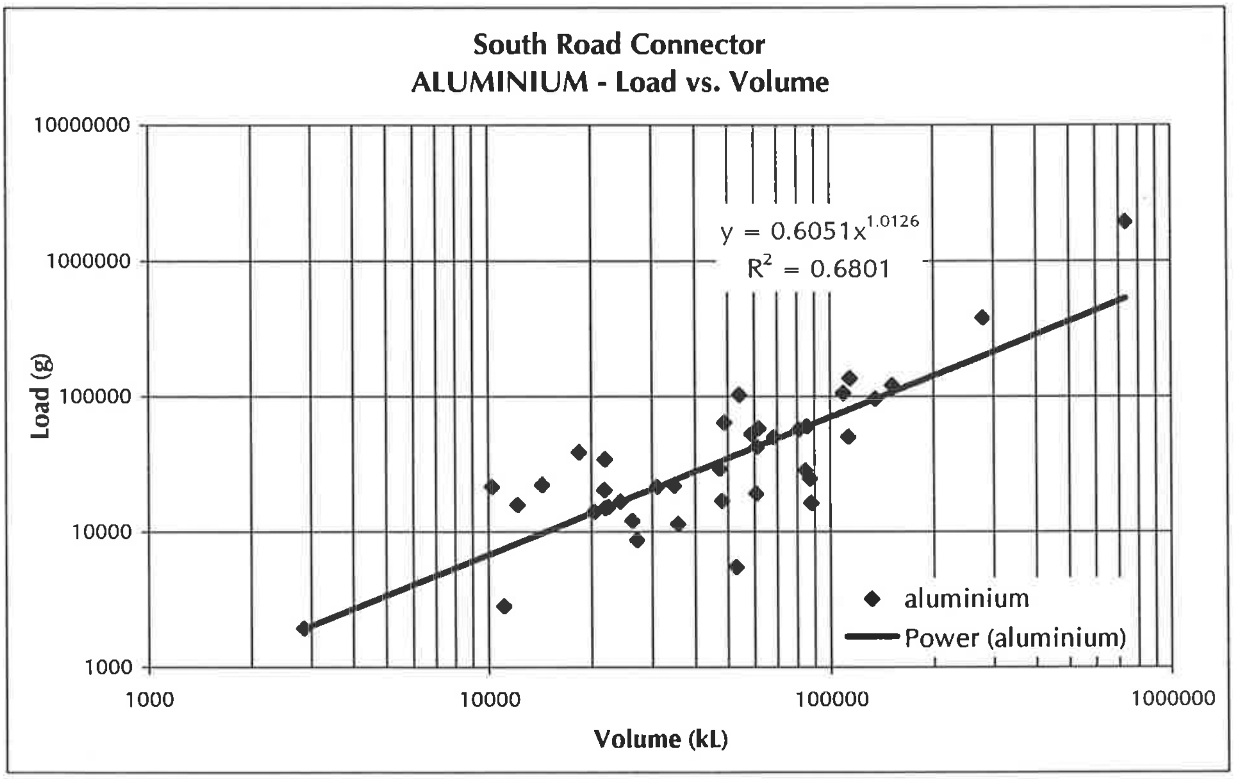

South Road Connector
AMMONIA (as N) - Load vs. Volume
1000000
100000
10000
1000
100
10
1
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.021x^1.1918
R^2 = 0.8142
ammonia
Power (ammonia)

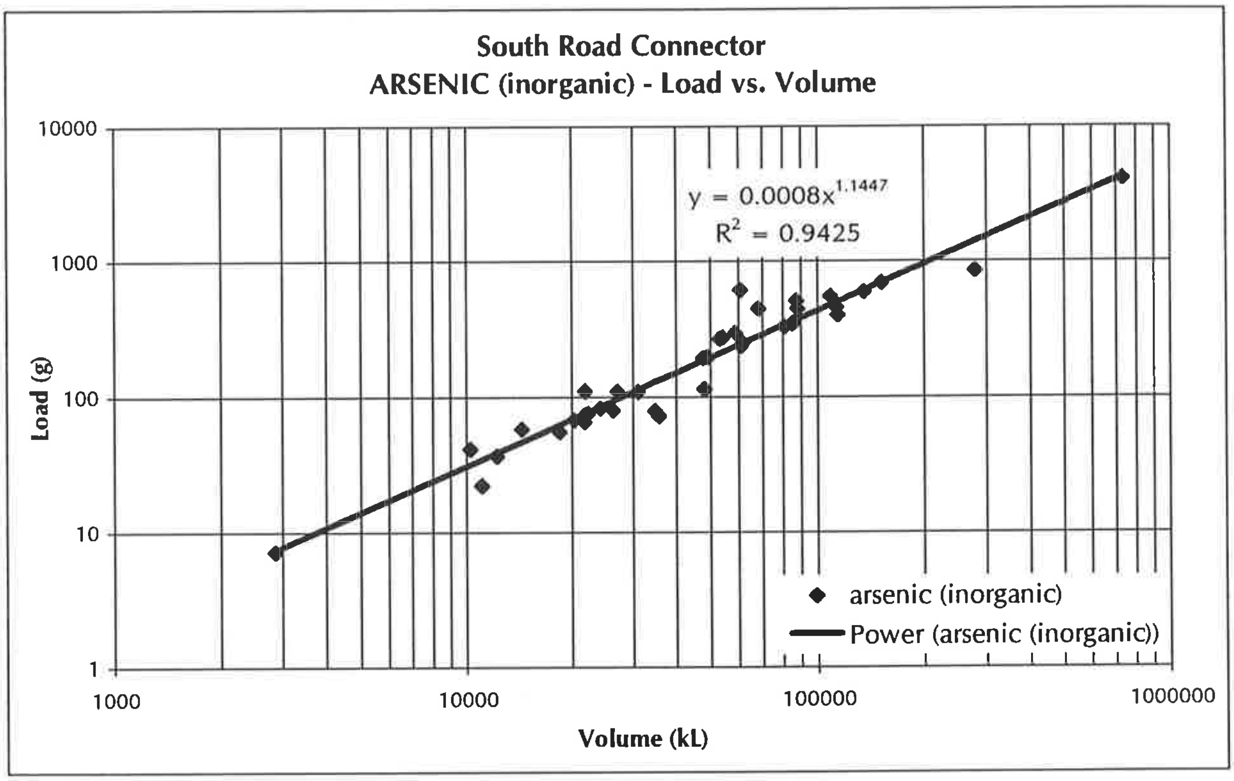

South Road Connector
TKN (as N) - Load vs. Volume
10000000
1000000
100000
10000
1000
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.6558x^1.0631
R^2 = 0.8139
TKN
Power (TKN)

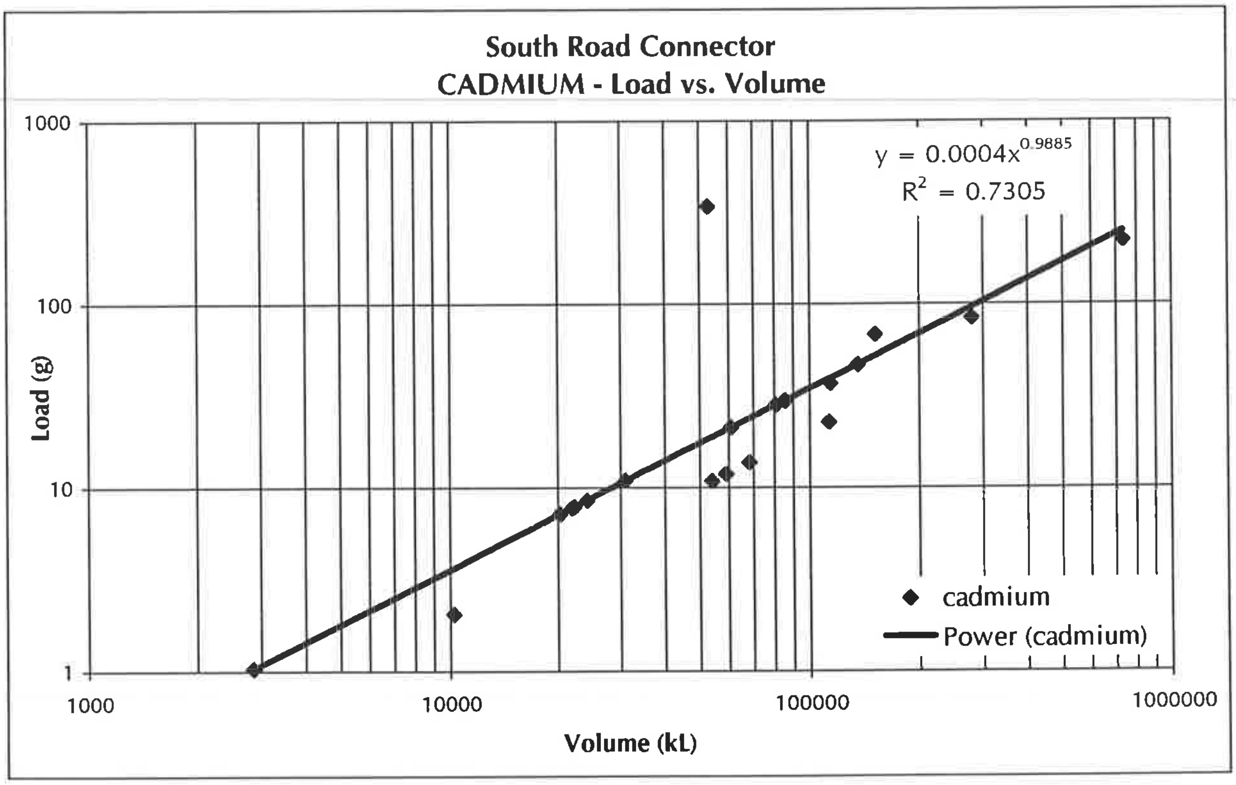

South Road Connector
NITRATE + NITRITE (as N) - Load vs. Volume
y = 2.8253x^0.7436
R^2 = 0.5128
Load (g)
100000
10000
1000
1000
10000
100000
1000000
Volume (kL)
nitrate + nitrite
Power (nitrate + nitrite)

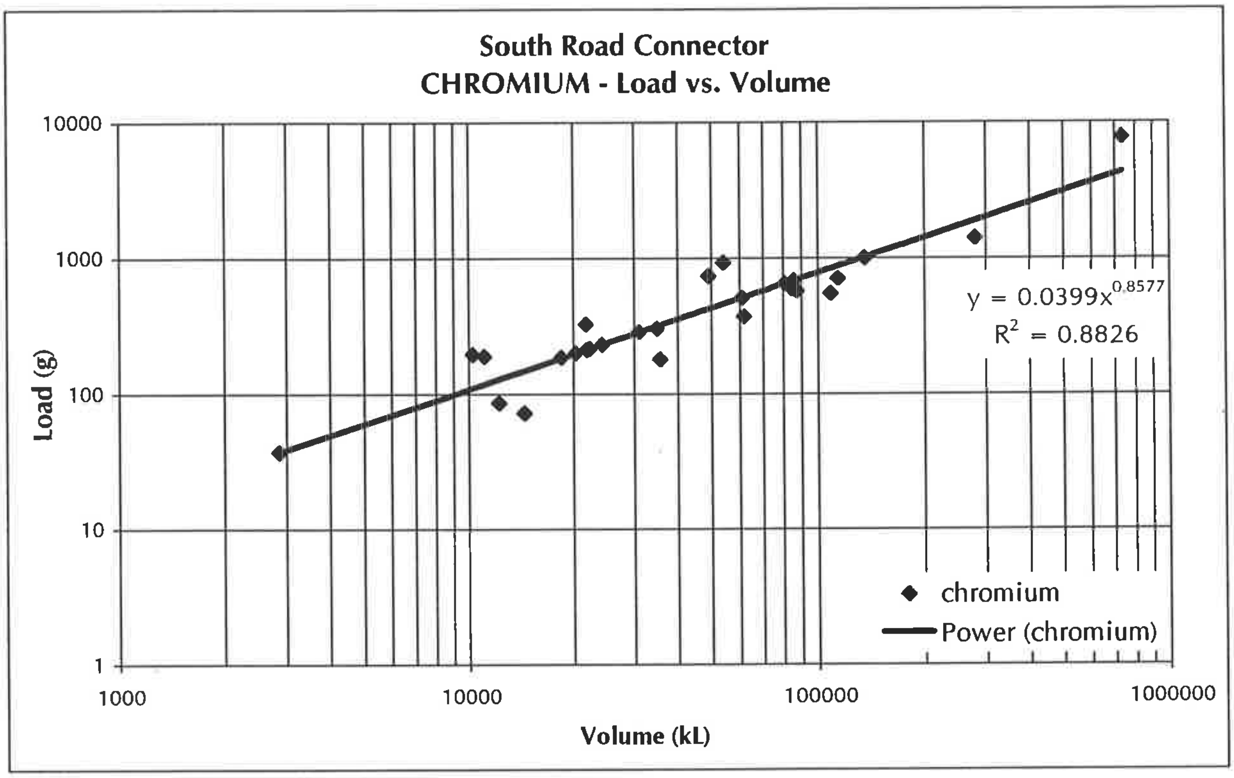

South Road Connector
FITLTERABLE REACTIVE PHOSPHORUS (as P) - Load vs. Volume
y = 0.0012x^1.3412
R^2 = 0.6709
Load (g)
1000000
100000
10000
1000
100
1000
10000
100000
1000000
Volume (kL)
filt. reactive P
Power (filt. reactive P)

