

Technical Report

Adelaide
Coastal
Waters
Study



Stage 2 Research Program 2003 - 2005

Technical Report No. 18 July 2006

Volumes of inputs, their concentrations and loads received by Adelaide metropolitan coastal waters

Final Technical Report for ACWS Task IS 1



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Volumes of inputs, their concentrations and loads received by Adelaide metropolitan coastal waters

Author

Jeremy Wilkinson, Nicholas White, Lynn Smythe, John Hutson, Erick Bestland, Craig Simmons, Sebastien Lamontagne¹ and Howard Fallowfield.

Department of Environmental Health
Flinders University of SA
GPO Box 2100
Adelaide SA 5001

¹ CSIRO Land and Water
PMB 2 Glen Osmond,
South Australia 5064

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ISBN 1 876562 99 4

July 2006

Reference

This report can be cited as:

Wilkinson J, Nicholas White, Lynn Smythe, John Hutson, Erick Bestland, Craig Simmons, Sebastien Lamontagne and Howard Fallowfield. 2005. *Volumes of inputs, their concentrations and loads received by Adelaide metropolitan coastal waters*. ACWS Technical Report No. 18 prepared for the Adelaide Coastal Waters Study Steering Committee, September 2005. Flinders Centre for Coastal and Catchment Environments, Flinders University of SA



Acknowledgement

This report is a product of the Adelaide Coastal Waters Study. In preparing this report, the authors acknowledge the financial and other support provided by the ACWS Steering Committee including the South Australian Environment Protection Authority, SA Water Corporation, the Torrens Patawalonga and Onkaparinga Catchment Water Management Boards, Department for Transport Energy and Infrastructure, Mobil Refining Australia Pty Ltd, TRUenergy, Coast Protection Board and PIRSA. Non-funding ACWS Steering Committee members include the Conservation Council of SA, SA Fishing Industry Council Inc, Local Government Association, Department of Water Land and Biodiversity Conservation and Planning SA.



Acknowledgement

In addition, the authors would like to acknowledge the following individuals for their assistance with data or other invaluable material of knowledge: Lester Sickerdick and Nirmala Dinesh of SA Water. Bruce Nicholson, Pat Betham and Andrew Howith of Water Data Services. Steve Gatti of OCWMB, Peter Christy of EPA, Dennis Steffensen of AWQC., Peter Clemmett of BOM, Ken Schalk and Phil Crawley of BC Tonkins, Adelaide.

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1. Introduction

This is the final report of the ACWS Input Studies. The report provides answers to specific stakeholder questions. The report provides an overview of historical changes in terrestrial flows to sea from WWTP and stormwater sources. An examination of stormwater quality is presented and estimated loads are provided. Load estimates include overall mean annual loads from all sources, i.e. atmospheric, groundwater, point sources and stormwater. A breakdown of terrestrial loads by geographical location is provided and a comparison with historic loads is given. Seasonal variations in nutrient concentrations, loads and yields are provided. For suspended solids and heavy metals (Cu, Pb and Zn), only seasonal loads and yields are given.

Load estimates and load discharge relationships produced by this study feed into the coastal circulation modelling carried-out by UWA, and all data used and produced by the Input Studies are available as a research archive accessible to future workers in this field.

The TR1, TR3 and TR10 reports are companion reports to the final report and provide additional background to WWTP loads and water quality, landuse and hydrology for stormwaters, and reconstruction of historical flows. Report TR4 gives a thorough presentation of the groundwater investigation.

The final Chapter of this report presents population, flow and seagrass die-back data in relation to major input events along the Metropolitan Adelaide Coastline.

The first task of the report is to answer the stakeholder questions. These are presented below and act as an executive summary.

1.1 The ACWS and Input Studies

The objective of the ACWS is to develop understanding and tools to enable sustainable management of Adelaide’s coastal waters by identifying causes of ecosystem modifications and the actions required to halt and reverse degradation. The study focussed on:

- Seagrass loss
- Seabed instability
- Water quality degradation

The main outcome areas of the study are to provide new knowledge and information, options for management actions, a program to assess effectiveness of management actions (including monitoring program), and to communicate the results to stakeholders.

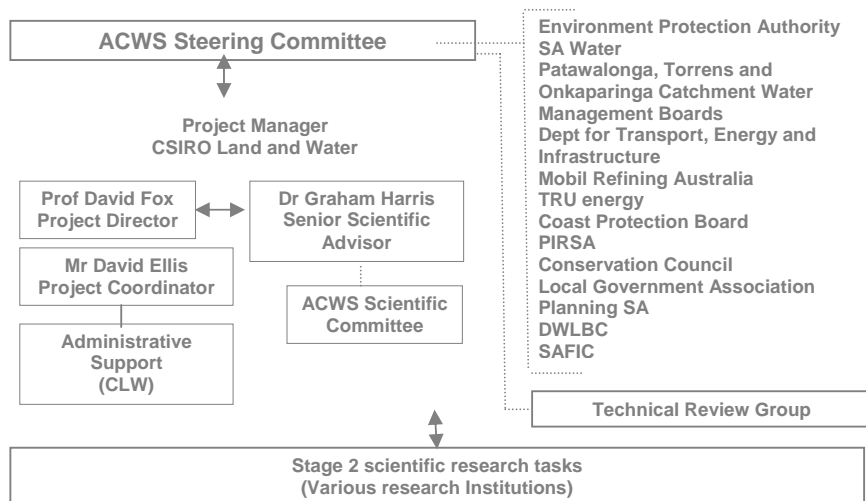


Figure 1.1. Management structure of the ACWS

Figure 1.1 presents a relationship diagram of the management structure of the ACWS, the right hand text box details stakeholder organizations.

The project has three stages:

- Stage 1 – identification of stakeholder requirements, project scope and clearly defined research study tasks for Stage 2
- Stage 2 - project management of specific research tasks, and
- Stage 3 - synthesis and reporting of results

The stakeholder issue themes that were identified during Stage one of the study are:

- Water quality (objectives / standards / assimilative capacity and maximum contaminant loads)
- Recreational water quality / environmental health
- Disturbance indicators
- Coastal processes and sediments
- Marine habitats – status and protection of reefs / estuaries / inlets / wetlands / mangroves

The study structure and framework is presented in Figure 1.2. Essentially the study investigates the input disturbances to the coastal system. The input information is used to drive the ecological processes (EP 1) studies of sea-grass responses and as input to the hydrodynamic modelling (PPM 1) used to determine the transport and fate of contaminants in relation to tidal, current and receiving water behaviour. The findings of the hydrodynamic and ecological studies, in turn, feed into the design of an adaptive management framework. This is intended to assist planners and managers with decisions about future amelioration strategies and with issues concerning development impacts on the coastal environment.

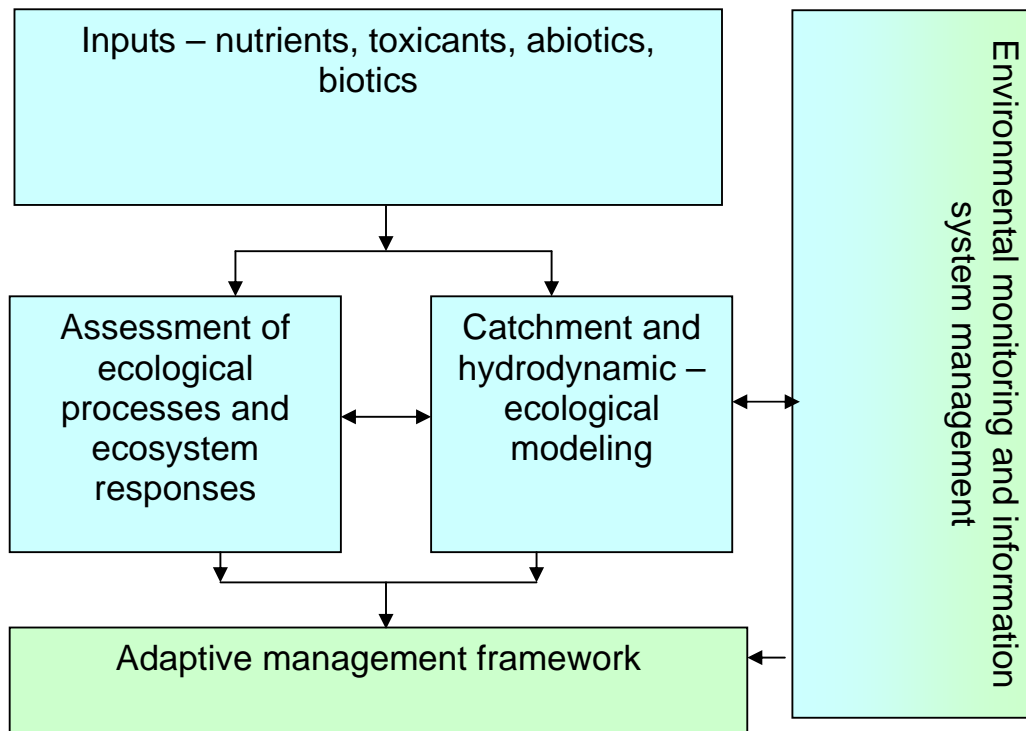


Figure 1.2. Study design of the ACWS.

This report presents the findings of the Input Studies (IS 1). IS 1 is one of the six research tasks detailed below:

- IS 1 Quantification of diffuse and point source terrestrial inputs
- EP 1 Assessment of the effects of inputs to the Adelaide coastal waters on seagrass ecosystems and key biota
- RS 1 Remote sensing study of marine and coastal features and interpretation of changes in relation to natural and anthropogenic processes
- PPM 1 Coastal sediment budget
- PPM 2 Physical oceanographic studies in the Adelaide coastal waters using in-situ observations, high-resolution modelling and satellite techniques
- EMP 1 Environmental Monitoring Program – spatial/temporal design; statistical analysis; QA/QC.

IS 1 – The Input Studies

IS 1 was an integrated program of studies designed to determine both the quantity and quality of stormwater, wastewater, groundwater and atmospheric fall-out entering the coastal Study area.

This study comprised four sub-programs:

- Sub-program 1: “Stormwater Flows from Major and Minor Catchments: Audit and Monitoring”;
- Sub-program 2: “Audit of the quality and quantity of effluent discharging from wastewater treatment plants (WWTPs) into the marine environment”;
- Sub-program 3: “Groundwater discharge to the coastal environment: Flow quantity and quality assessment”
- Sub-program 4: “Wetfall and dryfall input directly into the coastal zone”.

The task was initiated by an intensive literature and data review (Wilkinson et Al., 2003; 2005) that built upon the substantial review work commenced during Stage 1 of the Study by and for a number of organisations represented on the ACWS Scientific Committee. Available data were further compiled, interpreted and presented for incorporation and use in the ecological assessment and hydrodynamic modelling tasks.

The literature and data review identified information gaps and inconsistencies and allowed a program of research to be identified, approved and implemented to fill these gaps.

The objectives of the four sub-programs are set out below:

IS 1 SP1 – Stormwater flows: A thorough audit and compilation of existing data was carried-out for current and historical data on major catchments, minor catchments and stormwater drains, in order to identify gaps and determine the quality of these data. It was anticipated that there would be many gaps in the present data, particularly with respect to the minor catchments and drainages discharging directly into the coastal zone. Following the audit, where required selected catchments and stormwater drains were monitored and sampled.

Stormwater flows were monitored, sampled, and analysed for their loads of nutrients, inorganic species, suspended solids (sediment load) and heavy metals. This was done in order to contribute to the understanding of the relative importance of various diffuse sources of pollutant and nutrient load to the coastal environment. It was intended that sediment samples from Gulf St Vincent would be analysed for the same suite of indicators as stormwater sediments, this was to be coordinated with the sediment mapping and characterisation program, but did not go ahead due to an administrative oversight.

IS 1 SP2 - WWTP Inputs: to the objectives were:]

- to compile existing data on the quality and quantity of effluent discharging into marine and riverine environments from Adelaide's WWTPs;
- to analyse and interpret existing data on the quality and quantity of effluent and to develop relationships between flow and loads discharging into marine and riverine environments from Adelaide's WWTPs;
- to identify the constituents that are currently not being monitored in WWTP discharges and that may be of concern to seagrass and marine health (i.e. identify gaps in the current WWTP monitoring program);
- to provide details of WWTP nutrient loads to the nitrogen budget project and to develop relationships between flow and loads;
- based on results of efforts to identify additional constituents of concern to seagrass that are not currently monitored, this task may inform and direct a review and possible expansion of the existing WWTP monitoring program.

The outcomes of IS 1-SP2 are written-up in the ACWS Technical Report 1 (Wilkinson et al., 2003) and in part summarised in this report which provides a comparison of the relative magnitude of inputs.

IS 1 SP3 - Groundwater Inputs: The objectives of this task were:

- to determine the likely groundwater discharge (quantity and quality as well as spatial distribution) to the coastal environment from aquifers underlying the Adelaide plains;
- to compare quantity and quality of groundwater discharge with other inputs to coastal environment e.g., stormwater, river loads, sewage and to establish its relative importance as an input to the zone (Section 1.2 below).

IS 1 SP4 – Dryfall and Wetfall Inputs: The project conducted a thorough audit and compilation of existing data as well as the analysis of existing suspended sediment samples to evaluate the contribution from atmospheric sources of dissolved and solid loads to coastal waters. The objective of this task was to determine the relative contribution of nutrients and other contaminants to the coastal waters from atmospheric sources (reported in ACWS Technical Report No. 17).

1.2 Stakeholder issue responses and overall IS 1 loads

This section answers the following ACWS Scientific Steering Committee research questions:

- *“3.2.1.3 How does stormwater’s contribution to nutrients in the Gulf compare in significance to discharges from the wastewater treatment plants? Is nitrogen the only nutrient input of concern?”*

Stormwater nutrient inputs to the ACWS coastline are small relative to WWTP inputs at all times of the year, only on rare occasions do stormwater nutrient inputs exceed WWTP loads, such events are short-lived. Investigations by EMP1 (Simon Bryars et al.) indicate that nitrogen is the key nutrient.

- *“3.2.1.2 (and 3.2.2.2) What are the relative contributions of all contaminant inputs to the system and how do these compare to the impacts of WWTP sourced nutrients in terms of significance of effects?”*

Figure 1.1 presents the bulk inputs from the major sources of liquid volume and nutrient nitrogen and phosphorus to the ACWS area. The estimated groundwater flow to the ACWS coastline is very small (ca. 1 % by volume). Wetfall is calculated as total rainfall to a 10 km wide rectangular

strip running from the Gawler River to Sellicks Creek. In the sense of a liquid input, wetfall presents a major contribution, however, it must be remembered that rain would have historically fallen in this zone and only represents an anthropogenic impact in so much as the current nitrogen concentration of rainfall is elevated relative to unpolluted air. Despite this elevated nitrogen concentration, wetfall only contributes around 1 % of total nitrogen. The delivery of nitrogen oxides from rainwater is somewhat more significant, but more as a consequence that the other inputs contribute relatively less $\text{NO}_x\text{-N}$ (Figure 1.3).

The Penrice discharge into the south arm of the Port Adelaide River is the major single input of nitrogen along the coastline, the significance of this input is not clear since the load is discharged into Barker Inlet rather than directly into the coastal strip and seagrass beds. The Penrice nitrogen load is ammonia nitrogen a component of total Kjeldahl nitrogen (TKN).