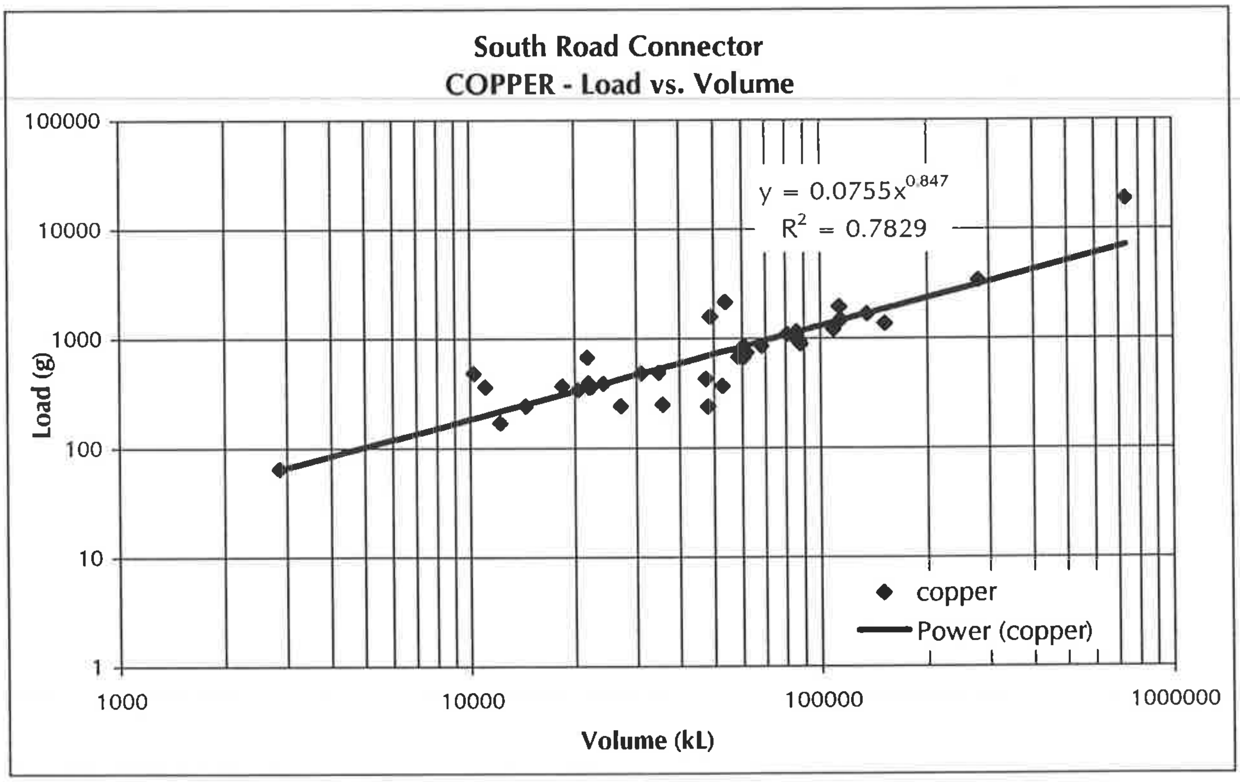

South Road Connector
TOTAL PHOSPHORUS - Load vs. Volume
y = 0.0125x^1.2571
R² = 0.9207
Load (g)
1000000
100000
10000
1000
1000
10000
100000
1000000
Volume (kL)
total phosphorus
Power (total phosphorus)

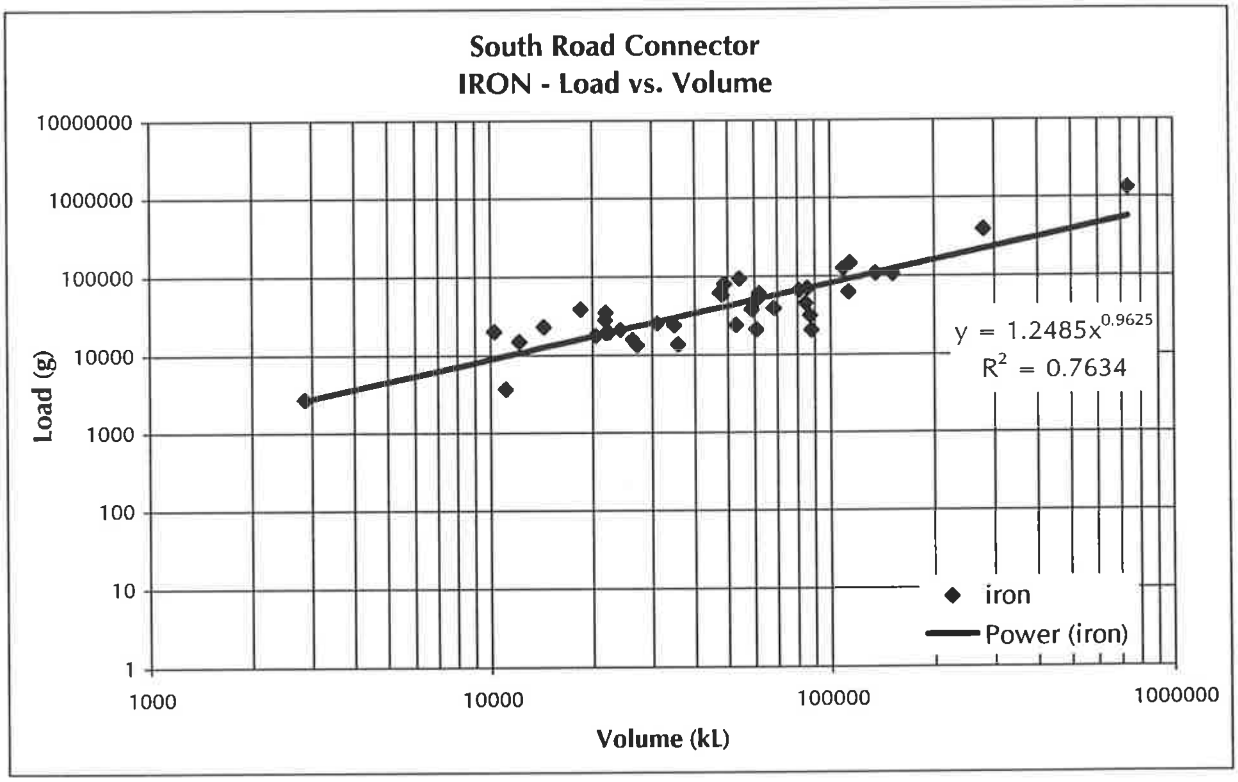

South Road Connector
ALUMINIUM - Load vs. Volume
y = 0.6051x^1.0126
R² = 0.6801
Load (g)
10000000
1000000
100000
10000
1000
1000
10000
100000
1000000
Volume (kL)
aluminium
Power (aluminium)

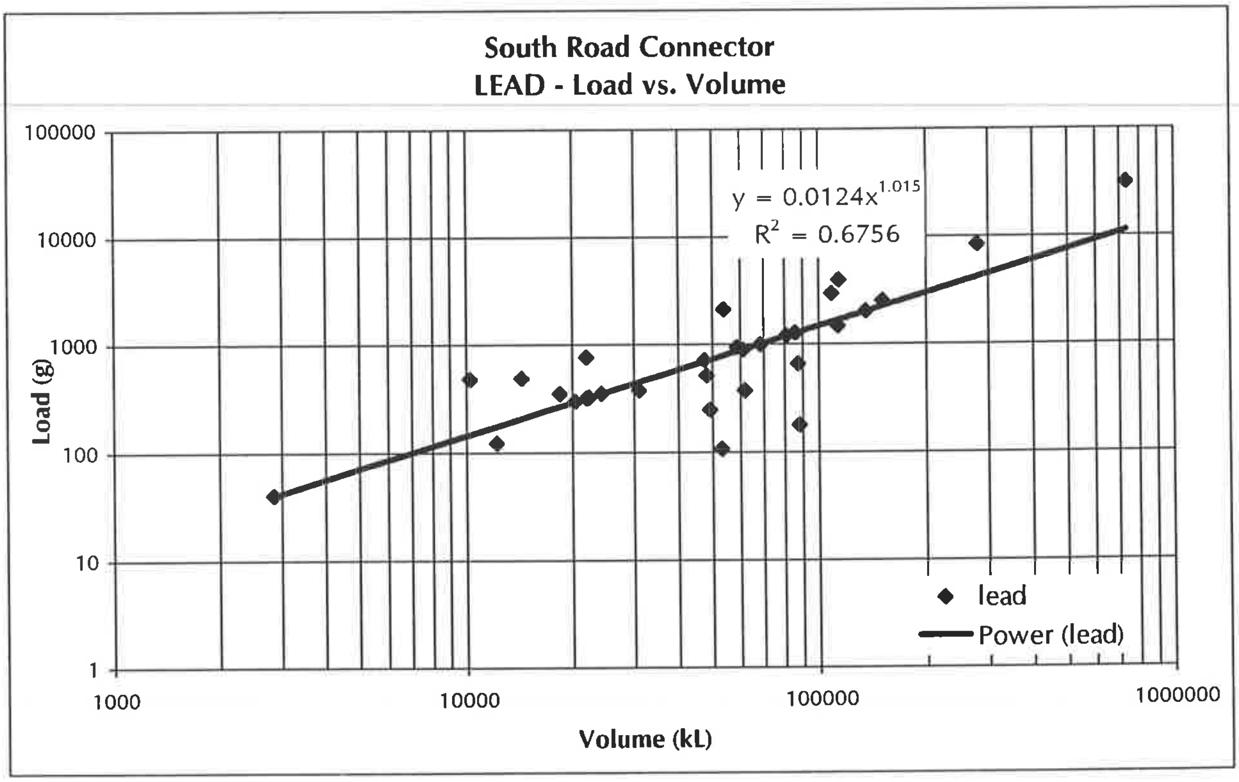

South Road Connector
ARSENIC (inorganic) - Load vs. Volume
y = 0.0008x^1.1447
R² = 0.9425
Load (g)
10000
1000
100
10
1
Volume (kL)
1000
10000
100000
1000000
◆ arsenic (inorganic)
Power (arsenic (inorganic))