TKN is the dominant form of nitrogen from terrestrial inputs (both WWTPs and stormwater) at around 2400 tonnes a year. $\text{NO}_x\text{-N}$ only accounts for around 500 tonnes N per year. Of the direct coastal discharges, WWTPs are the largest component of the input nutrient load. Section 5.2 presents these inputs for each location ordered geographically from north to south.

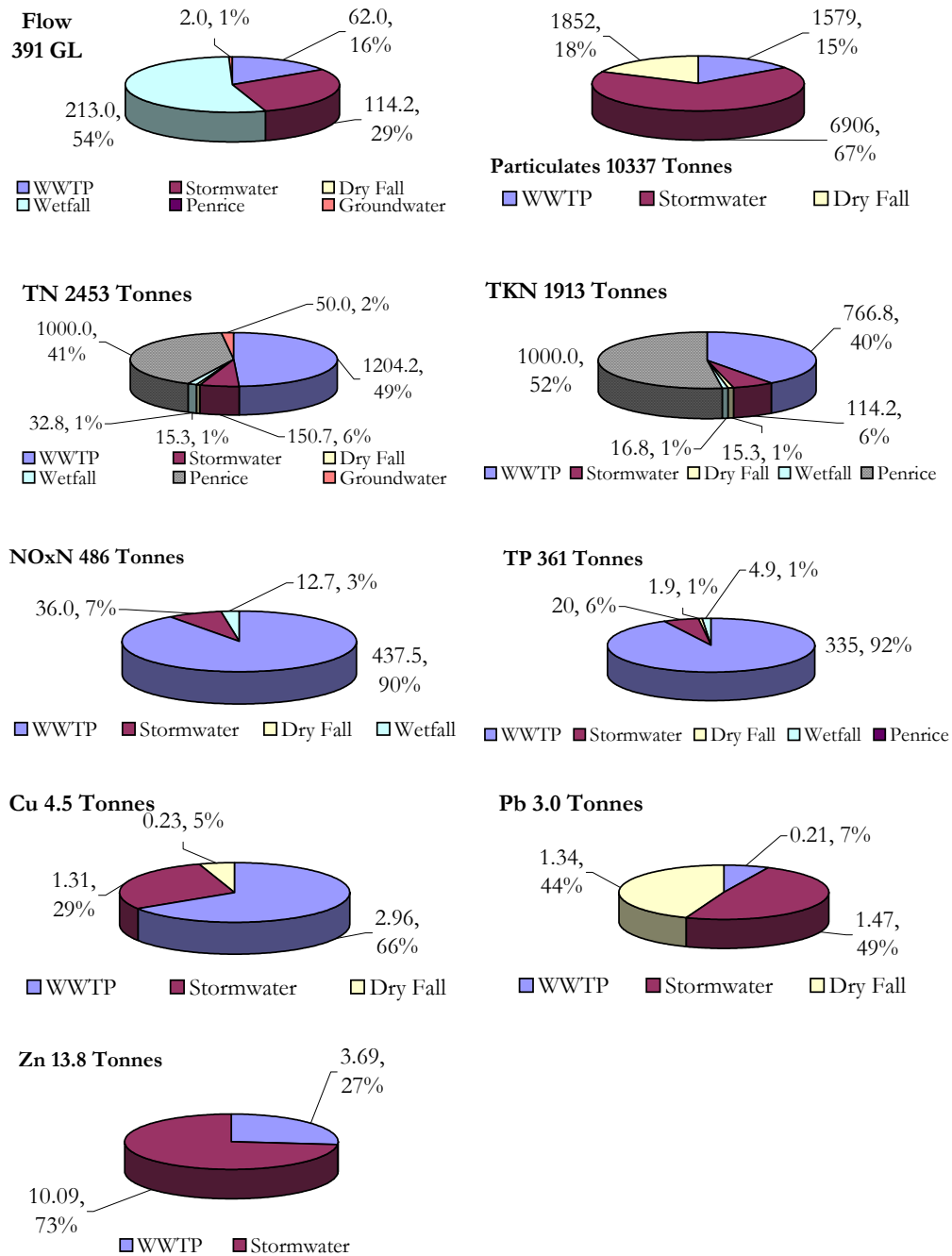


Figure 1.3. Overall flows/volumes, particulate matter, nitrogen, phosphorus, and heavy metal loads to the Adelaide Coastal zone from the four IS1 sub-programs.

Figure 1.3 also presents the suspended load and heavy metal loads for the four IS1 sub-programs. Unlike the WWTP discharges which dominate nutrient loadings, the suspended load is dominated by stormwater inputs from the major creeks and storm drains and accounting for around 80 % of the 6700 tonnes per year of suspended load. These figures are broken into component inputs in Section 5.2.

Zinc is the metal discharged in the greatest load to the coastal strip with around 14 tonnes per year and around 75 % of this comes from stormwater. The pattern for lead is similar at around

90 % from stormwater, but a much lower total load than zinc (only 1.7 tonnes per year). For copper the pattern is reversed. The loading of copper is dominated (around 70 %) by WWTP inputs and this is presumably due to great extent to erosion of copper pipework.

- *“3.2.7.2 What are the principal agents of concern in stormwater? Are pesticides, suspended solids, nutrients, heavy metals (or unknown inputs/interactions) creating the most significant environmental degradation?”*

This is answered below. The target ecosystem of the study has been the seagrass meadows along the Metropolitan Adelaide shoreline. Consequently, the focus of the input studies has been on the input components of most significance to seagrass die-back. For stormwater the most significant “contaminant” is turbidity in the main due to the disturbance of the seagrass light climate. More detailed questions about seagrass response to light climate are answered elsewhere in the study. The nutrient component of stormwater is small relative to WWTP inputs. Heavy metal inputs are of a similar order of magnitude to those from WWTPs, however, for seagrasses, heavy metals are not considered to be a significant stressor. Pesticide usage within catchments is known to be poorly regulated and many substances banned elsewhere in the world are widely used in Australia, although these substances may bio-accumulate in certain organisms, their impact on seagrasses is not known. A review of literature on pesticides in sediment, the water column and marine organisms is still needed.

- *“3.2.7.4 What are the relative impacts of the different outfalls along the coast in terms of assimilative capacity? Are discharges in deeper fast flowing water less detrimental than those in shallow slow moving water? Is relocating discharges from Port Adelaide River to Bolivar a good idea?”*

The best historical example of relative assimilative capacity was the difference in response of seagrass beds to the Glenelg WWTP and Port Adelaide WWTP sludge outfalls. The Glenelg outfall operated between 1963 and 1993 and had a barely discernable impact on seagrass condition in the vicinity of the outfall. The Port Adelaide outfall, which discharged in *deeper water*, caused the loss of hundreds of hectares of seagrass between 1978 and 1993. The impact of the Port Adelaide outfall was put down to the less energetic deeper waters at the discharge site compared to the shallower Glenelg location where currents and hence dispersive capacity were stronger (contrary to the supposition of question 3.2.7.4).

As for the assimilative capacity at other locations this is not easily measured, since the variability and differences in magnitude of the sources, and the differing conditions at each discharge location mean that each input can only be appraised on the basis of the current situation in light of historical inputs. Clearly the Torrens, Glenelg WWTP outfall and Patawalonga are centred on the main area of near shore seagrass loss, thus the simple conclusion would be that these inputs have exceeded the assimilative capacity of that zone of coastline. Evidence suggests that nearshore decline has slowed and as indicated in TR1 and above input quality has improved (Glenelg WWTP – N) and the total volume discharged annually was reduced by the construction of Kangaroo Creek Reservoir.

In addition, comparative data on stormwater quality suggest that the mineral stormwater turbidity component is much reduced now compared to the mid-1970s (see below).

South of Holdfast Bay, the coast slopes away more rapidly, thus there are not seagrass beds to the same extent as those northwards. In the very south of the study area, the Cactus Canyon Creek is deeply incised and may be contributing to cycles of seagrass development and loss, but this has not been a focus of the current study and might warrant further investigation should further residential development go ahead in the southern zone.

Specific questions regarding the relocation of the Port Adelaide discharge have been posed. Relocation of the Port Adelaide effluent to Bolivar will result in a more saline discharge to sea, this increase in salinity may be beneficial. Reduced salinity has not been shown to be detrimental to mature seagrass plants, but is detrimental to seedlings (work by Bryars et al. *get correct reference*). Relocation of the Port Adelaide effluent will not cause and increase in nutrient load discharged from Bolivar or deterioration in effluent quality. This is because the influence of the new input is offset by the EIP improvements in nutrient removal and due to the reduction in effluent volume afforded by the Virginia Pipeline irrigation scheme (see TR 1 – Wilkinson et al. 2004). In addition, the High Salinity Plant (HSP) will produce a cleaner effluent than did the old Port Adelaide WWTP (PATW). The PATW transfer to Bolivar may be beneficial to the Barker Inlet system since the key source of phosphorus to that system will have been removed. The closure of PATW also results in a 25 to 30 % reduction in the total nitrogen discharged into Barker Inlet, it would be difficult to argue that this could be seen as anything other than a beneficial outcome.

- “3.2.7.8 *What is the relative contribution of road runoff to any turbidity induced impacts?*”

A more targeted question might have been asked here. Stormwater *is* the key source of turbidity induced impacts. The question remains, to what extent is the turbidity of stormwater determined by runoff directly from the road surface when compared to soil/regolith erosion from gardens, roadside verges, other areas of bare earth, and leaf and organic matter?

The significance of road runoff is evidenced by the presence of lead and reduction in lead concentration in the period from 1995 to 2005. This does not confirm that the turbidity was derived from the road surface, since lead in exhaust fumes and associated with particulates will have deposited on the road surface and in the vicinity of the roadway. Lead was positively correlated with suspended solids or turbidity at all sites where adequate data were collected (see Appendix III). These were locations with significant percentages of urban and residential development. Despite the correlation of lead with turbidity, the binding of lead to particles may have occurred during transit through the stormwater system. The association of lead with particles may be an artefact of the similar transport characteristics of these particles. The observations of the author during the course of the study were that erosion of mineral particulates from exposed ground on sloping surfaces was the most apparent contributor to stormwater turbidity. Organic matter, decayed leaf matter, would be suspected as a major contributor to particulate discharges, however, water quality determinations showed that the volatile component of suspended solids was small in most cases. Having made this statement, it is obvious that gross pollutant traps retain large quantities of stormwater-borne organic material. Leaf matter clearly contributes to stormwater colouration and gum tree throughfall is also coloured. This colouration will also reduce light intensity and penetration in the water column, but to a lesser extent than reflective mineral turbidity.

In conclusion, further studies are required to determine with certainty the contribution of direct road runoff to turbidity relative to other stormwater components.

- “3.2.7.9 *Are the treated effluent discharges the only cause of the coastal water decline?*”

No, the treated effluents are not the only cause of coastal water decline. The sea-grass retreat margin in the Central Zone of the ACWS study area is clearly associated with the extent of regular periods of near shore turbidity. This turbidity is caused by turbid stormwater discharges from the Torrens River and Patawalonga Creek systems. It is likely that the periods of turbidity are also associated with wind-wave resuspension of fine material deposited from turbid stormwater during quiescent conditions following stormwater events. The design of IS1 has not been to address wind-wave resuspension, however, there is a high likelihood that this is a significant component of near shore turbidity. Anecdotal evidence from casual observations by the author has been that periods of high turbidity in the near-shore zone do occur during dry

weather when strong on-shore winds occur.

- “3.2.7.13 What should drive management objectives for the Torrens, Patawalonga and Onkaparinga Rivers (given that ANZECC Guidelines are not reasonably achievable)?

To answer this question clearly it is important to set the context of the answer in relation to each system.

- *Onkaparinga*

Management objectives for the Onkaparinga River are currently influenced by river health in association to environmental flows. It is appropriate that the ecological health of the Onkaparinga River is the focus for management of the system. The Onkaparinga is unlike the Torrens and Patawalonga in that its location means that it has an open estuary mouth that meets the coastline in a valley bordered by hills and cliffs to the north and south. The consequence of the Mt Bold reservoir is that discharge volumes from the Onkaparinga system are significantly reduced compared to historical flows. It is not clear whether this has been beneficial or otherwise to the health of coastal waters. One consequence of this, however, is that loads of suspended material delivered to the coastal zone by this river will have been reduced as overtopping of the reservoir has declined.

In relation to the major areas of seagrass decline in the Central Zone of the ACWS area, this may have resulted in a reduction in turbid water drifting northwards (the likelihood of this would have to be assessed by the other project programs).

- *Patawalonga and Torrens*

The Patawalonga and Torrens Rivers and their tributaries, prior to European settlement, meandered out from the Adelaide Hills Face across the Adelaide Plain. Both rivers terminated in back-dune swamplands. The swamps were connected along from north to south with the South Arm of the Port Adelaide River which now includes the West Lakes System. The significance of the pre-European era is that any direct hydraulic connection between the creeks and rivers draining out across the Adelaide Plain was minimal. The consequence of this is that the nearshore shore zone of the Central Zone of the ACWS area would have received little if any direct turbid runoff and only from the Patawalonga. The swamps would have acted as a low energy depositional zone for any soil eroded by larger storm-events. The creek systems crossing the plain would have provided attenuation of discharges arising in the Adelaide Hills. The lack degree of vegetation cover, lack of soil compaction and lack of impervious surfaces would have resulted in only a small proportion of rain falling on the plain would have become runoff and streamflow. Large flood events in the Upper Torrens system would have flooded the marshes, this would have afforded settlement of suspended material and the excess water would have drained into Barker Inlet as well as seeping to groundwater and across the dune-face to the sea. This sets the context against which to contrast the current situation in relation to seagrass decline, water resource and climate change related issues.

The current situation of the Torrens and Patawalonga systems is that they are configured to afford maximum flood alleviation. Many kilometres of the feeder creeks and drains are concrete lined and straightened. This maximises flow velocity and transmission of water downstream. It minimises the potential for creek-bed recharge of groundwater, or the settlement of suspended matter. A consequence of the high degree of urbanisation is that the Torrens, Patawalonga and Ocean Catchment have the highest runoff yield of any of the creeks or areas draining the Adelaide Metropolitan area (90 to 110 mm/y).

The opinion of the author is that the management objectives for stormwater discharge from the Patawalonga and Torrens systems must be considered in a holistic way.

There are three key areas against which management objectives must be considered:

- *Stormwater as a water resource*

This includes the promotion of initiatives for the capture of stormwater before it enters the drainage network and ties-in with initiatives to reduce mains water consumption. Once rainwater moves from the roof zone to ground level, the quality deteriorates rapidly. A key objective of management initiatives should be to encourage water users, both large and small, and en-masse, to harvest rainwater for all uses, especially green space watering. Building and development standards should be updated to specify on site drainage of non-harvested stormwater – i.e. encourage development with a smaller “dcp” (less impervious area that is directly connected to the extended stormwater drainage network). Appendix VI highlights the current Adelaide hydrological and water supply network and loops that could be closed to reduce flows to sea.