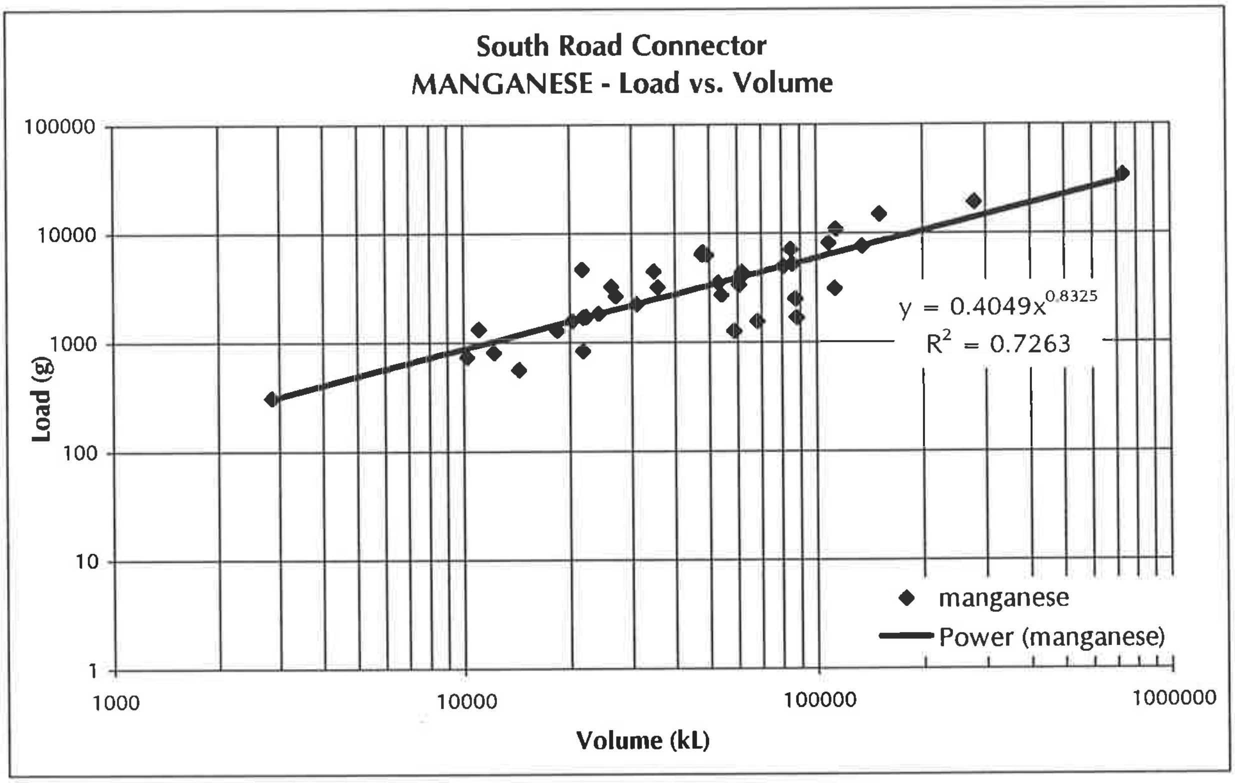

South Road Connector
CADMIUM - Load vs. Volume
y = 0.0004x^0.9885
R² = 0.7305
Load (g)
1000
100
10
1
Volume (kL)
1000
10000
100000
1000000
◆ cadmium
Power (cadmium)

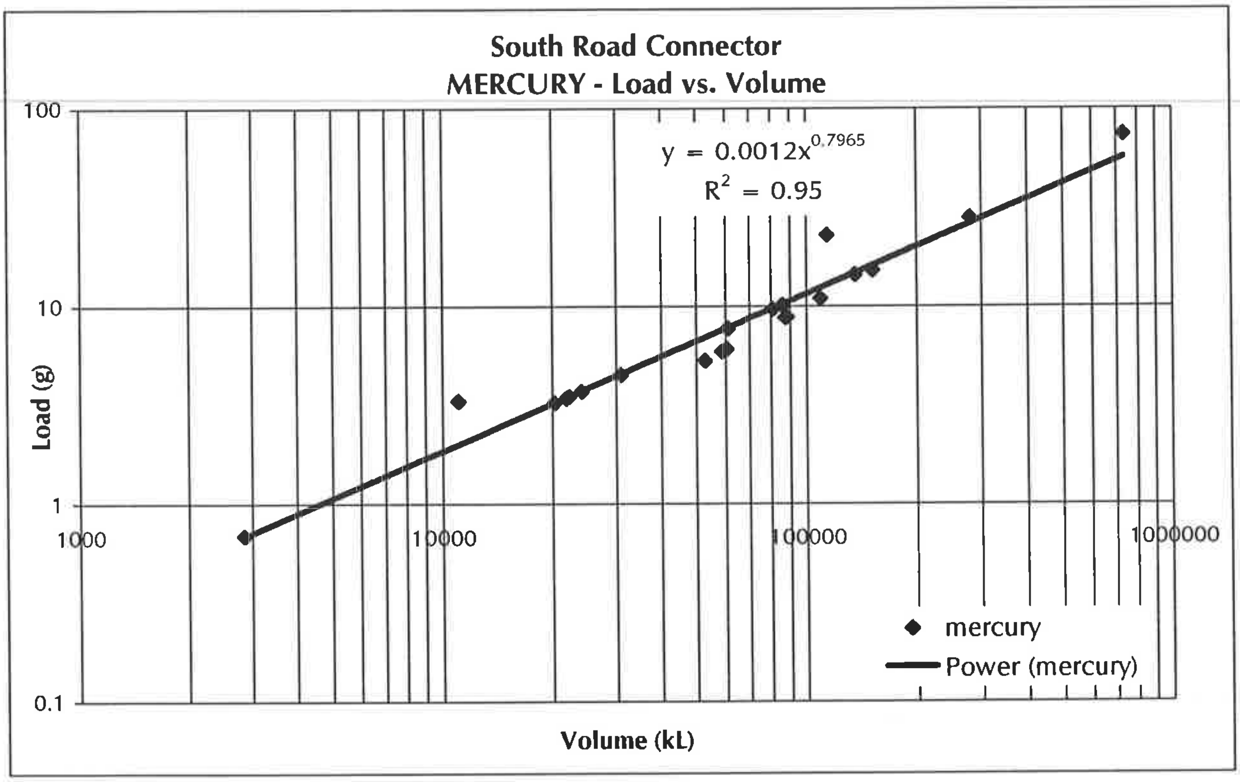

South Road Connector
CHROMIUM - Load vs. Volume
10000
1000
100
10
1
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.0399x^0.8577
R² = 0.8826
chromium
Power (chromium)

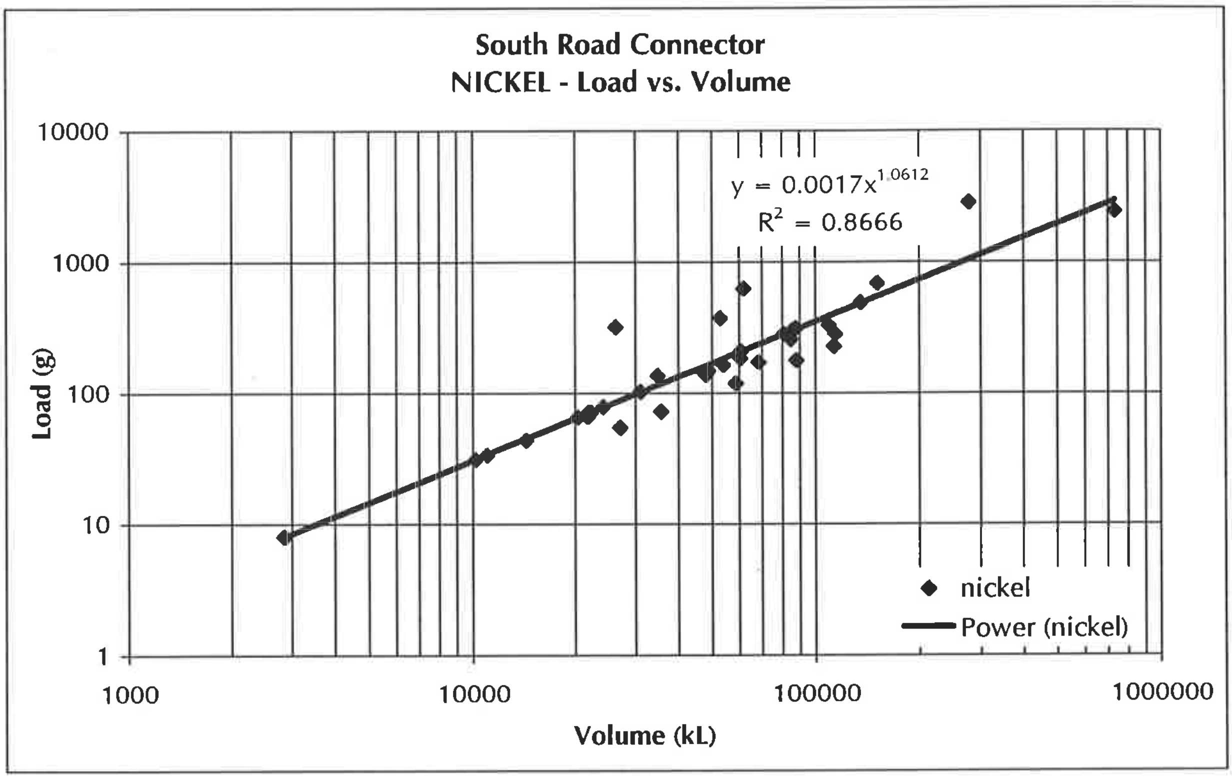

South Road Connector
COPPER - Load vs. Volume
100000
10000
1000
100
10
1
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.0755x^0.847
R² = 0.7829
copper
Power (copper)

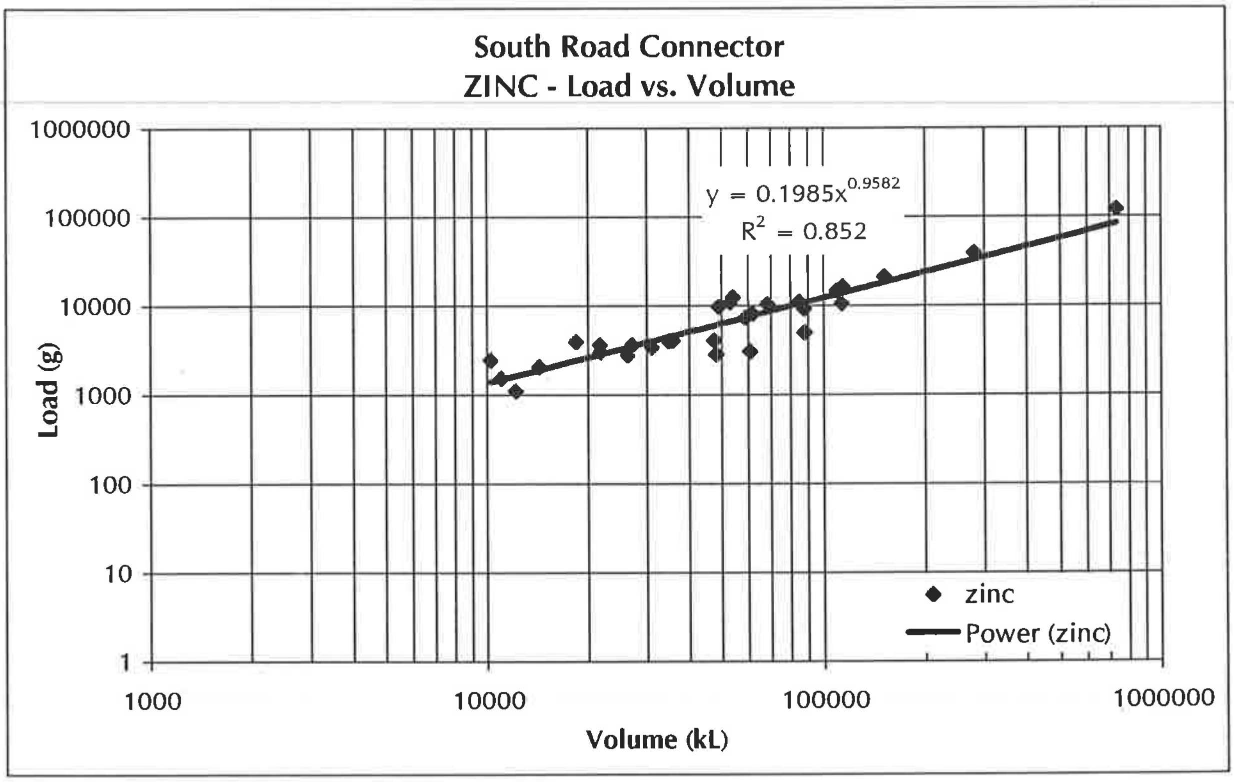

South Road Connector
IRON - Load vs. Volume
Load (g)
10000000
1000000
100000
10000
1000
100
10
1
Volume (kL)
1000
10000
100000
1000000
y = 1.2485x^0.9625
R^2 = 0.7634
iron
Power (iron)

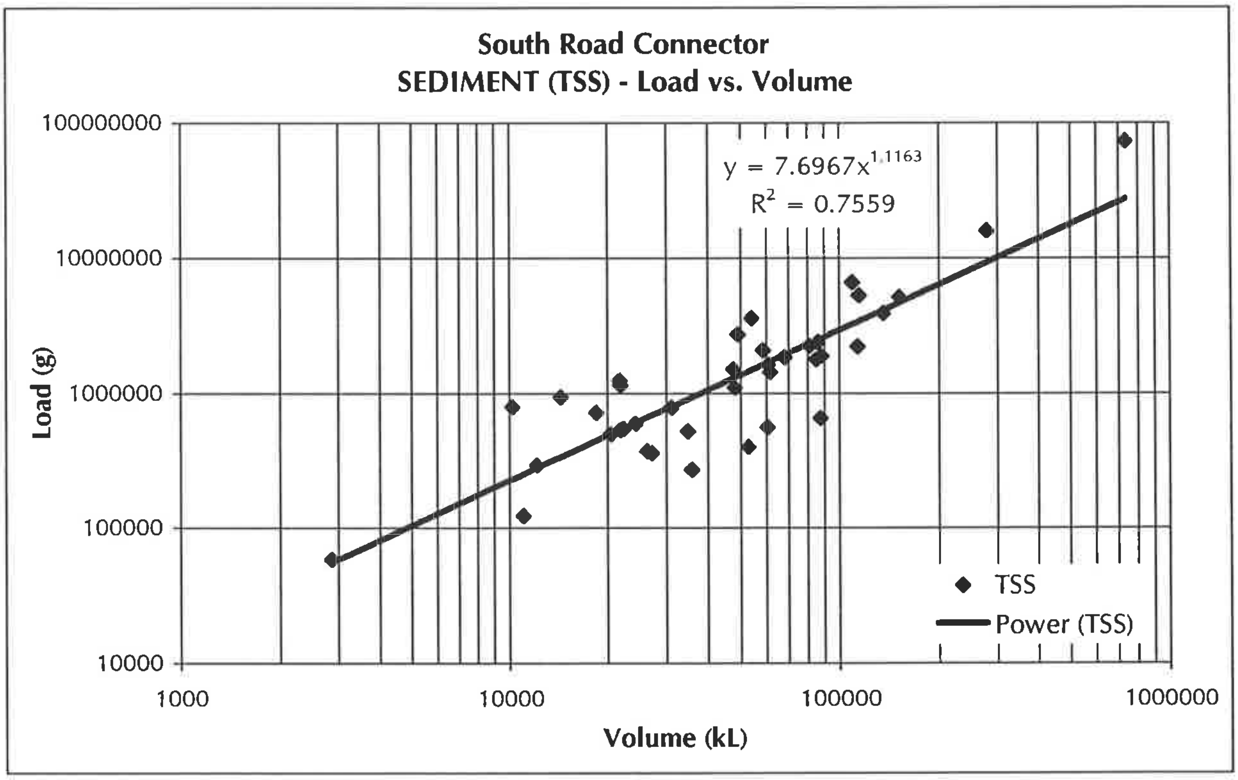

South Road Connector
LEAD - Load vs. Volume
Load (g)
100000
10000
1000
100
10
1
Volume (kL)
1000
10000
100000
1000000
y = 0.0124x^1.015
R^2 = 0.6756
lead
Power (lead)


South Road Connector
MANGANESE - Load vs. Volume
100000
10000
1000
100
10
1
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.4049x^0.8325
R² = 0.7263
manganese
Power (manganese)


South Road Connector
MERCURY - Load vs. Volume
100
10
1
0.1
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.0012x^0.7965
R² = 0.95
mercury
Power (mercury)


South Road Connector
NICKEL - Load vs. Volume
10000
1000
100
10
1
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.0017x^1.0612
R² = 0.8666
◆ nickel
Power (nickel)


South Road Connector
ZINC - Load vs. Volume
1000000
100000
10000
1000
100
10
1
Load (g)
1000
10000
100000
1000000
Volume (kL)
y = 0.1985x^0.9582
R² = 0.852
◆ zinc
Power (zinc)

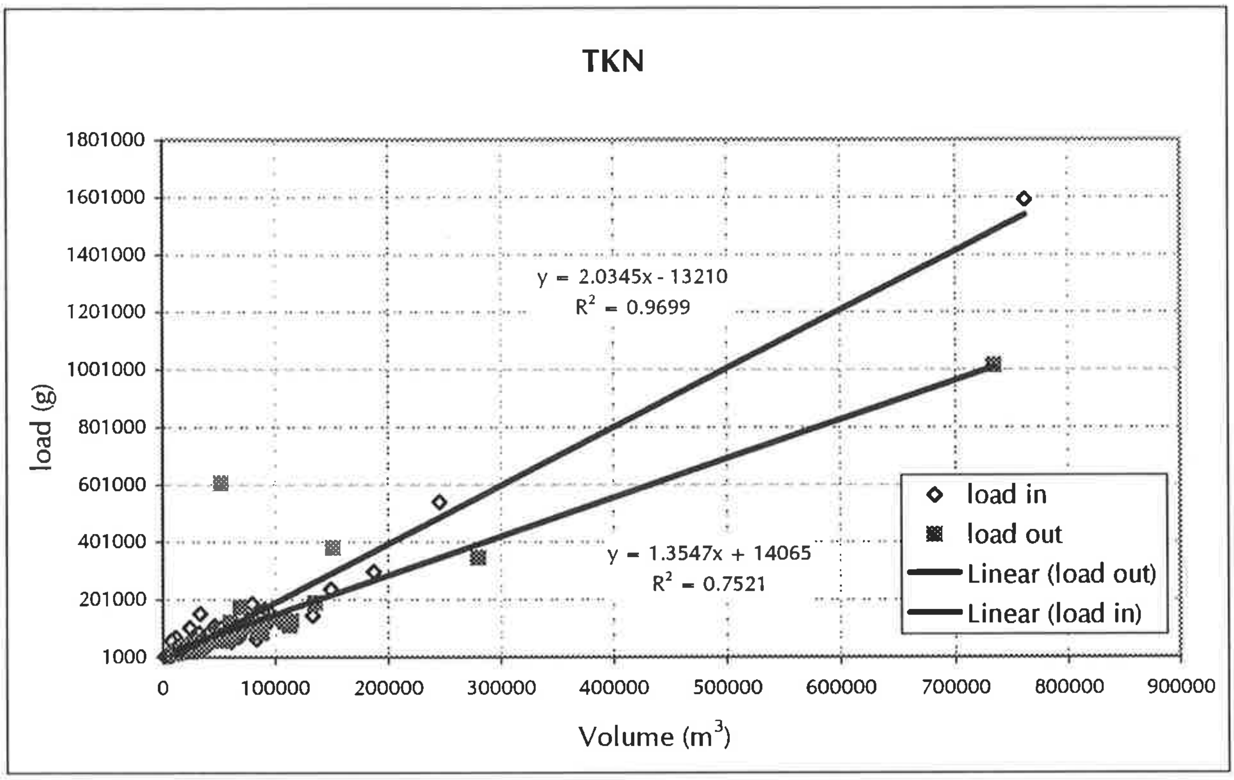

South Road Connector
SEDIMENT (TSS) - Load vs. Volume
y = 7.6967x^1.1163
R² = 0.7559
Load (g)
100000000
10000000
1000000
100000
10000
Volume (kL)
1000
10000
100000
1000000
TSS
Power (TSS)