- *Within catchment actions to improve stormwater quality*

Stormwater management objectives that tackle street cleansing and the emptying of side entry pits is an essential component in improving stormwater quality. Through contact with organic material stormwater quality deteriorates in both colour and nitrogen concentration. The collection and removal of this material before it comes into contact with runoff would improve runoff quality.

2. Quantities of terrestrial water inputs to ACWS area

This section of the report presents the variations and trends in overall storm and wastewater flows since 1940, the changes in seasonality in flows associated with urbanisation and gives examples of the relative contributions of different waste and stormwater inputs over time.

2.1 Long-term variations in stormwater and wastewater flows to sea

Figure 2.1 demonstrates the variations in annual mean air temperature (Adelaide), rainfall at Branden in the Sturt catchment, total stormwater runoff and total wastewater treatment plant flows. The air temperature demonstrate a decline between 1940 and 1960 followed by a steady increase with regular oscillations varying in period from 5 to 10 years. These oscillations in air temperature contribute to the variations in stormwater flows (which are derived from air temperature and rainfall via the model described by Wilkinson, 2005). In 1957, the minimum in air temperature resulted in a peak stormflow year with the total stormflow approaching 380 GL. The 20-year moving average stormflow (not presented here) clearly demonstrates the reduction in stormflow associated with the completion of the Kangaroo Creek reservoir in 1969. The upward trend in air temperature and reduction in rainfall since the early 1960s have contributed to reduced overall stormflows since that time. In the 1940s, storm runoff is predicted as being low and this coincides with elevated air temperature during this period. Table 2.1 provides a summary of estimated total wastewater and storm flows for ten-year intervals.

Table 2.1. Ten-year mean annual wastewater and stormwater flows between 1945 and 2005 (historic stormflows based-on measured and modelled flows as described in Wilkinson (2005)).

	1945-54	1955-64	1965-74	1975-84	1985-94	1995-2005
Wastewater Flow (GL)	26.5	47.8	48.9	58.6	73.5	75.2
Stormwater Flow (GL)	122.2	161.9	119.3	102.2	118.8	116.1

Monthly records of sewage flows into Glenelg and Port Adelaide WWTPs have enabled the estimation of overall WWTP flows of reclaimed wastewater to sea since 1945 (Appendix I, and Wilkinson et al. 2004). Figure 2.1 and Table 2.1 demonstrate the increase in wastewater flows relative to stormwater flows since 1945. The volume of wastewater discharged has increased three-fold from 1945 to the present day. The plot of wastewater flows shows a significant decrease at around 1967. This reduction in flow is associated with the switch of the Adelaide and northern suburbs flows from the Islington Sewage Farm to Bolivar WWTP. The tertiary lagoons at Bolivar afford significant reduction in the flow to sea due to evaporation. The discharge volume from Islington, to the North Arm of the Port Adelaide River, was estimated based-on the correspondence of the Bolivar and Port Adelaide sewage inflows (Figure 2.2, Appendix I). Two assumptions were made, firstly that there was minimal alteration from inflow to outflow at the Islington works and, secondly that the increase in load to the Islington Plant over time was consistent with that at Port Adelaide. The correlation of sewage flows at the two plants between August 1967 and August 1974 was $R^2 = 0.743$, such that *Islington Flow* = 3.11 *Bolivar Inflow* -99.4 ML. From 1975 onwards the sewage flow at Bolivar grows faster than that at Port Adelaide as a consequence of the expansion of the northern suburbs. At Port Adelaide the potential for further major residential expansion was relatively much smaller. The establishment of Bolivar and consequent transfer of the largest metropolitan wastewater flow from Islington meant that rather than this discharge entering the Barker Inlet system via the North Arm of the

Port Adelaide River the discharge now entered Gulf St Vincent directly without the potential for mixing and biological interactions.

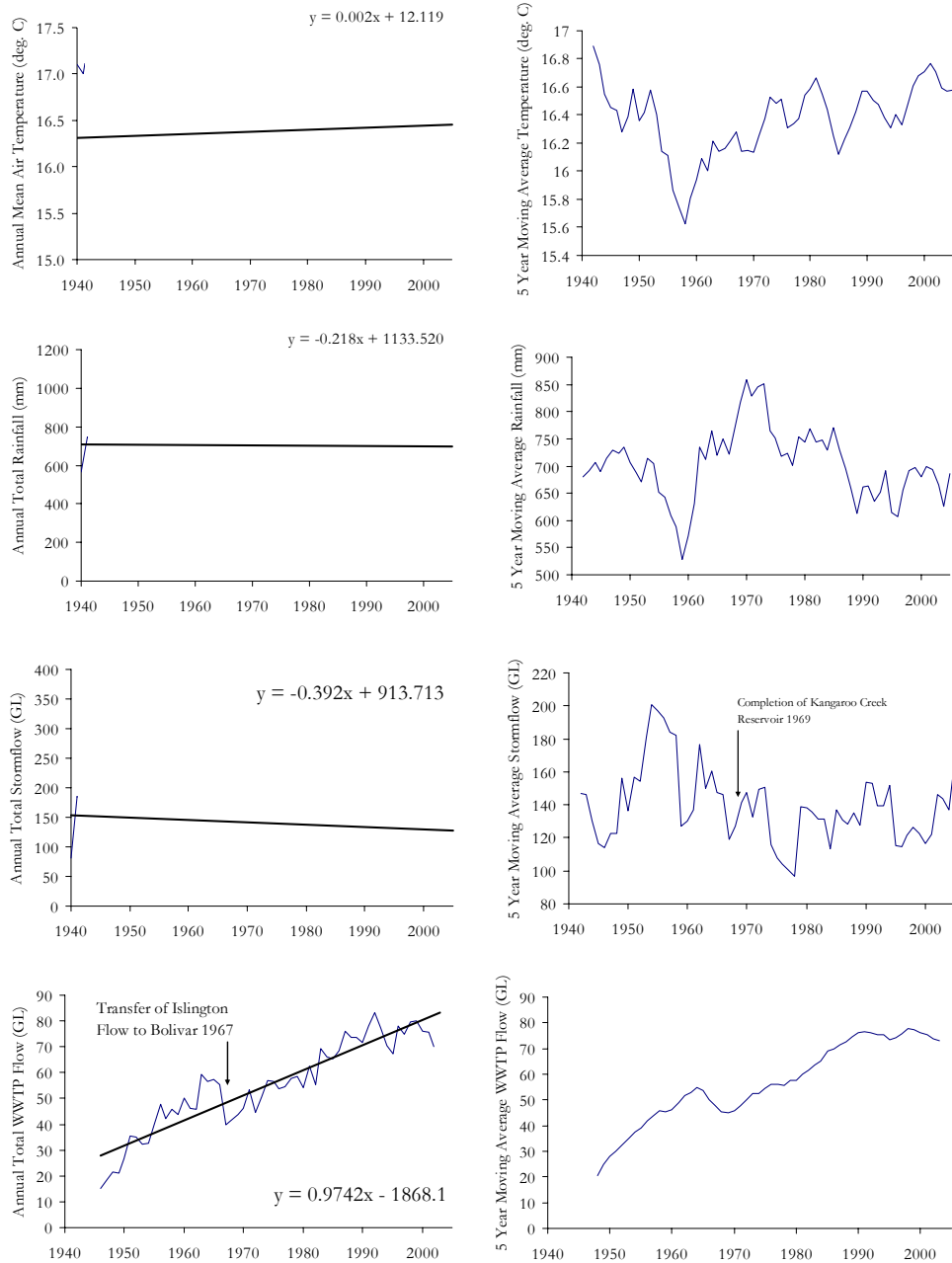


Figure 2.1. Variations and trends in air temperature, rainfall, stormflow volume and wastewater volume from 1940 to the present day (historic stormflows based-on measured and modelled flows as described in Wilkinson (2005)).