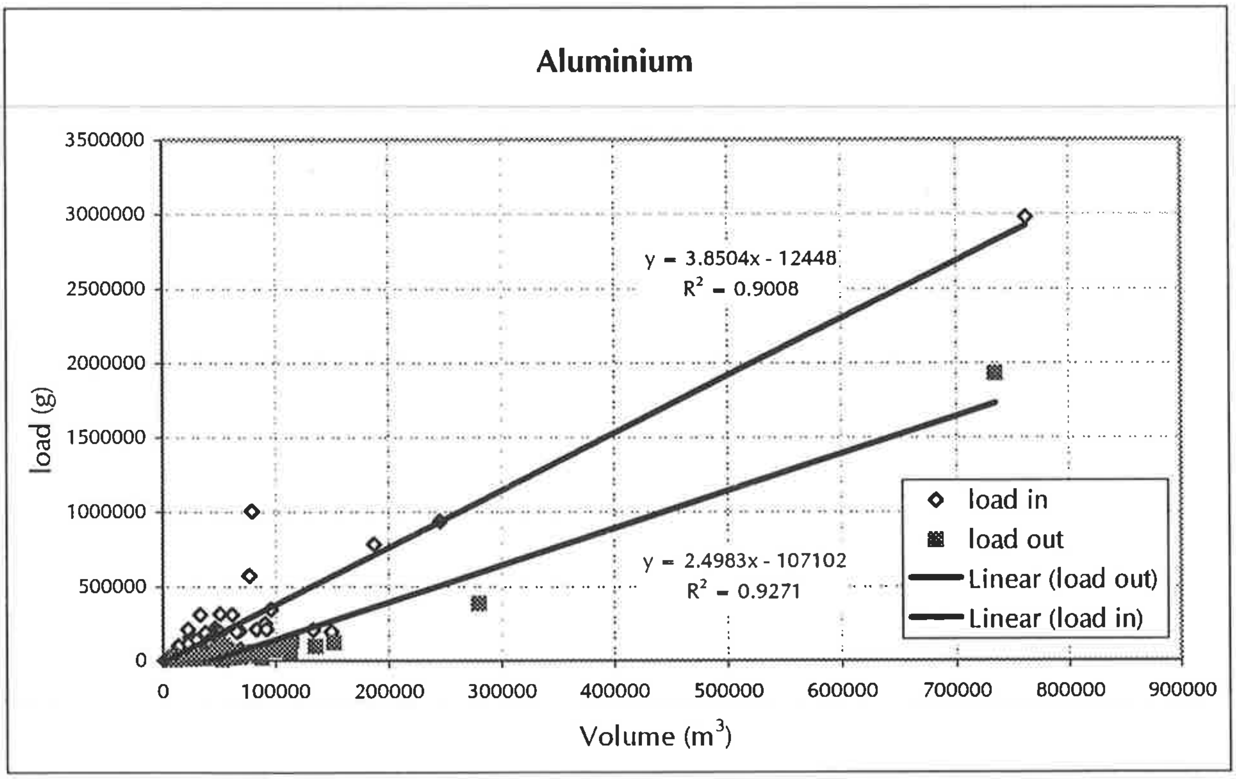
Appendix F

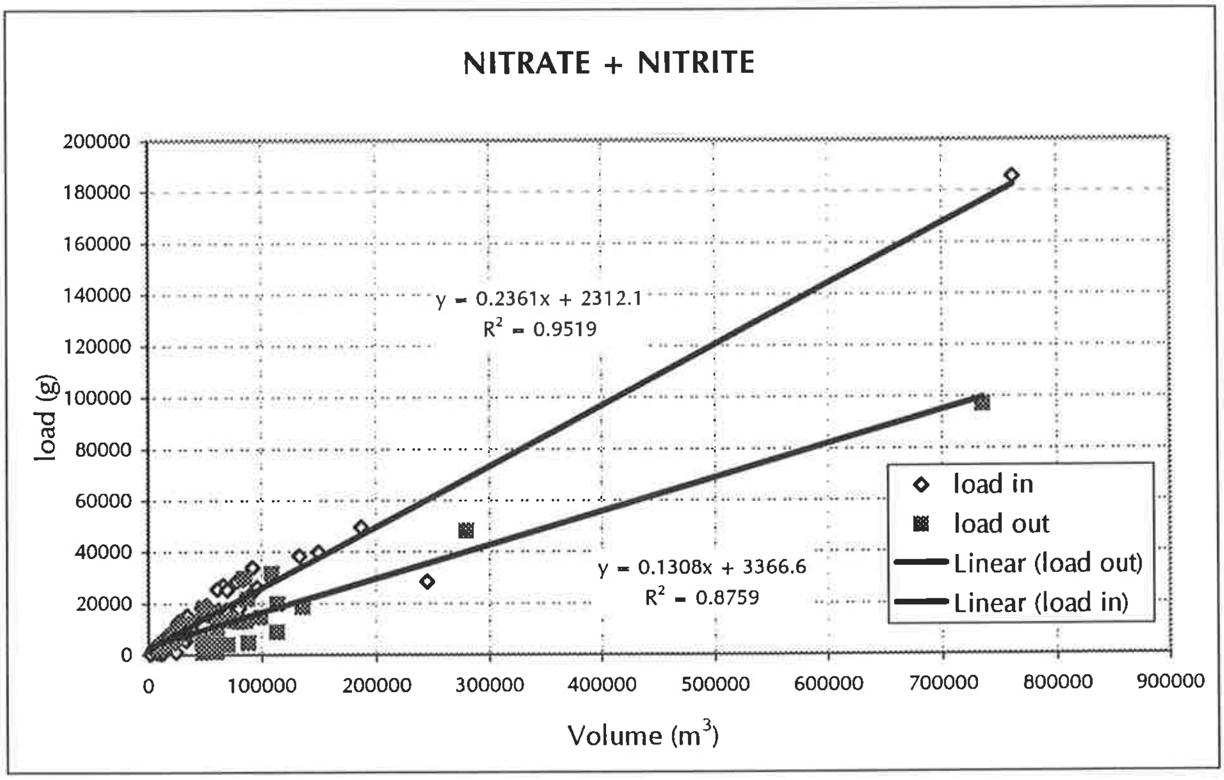
BARKER INLET WETLAND POND 4 - LOAD IN & LOAD OUT AS A FUNCTION OF EVENT SIZE

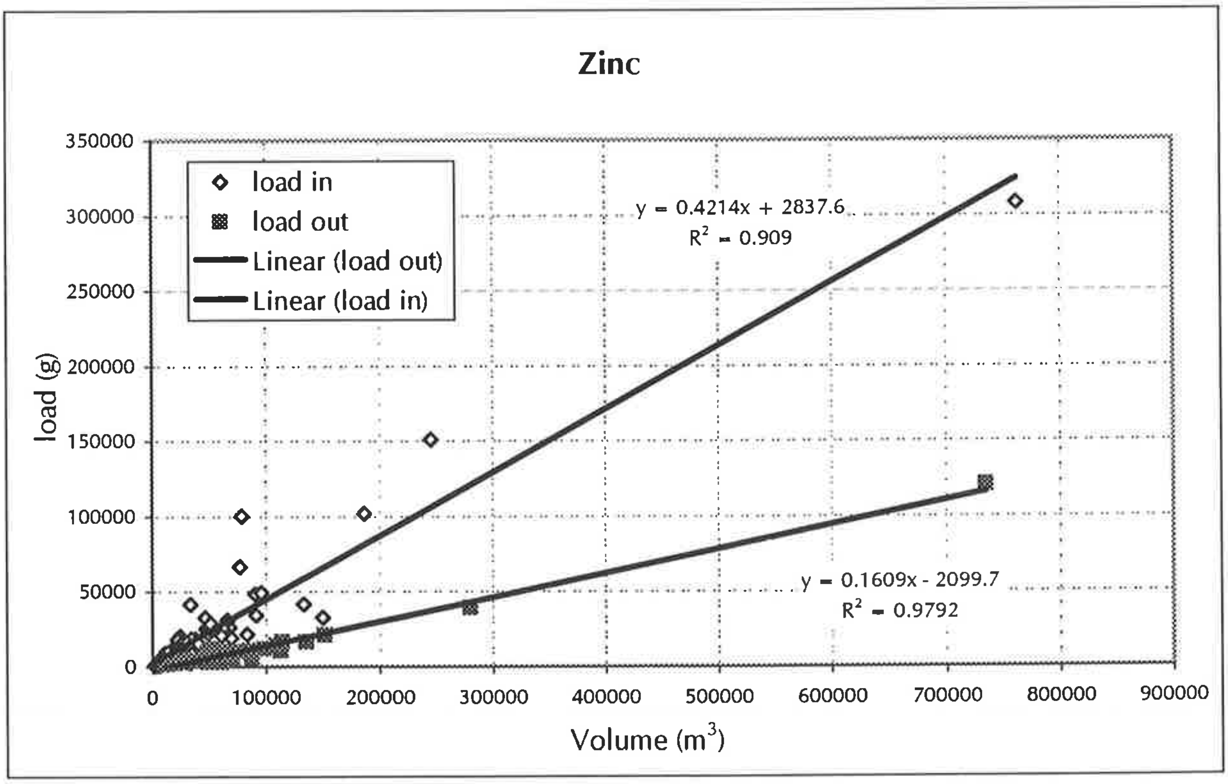
Contents:

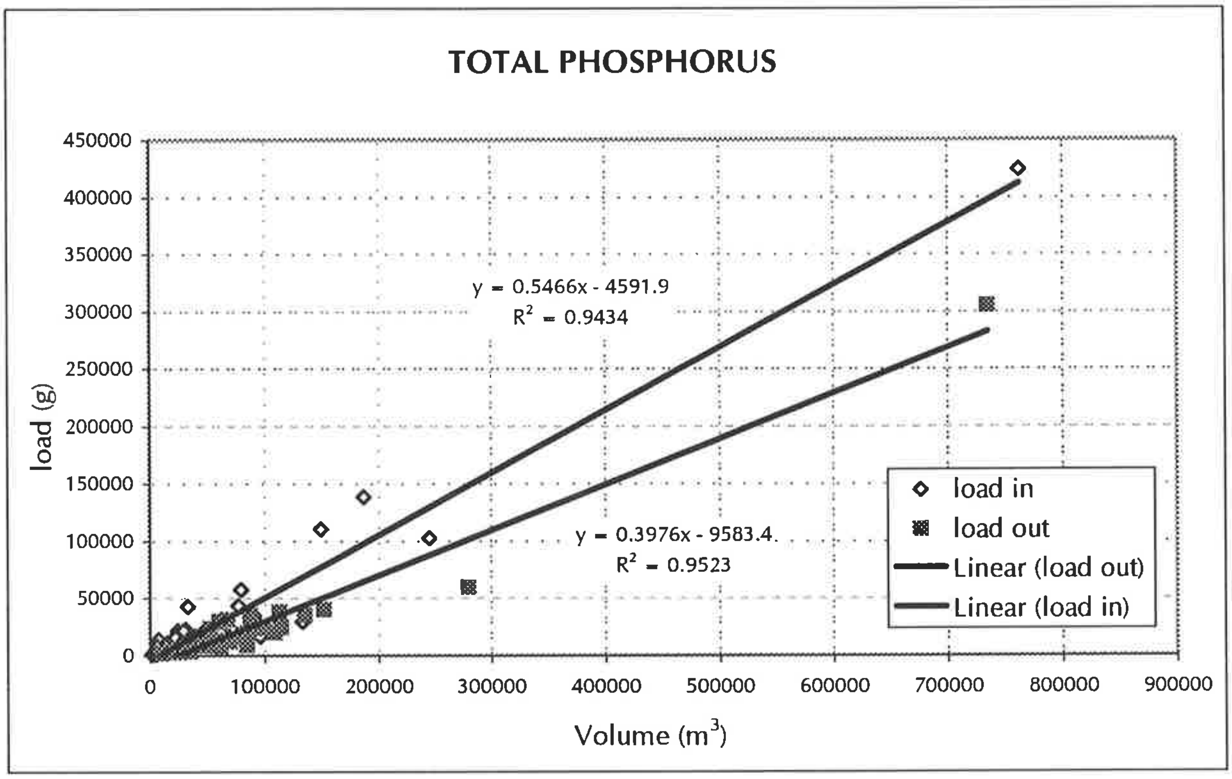
Plots showing load and load out as a function of event size, all parameters

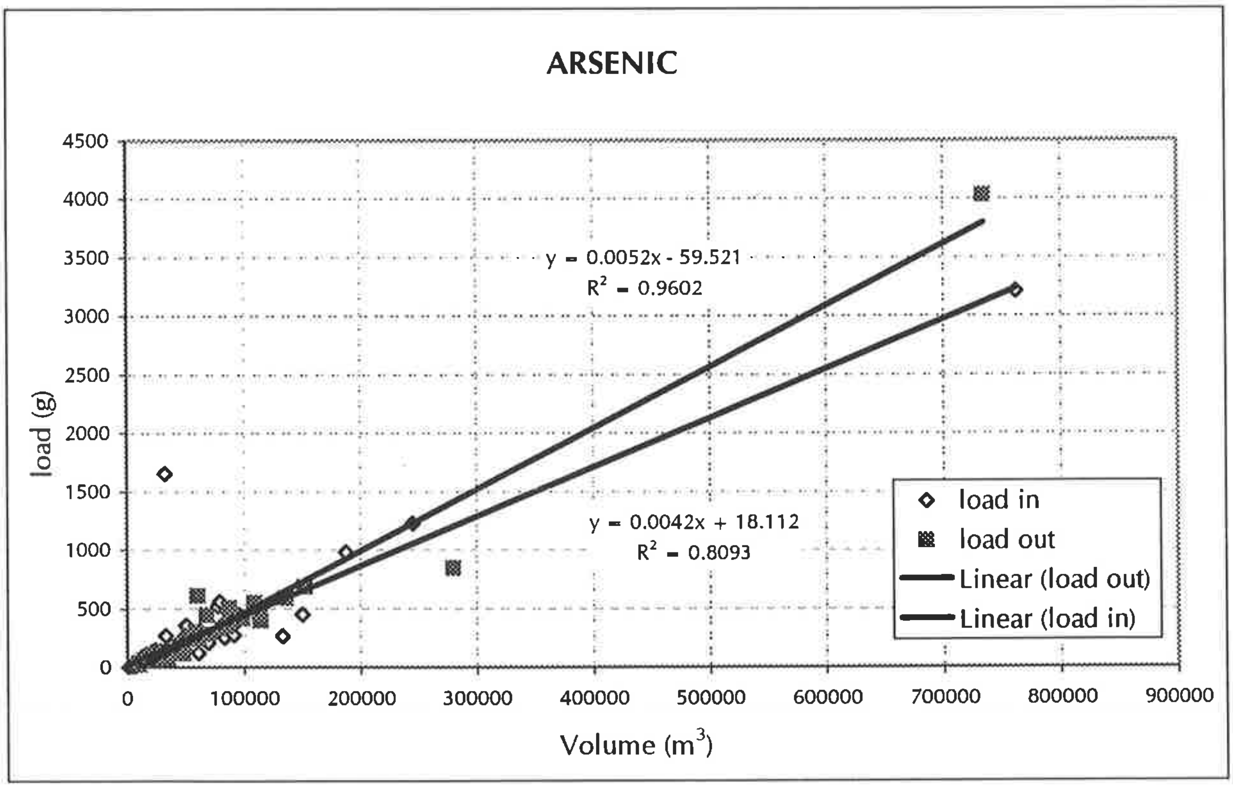

TKN
1801000
1601000
1401000
1201000
1001000
801000
601000
401000
201000
1000
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m^3)
y = 2.0345x - 13210
R^2 = 0.9699
y = 1.3547x + 14065
R^2 = 0.7521
load in
load out
Linear (load out)
Linear (load in)