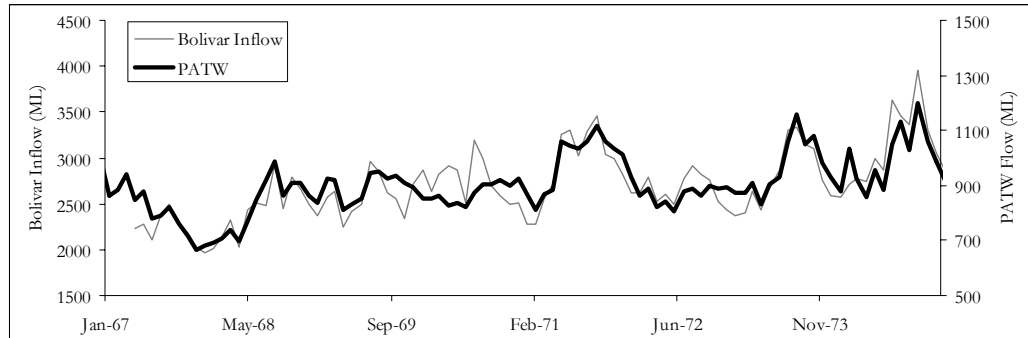


Figure 2.2. Bolivar WWTP and Port Adelaide flows between 1967 and 1974.

Overall, the estimated flows to sea indicate a steady increase in discharge volume of around 1 GL per year until the late 1990s when re-use schemes for recycled water began to reduce the annual flow. The trend in stormwater discharges shows a decline of around 0.4 GL per year since 1940. This is due to rainfall and temperature variations as well as the damming of the Little Para and Torrens watersheds. Although these figures demonstrate a reduction in overall stormwater flows, the summer component of stormwater discharges has increased as described in Section 2.2 below.

2.2 Seasonal variations in overall stormflow volume from 1945 to 2005

Since 1945 when regular records of monthly mean sewage flow started, the total volumes of wastewater flow have increased three-fold as mentioned above. In addition to the increase in wastewater volumes over this time, stormflow gauging records and modelling by Wilkinson (2005) show an overall reduction in stormflows and changes in the seasonal distribution of stormflows (Figure 2.3a). These data suggest a broadening of the stormwater season, i.e. the duration of the season is longer, with reduced winter flows as a consequence of watershed damming and increased summer flows resulting from urbanisation. The overall total summer (December to end-April) stormflows have increased six-fold (Figure 2.3b) from 1.2 GL in the period 1945-54, to 7.6 GL in 1995-2005. The greatest increase in summer stormflows is suggested from the mid-1960s (Figure 2.4). Figure 2.3b also shows the growth in summer wastewater discharges, the consequence of the increases in summer flows would be enhanced nutrient availability from reclaimed water and the increasing frequency of turbid stormwater slicks that shade nearshore plants (contaminant loadings are discussed in Chapter 5).

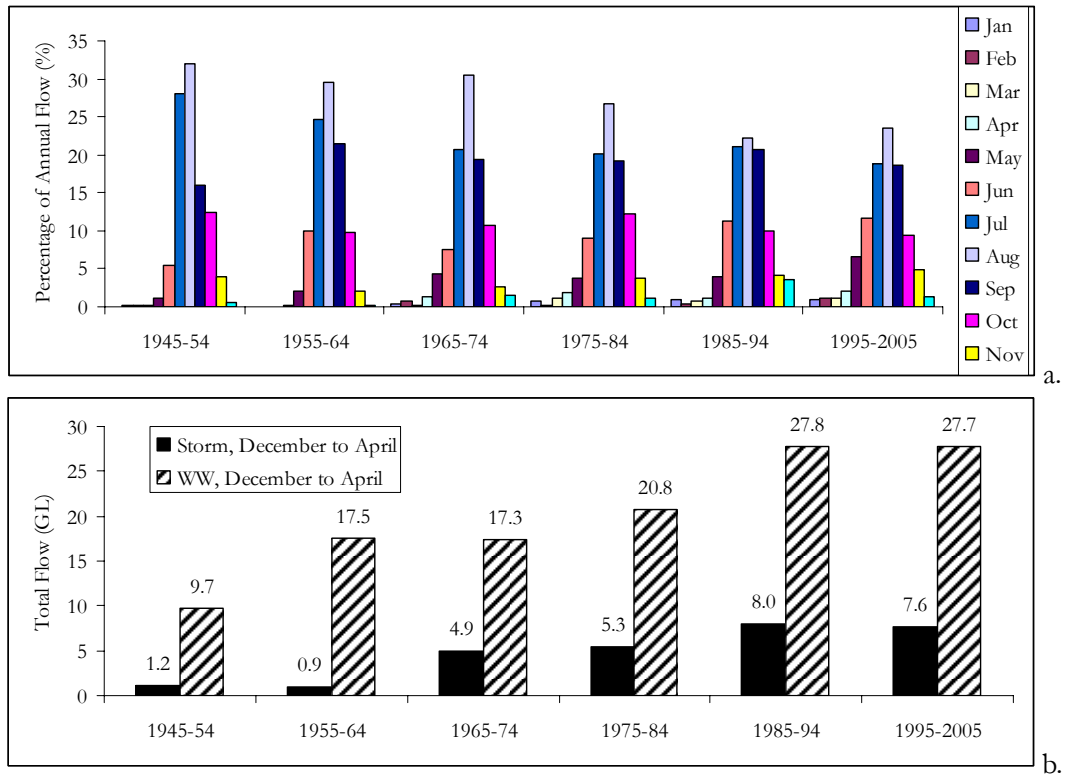


Figure 2.3a. Monthly distribution of mean total annual stormflow for ten-year periods from 1945 to 2005, and b. total summer storm and wastewater flow volumes for the same intervals as a above (historic stormflows based-on measured and modelled flows as described in Wilkinson (2005)).

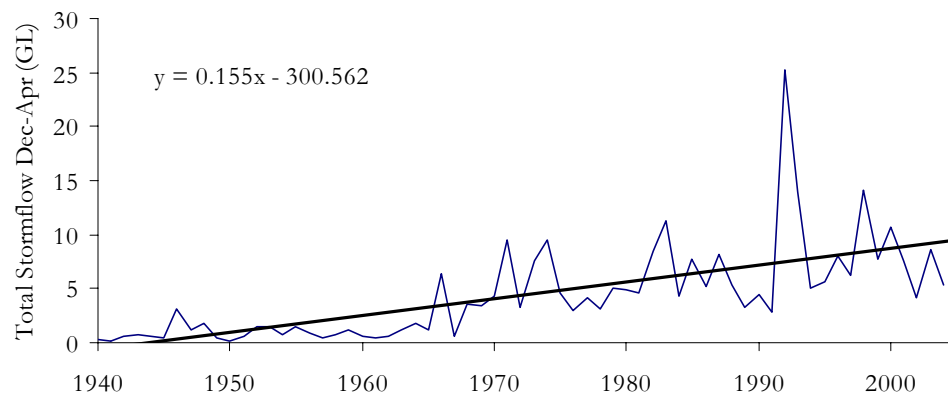


Figure 2.4. Year on year variation in total summer stormflows (December to end April) from 1940 to 2005 showing a marked increase from the mid 1960s.

Figures 2.3 and 2.4 have demonstrated the overall changes in wastewater and stormwater flows for all major inputs summed together.

2.3 Variations in flow volume of major rivers and creeks from 1945 to 2005

Figure 2.5 presents mean monthly stormwater volume estimated for two periods 1945-54 and 1995 to 2005 for each individual major stormwater source.