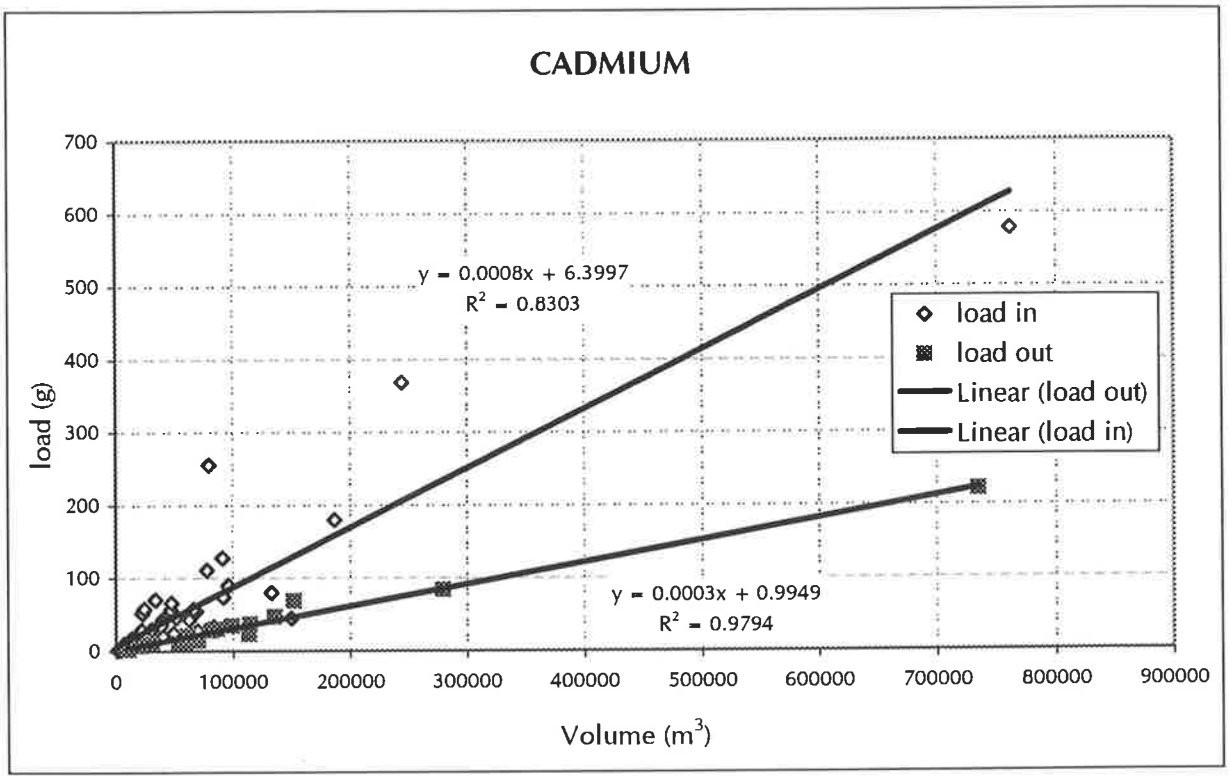

Aluminium
3500000
3000000
2500000
2000000
1500000
1000000
500000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m^3)
y = 3.8504x - 12448
R^2 = 0.9008
y = 2.4983x - 107102
R^2 = 0.9271
load in
load out
Linear (load out)
Linear (load in)

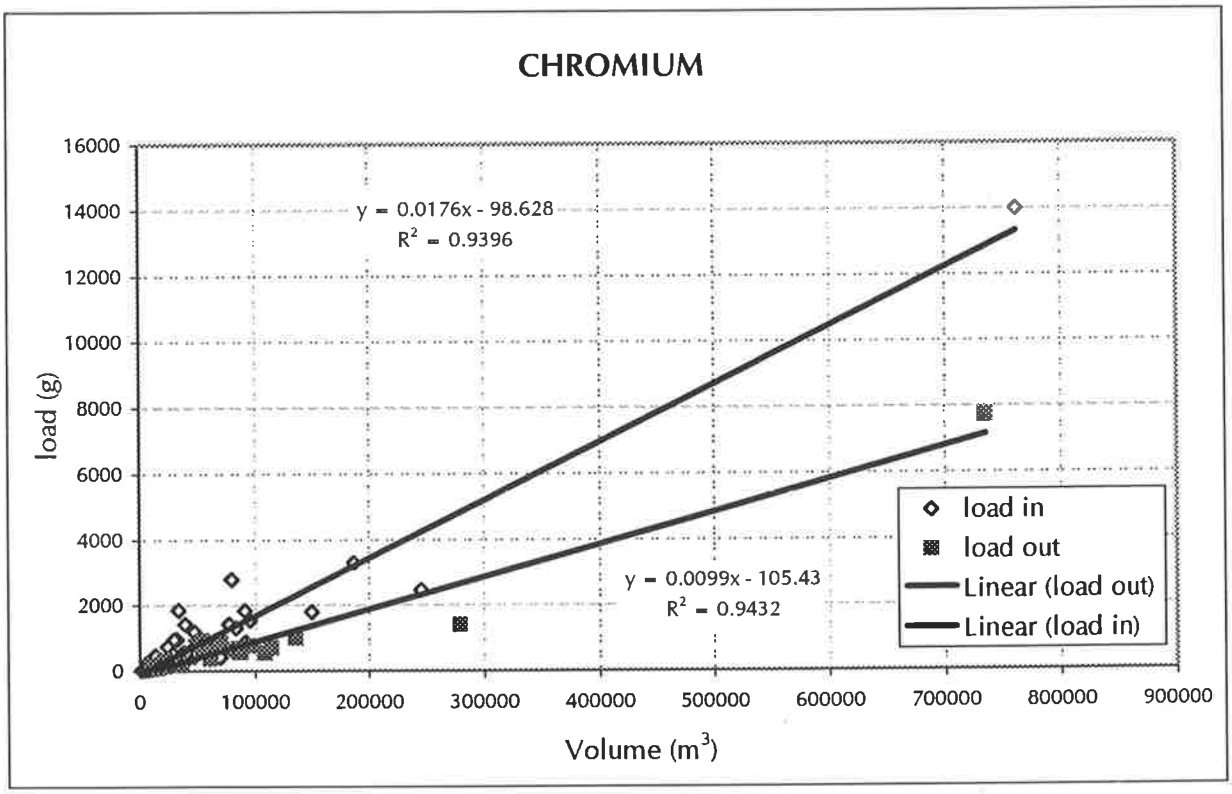

NITRATE + NITRITE
load (g)
200000
180000
160000
140000
120000
100000
80000
60000
40000
20000
0
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m^3)
y = 0.2361x + 2312.1
R^2 = 0.9519
y = 0.1308x + 3366.6
R^2 = 0.8759
load in
load out
Linear (load out)
Linear (load in)

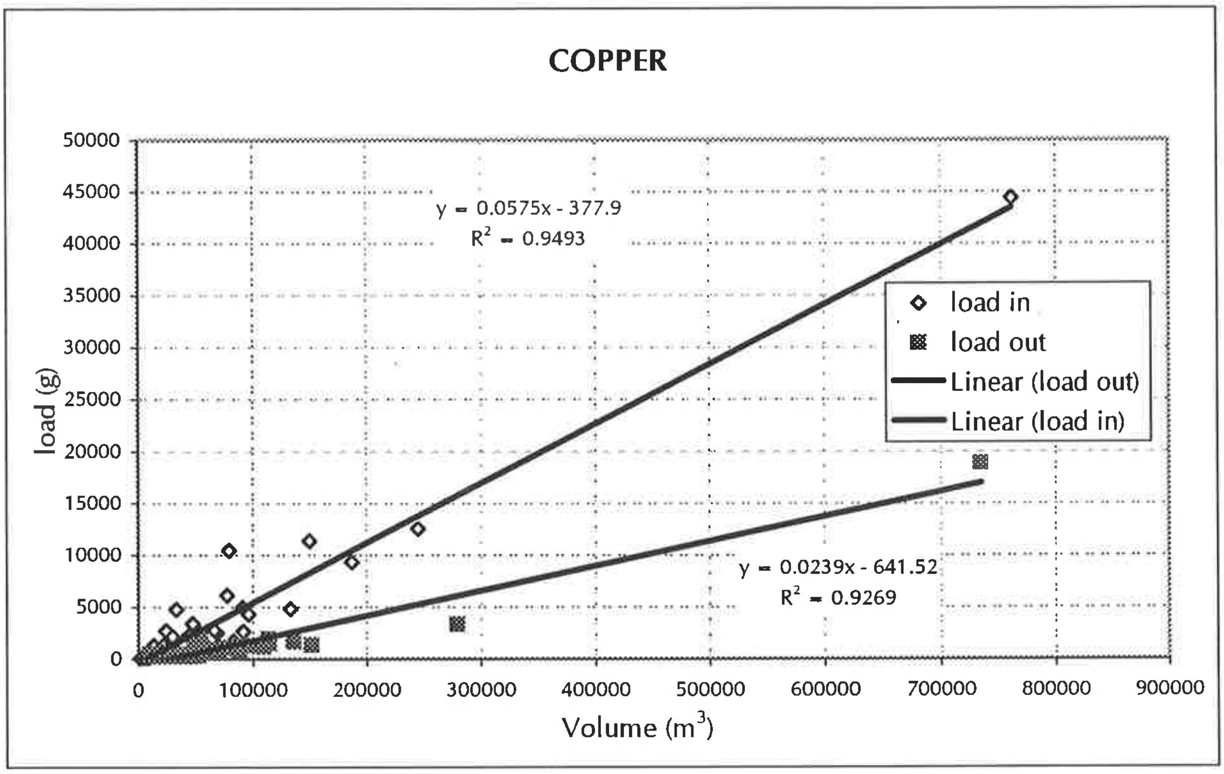

Zinc
load (g)
350000
300000
250000
200000
150000
100000
50000
0
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m^3)
load in
load out
Linear (load out)
Linear (load in)
y = 0.4214x + 2837.6
R^2 = 0.909
y = 0.1609x - 2099.7
R^2 = 0.9792

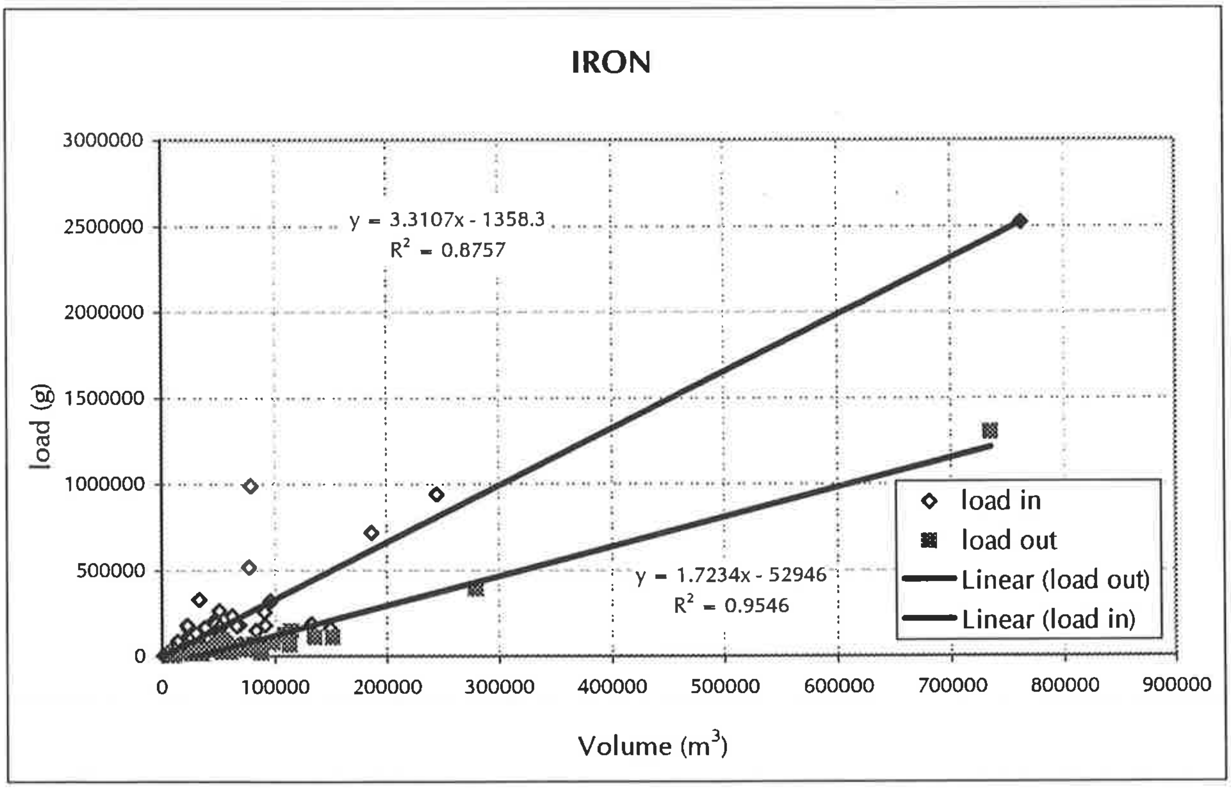

TOTAL PHOSPHORUS
450000
400000
350000
300000
250000
200000
150000
100000
50000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 0.5466x - 4591.9
R2 = 0.9434
y = 0.3976x - 9583.4
R2 = 0.9523
load in
load out
Linear (load out)
Linear (load in)