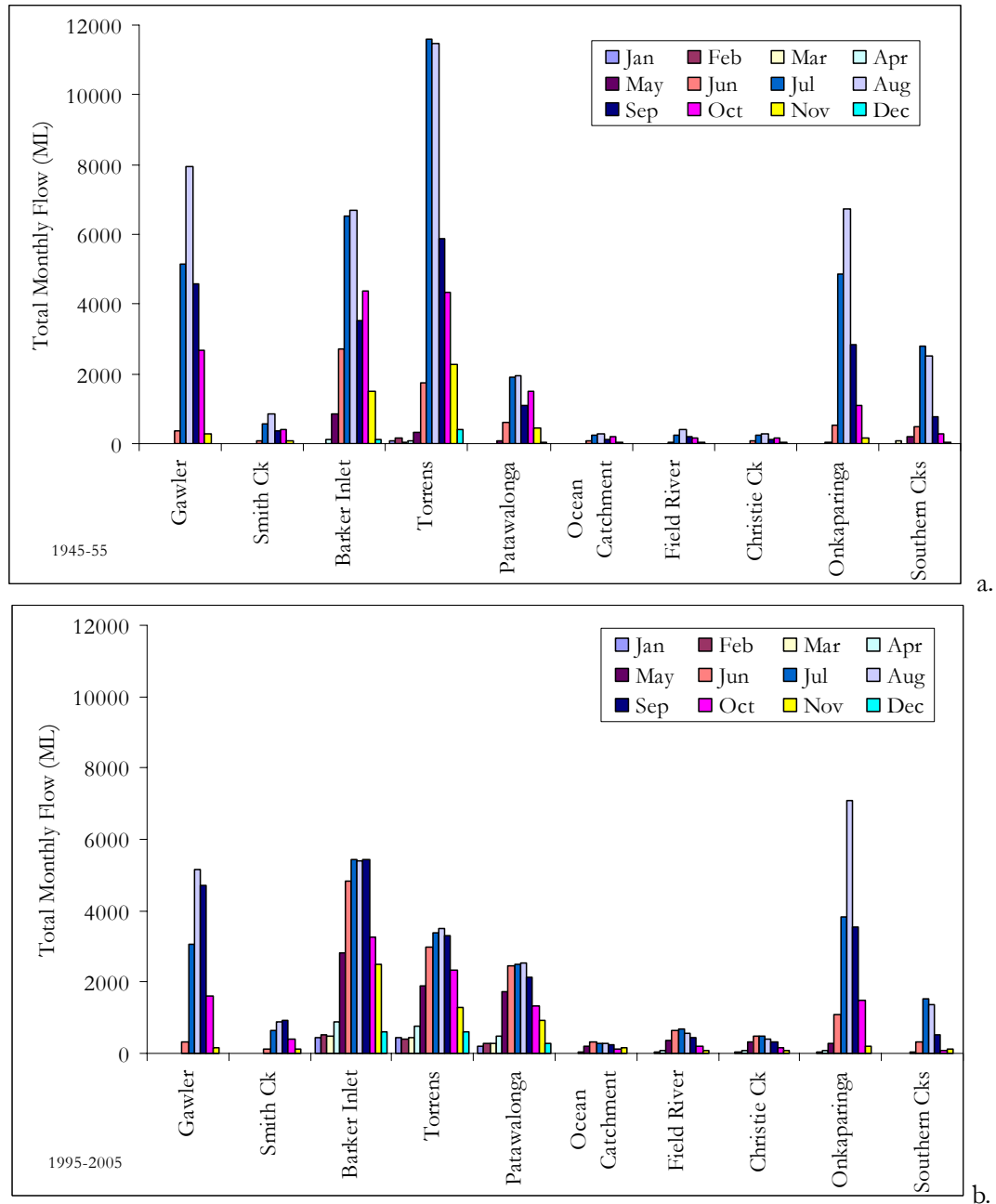


Figure 2.5. Monthly mean storm flow from major stormwater sources in a. 1945-54 and b. 1995-2004 (historic stormflows based-on measured and modelled flows as described in Wilkinson (2005)).

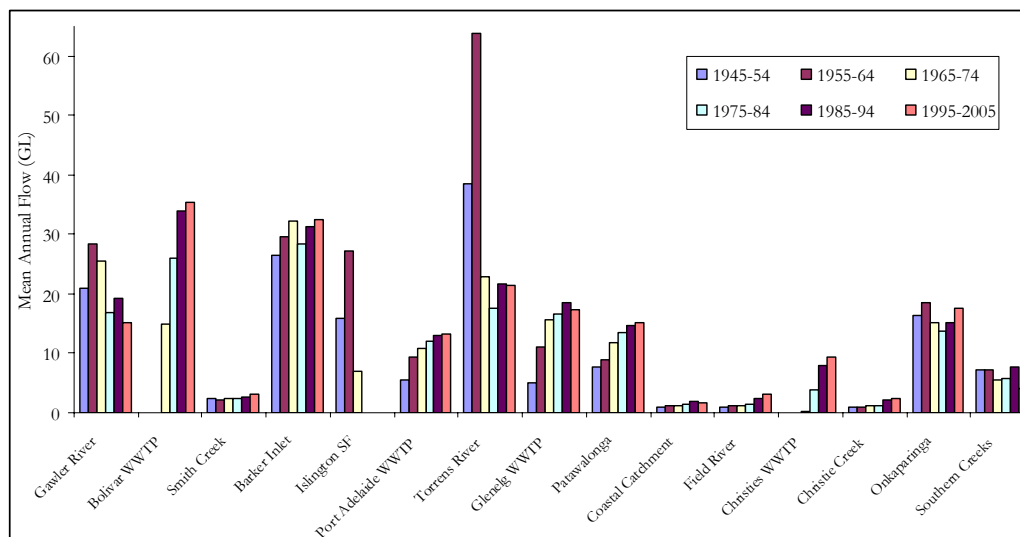


Figure 2.6. Mean annual flow volume in GL for Storm and Wastewater Inputs to Adelaide Coastal Waters for ten-year intervals from 1945 to 2005.

Table 2.2. Mean annual flow volumes in GL for storm and wastewater inputs to Adelaide Coastal Waters for ten-year intervals from 1945 to 2005.

	1945-54	1955-64	1965-74	1975-84	1985-94	1995-2005
Gawler River	21.01	28.36	25.61	16.86	19.27	15.06
Bolivar WWTP	-	-	14.92	26.11	34.06	35.34
Smith Creek	2.36	2.23	2.37	2.41	2.68	3.06
Barker Inlet	26.41	29.65	32.27	28.43	31.20	32.59
Islington SF	15.88	27.28	7.10	-	-	-
Port Adelaide WWTP	5.66	9.43	10.83	12.02	12.98	13.20
Torrens River	38.40	63.68	22.85	17.64	21.63	21.33
Glenelg WWTP	4.98	11.05	15.66	16.67	18.49	17.26
Patawalonga	7.64	8.99	11.78	13.41	14.69	15.18
Coastal Catchment	1.02	1.13	1.22	1.41	1.87	1.68
Field River	1.02	1.17	1.27	1.34	2.43	3.15
Christies WWTP	-	-	0.36	3.78	7.95	9.38
Christie Creek	0.95	1.07	1.11	1.17	2.06	2.41
Onkaparinga	16.26	18.47	15.20	13.71	15.23	17.65
Southern Creeks	7.17	7.12	5.58	5.78	7.70	4.01

The data presented in Figure 2.5a and b are plotted with the same y-axis scale to help indicate the relative changes in volume between the two periods. The information presented demonstrates that the patterns in overall changes in stormflow seasonality are most strongly seen in the central metropolitan zone, including the creeks feeding Barker Inlet and south to Christie Creek. In this central urbanised area, the stormwater season has broadened in all cases. The overall stormflows from these inputs have, in general, increased with time (Table 2.5, Figure 2.6). An exception is the Torrens (and to a lesser extent the Little Para in its contribution to Barker Inlet inflows) as a result of the damming of the watershed; this has

reduced mid-winter (July and August) extreme stormflows (and to a lesser extent in the Gawler River).