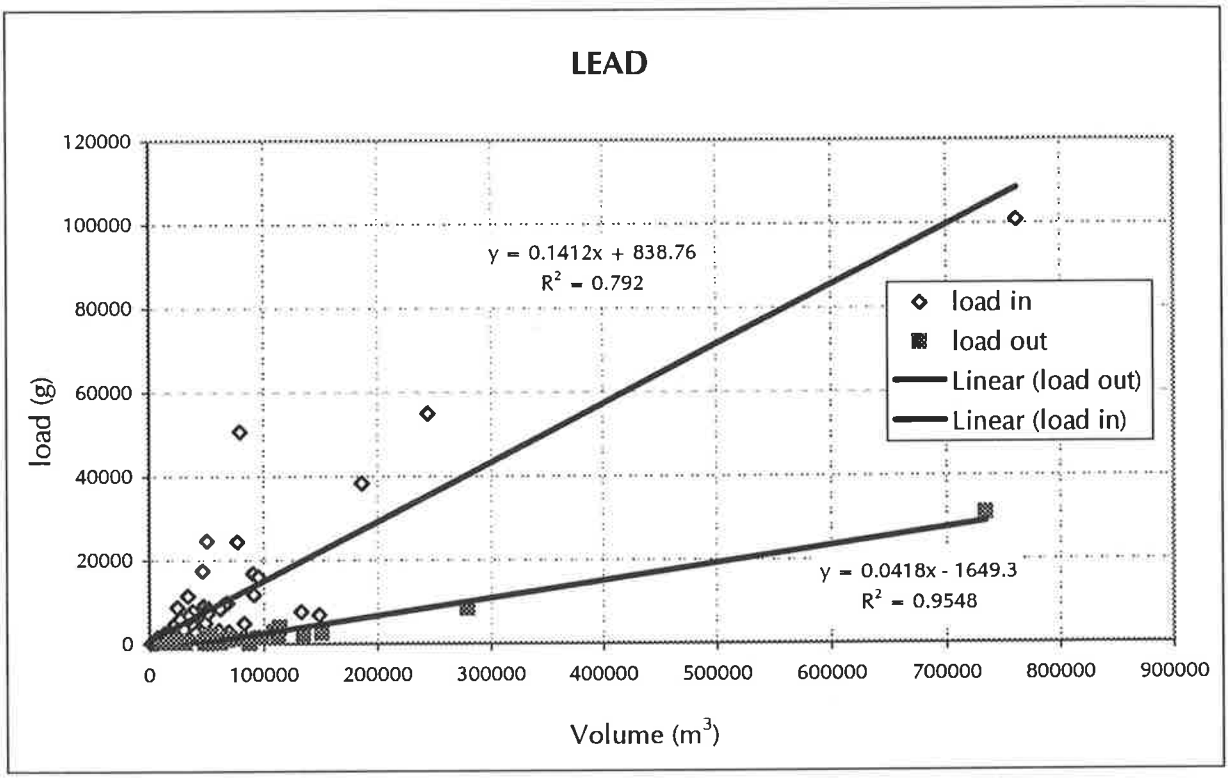

ARSENIC
4500
4000
3500
3000
2500
2000
1500
1000
500
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 0.0052x - 59.521
R2 = 0.9602
y = 0.0042x + 18.112
R2 = 0.8093
load in
load out
Linear (load out)
Linear (load in)

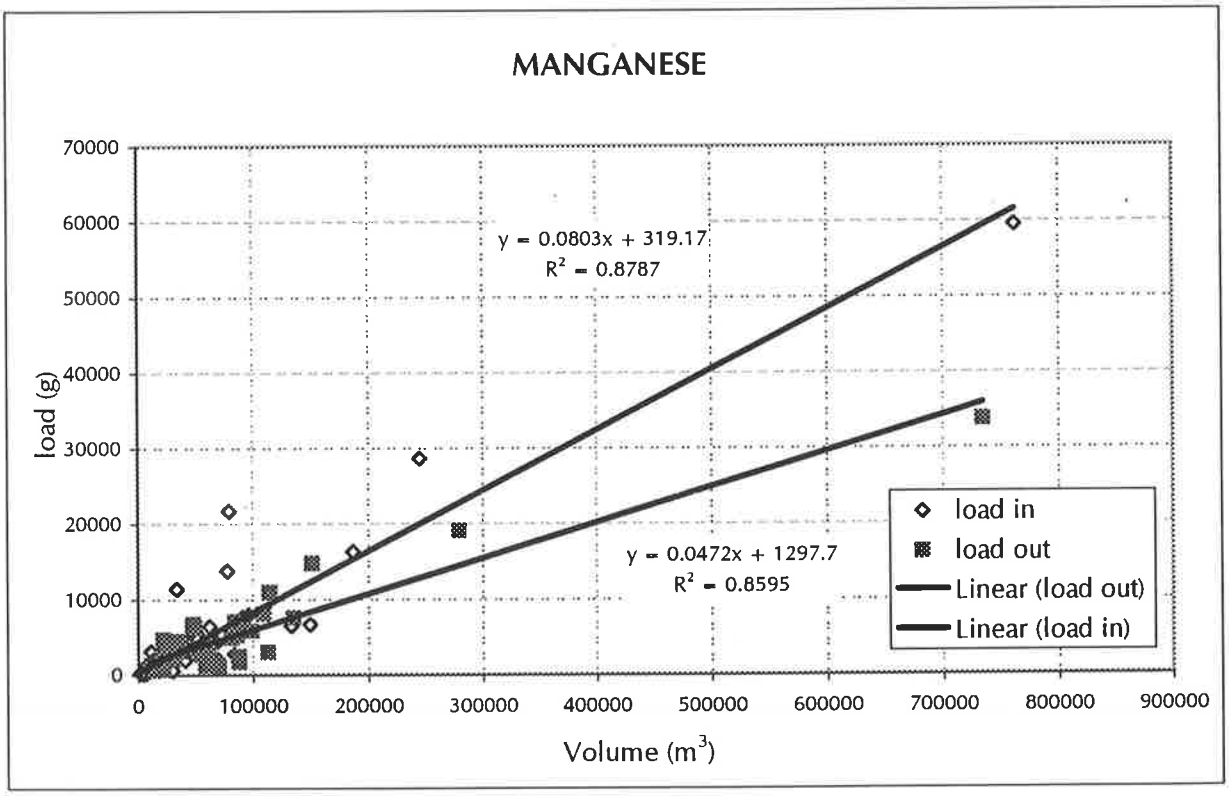

CADMIUM
700
600
500
400
300
200
100
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 0.0008x + 6.3997
R2 = 0.8303
y = 0.0003x + 0.9949
R2 = 0.9794
load in
load out
Linear (load out)
Linear (load in)

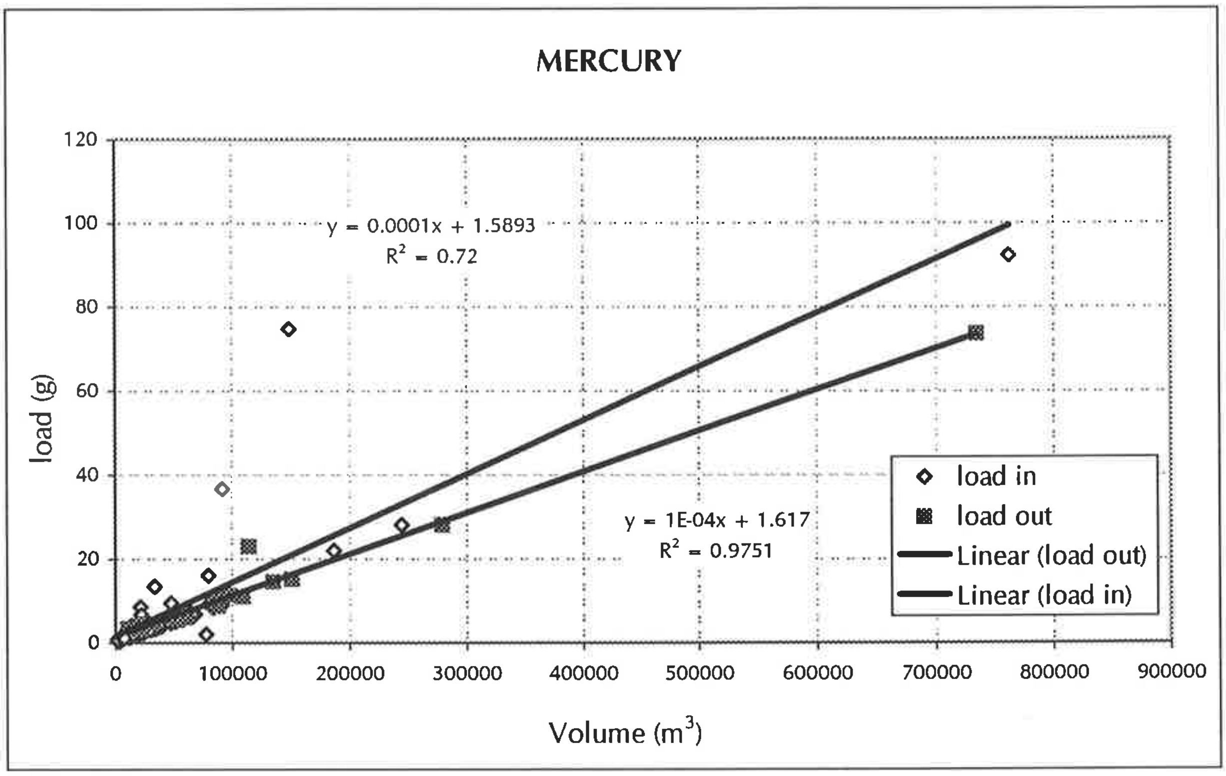

CHROMIUM
16000
14000
12000
10000
8000
6000
4000
2000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 0.0176x - 98.628
R2 = 0.9396
y = 0.0099x - 105.43
R2 = 0.9432
load in
load out
Linear (load out)
Linear (load in)

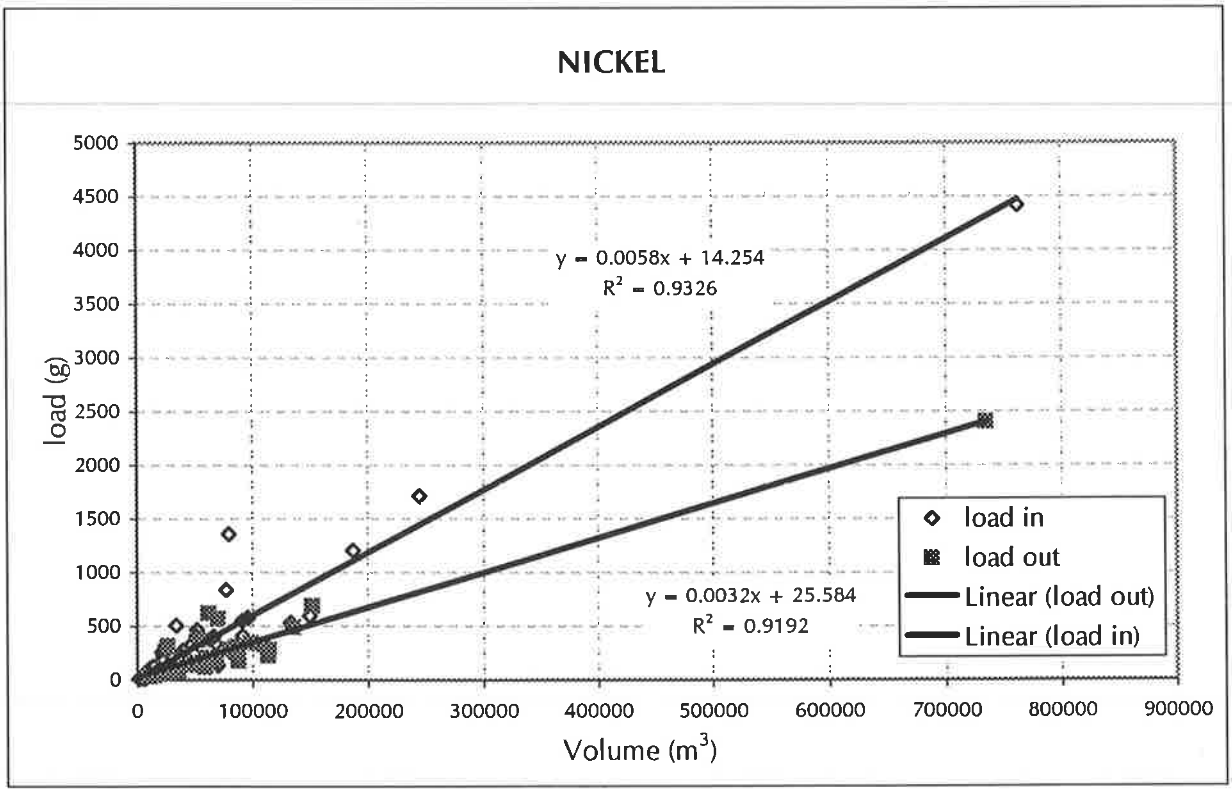

COPPER
50000
45000
40000
35000
30000
25000
20000
15000
10000
5000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 0.0575x - 377.9
R2 = 0.9493
y = 0.0239x - 641.52
R2 = 0.9269
load in
load out
Linear (load out)
Linear (load in)

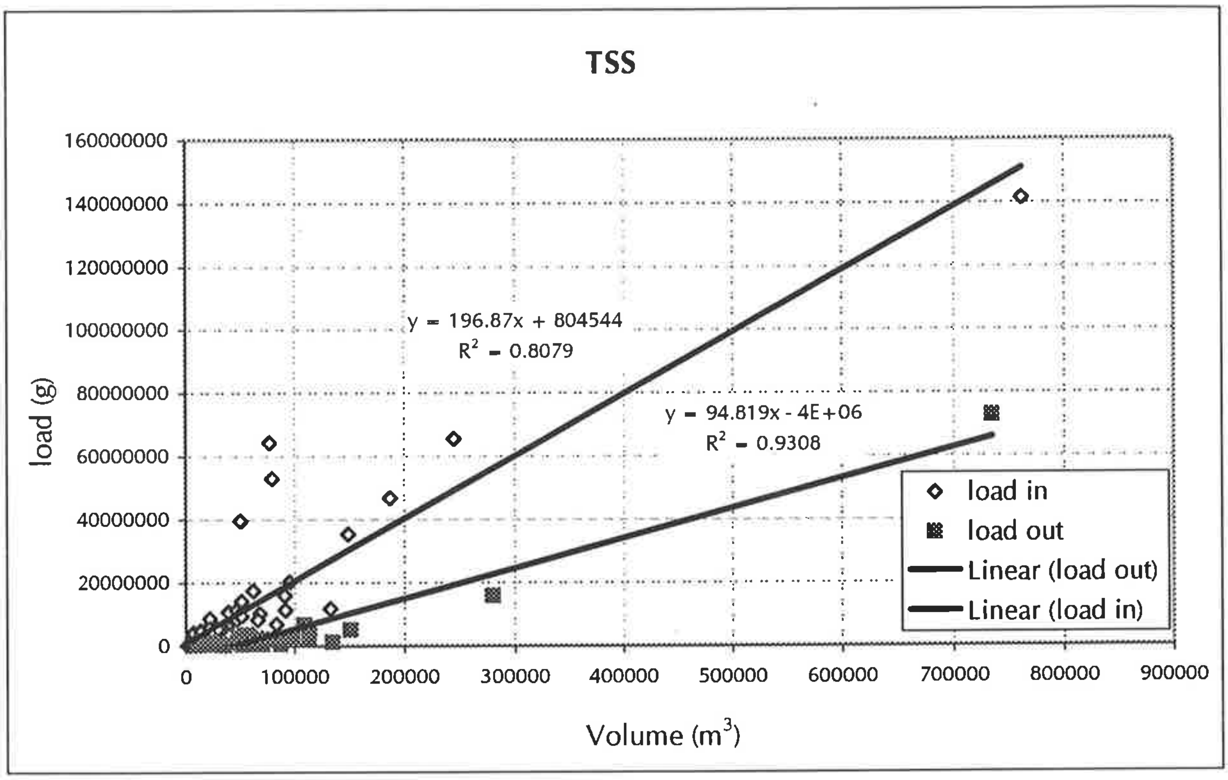

IRON
3000000
2500000
2000000
1500000
1000000
500000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 3.3107x - 1358.3
R2 = 0.8757
y = 1.7234x - 52946
R2 = 0.9546
load in
load out
Linear (load out)
Linear (load in)

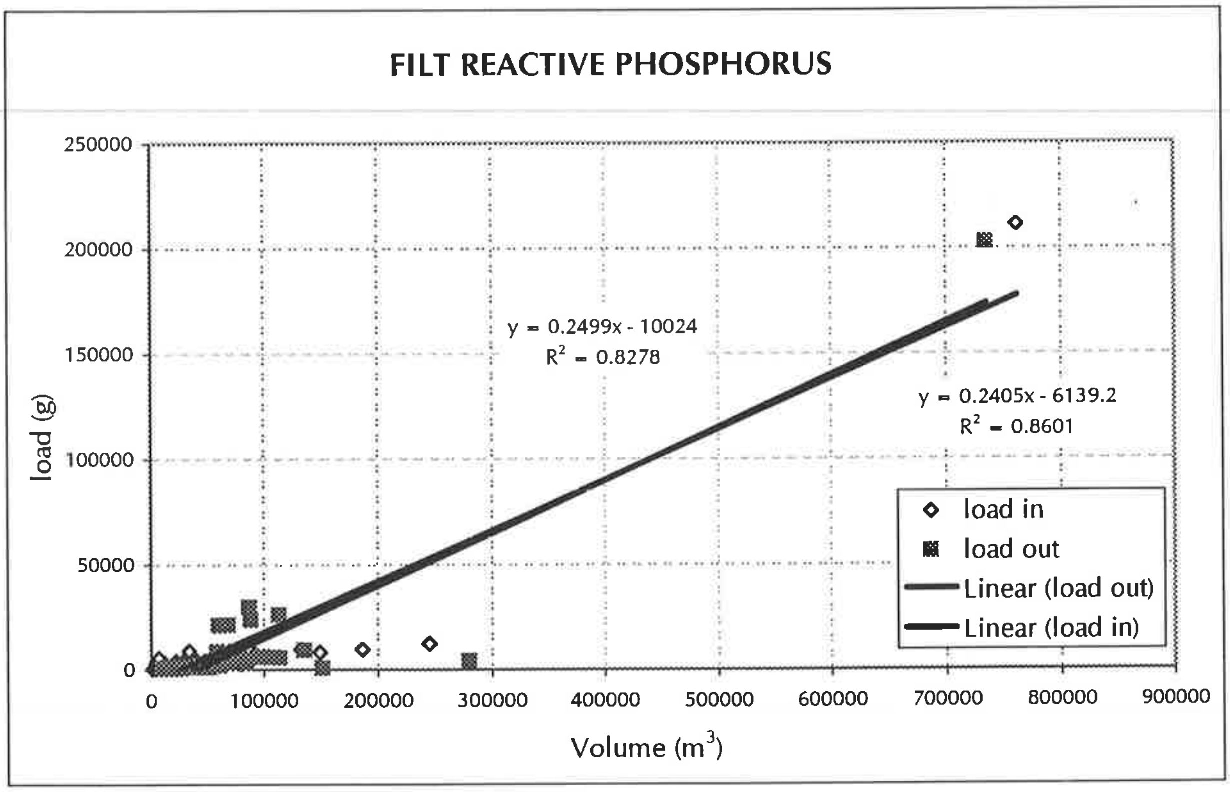

LEAD
120000
100000
80000
60000
40000
20000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m^3)
y = 0.1412x + 838.76
R^2 = 0.792
y = 0.0418x - 1649.3
R^2 = 0.9548
◇ load in
▨ load out
Linear (load out)
Linear (load in)

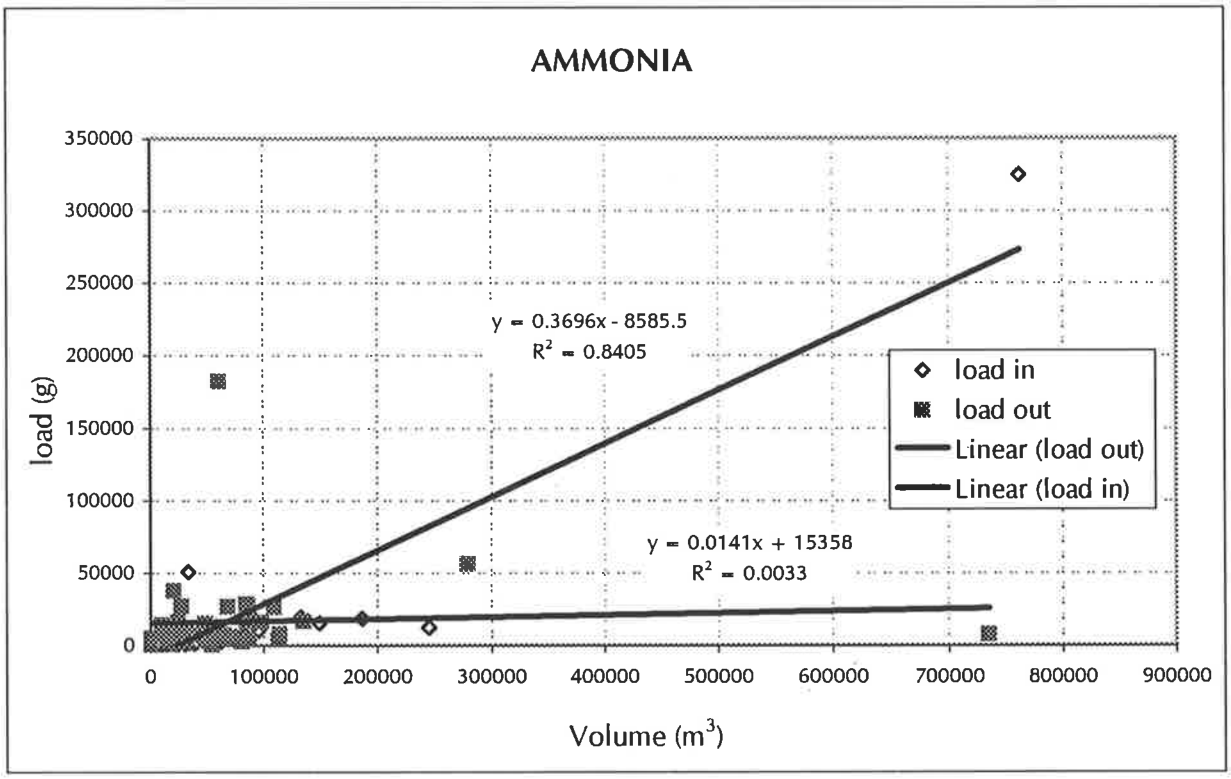

MANGANESE
70000
60000
50000
40000
30000
20000
10000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m^3)
y = 0.0803x + 319.17
R^2 = 0.8787
y = 0.0472x + 1297.7
R^2 = 0.8595
◇ load in
▨ load out
Linear (load out)
Linear (load in)


MERCURY
120
100
80
60
40
20
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 0.0001x + 1.5893
R2 = 0.72
y = 1E-04x + 1.617
R2 = 0.9751
load in
load out
Linear (load out)
Linear (load in)


NICKEL
5000
4500
4000
3500
3000
2500
2000
1500
1000
500
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 0.0058x + 14.254
R2 = 0.9326
y = 0.0032x + 25.584
R2 = 0.9192
load in
load out
Linear (load out)
Linear (load in)


TSS
160000000
140000000
120000000
100000000
80000000
60000000
40000000
20000000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 196.87x + 804544
R2 = 0.8079
y = 94.819x - 4E+06
R2 = 0.9308
load in
load out
Linear (load out)
Linear (load in)


FILT REACTIVE PHOSPHORUS
250000
200000
150000
100000
50000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m3)
y = 0.2499x - 10024
R2 = 0.8278
y = 0.2405x - 6139.2
R2 = 0.8601
load in
load out
Linear (load out)
Linear (load in)


AMMONIA
350000
300000
250000
200000
150000
100000
50000
0
load (g)
0
100000
200000
300000
400000
500000
600000
700000
800000
900000
Volume (m³)
y = 0.3696x - 8585.5
R² = 0.8405
y = 0.0141x + 15358
R² = 0.0033
load in
load out
Linear (load out)
Linear (load in)