The estimates of ten-yearly mean annual flows presented in Table 2.5 and Figure 2.6 indicate that the greatest increases in flow have occurred in the Field River and Christie Creek, where annual flows have increased 2.5 to 3 fold. Table 2.3 presents corrections to figures for Christie Creek presented in Wilkinson et al. (2005), where it was stated that the Christie Creek discharge values derived from the gauged control structure at Galloway Road were anomalous and indicated a catchment yield more than twice that of any other creek in the metropolitan area. The source of the anomaly has since been traced to the rating curve used in the stage (river level) / discharge relationship used to estimate the flows. This relationship was a theoretical rating and has since been recalculated. This now provides discharge estimates consistent with other creeks, and the new time-series of flows was used to calibrate the stormflow model presented in Wilkinson (2005).

Table 2.3. Correction of Christie Creek discharge and catchment yield. area and runoff per unit area for selected major rivers and creeks in the ACWS study area (previously reported (Wilkinson et al. (2005) values are in parentheses).

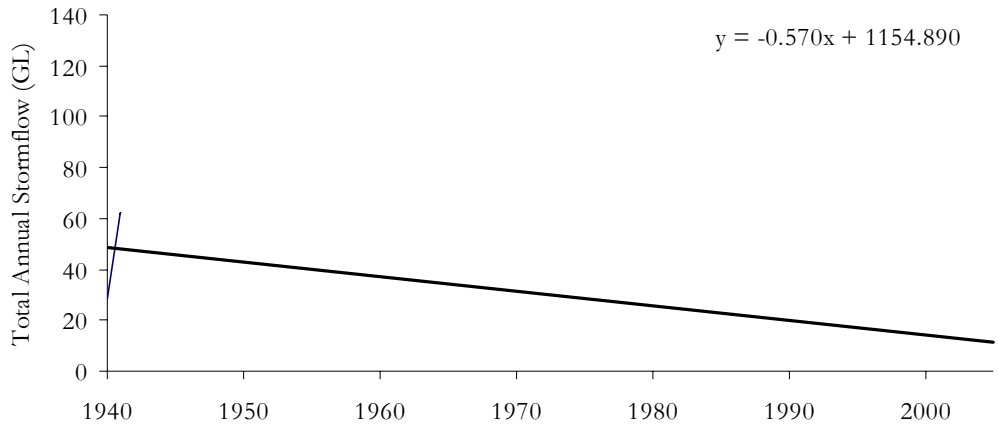
	Catchment area (km ²)	Mean annual flow (GL)	Catchment Yield (ML/km ² = mm)
Christie Creek ¹	37.8	<u>2.7</u> (8.1)	<u>72.1</u> (214.3)

1. Annual mean of flows for April 01 to April 03.

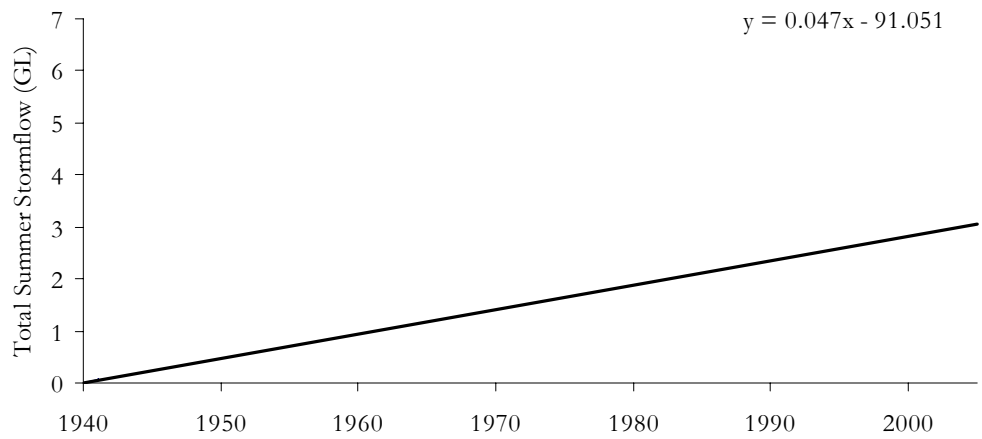
2.4 Significance of changes in storm and wastewater flow volumes

The changes in storm and wastewater flows highlighted above have important implications in terms of changes in the overall loads of pollutants and their distribution along the coastline and consequently the nearshore seagrass beds where the changes have taken place.

In the longer-term historical context is the impact of cutting through the “breakout channel” of the Torrens River. From Table 2.2 and Figure 2.6, it can be seen that the area of coastline where Breakout Creek emerges, went from effectively zero stormwater flow to anywhere between 3.5 and 115 GL annually. From 1940 to 1969, prior to the completion of Kangaroo Creek Reservoir, the mean annual flow in the Torrens was 46.3 GL, and from 1969 to 2004 was less than half that figure at only 19.7 GL (Figure 2.7a). The simple conclusion that might be drawn from this observation is that this reduction in total annual flow from the Torrens was “good” for seagrass survival, however, that would be ignoring the changes in the seasonal distribution of flows (highlighted in Section 2.2 above). Figure 2.7b suggests increasing summer flow volume, which, while only small by comparison to total annual flow, would contribute greater summer seagrass shading. As stated in Section 2.2, this increase in summer flows applies to the Patawalonga system and the Ocean Catchment storm drains, although in the case of the Patawalonga, the overall flows have increased with time, and in recent years the Barcoo Outlet has bypassed the Patawalonga Lake which formerly acted as a retardation lagoon and sink for suspended material (which led to contact recreational health risks and prompted construction of the new siphonic outlet). Flushing of the Patawalonga Lake by major flow events did result in the scouring of anoxic silts under the lock gates. The current system allows material to be flushed regularly without the creation of severe oxygen depletion of the coastal water. The load estimates for the Patawalonga in 1975-85 are based on the data for the Patawalonga Lake at King Street Bridge (Section 5.5, Appendix IV). In addition to the changing stormwater discharge patterns, the Glenelg WWTP discharge doubled every ten years from a mean of 5 GL in the 1945-54 period up to 15 GL by the mid-nineteen seventies (Table 2.2) and has subsequently only grown at a modest few GL every decade.



a.



b.

Figure 2.7. Annual flows to sea from the River Torrens from 1940 to 2004 showing, a. total flow volume, and b. total summer (December to end April) flows.

3. Stormwater Quality

Chapter 3 of this report focuses on stormwater quality, i.e. the concentrations of the key seagrass-impacting constituents of stormwater. The inter-elemental(?) concentrations are also presented with respect to flow and suspended solids or turbidity, this gives an indication of the tendency of dissolved constituents to be either diluted or concentrated with changing flow. Nitrogen speciation is presented in a seasonal context on a site by site basis where sufficient data are available. Christie Creek is demonstrated to receive dry weather flow from groundwater, and lead concentrations in stormwater are shown to have declined with airborne lead and use of leaded petrol.

3.1 Stormwater median concentrations

Tables 3.1 to 3.4 present median values of conductivity, suspended solids and nutrient concentrations in major stormwaters draining to the Adelaide Coastal Zone. Full tables of statistics are presented in Appendix II.

In general, stormwaters entering Adelaide Metropolitan waters have median suspended sediment concentrations of around 20 to 40 mg/L (see also Tables A2.1 to A2.4). The southern creeks have lower suspended sediment concentrations than the central zone creeks and storm drains. This is possibly due to the sampling regime; ambient sampling will tend to sample a low proportion of storm events compared to event-based grab samples, in addition composite flow proportional results include a greater coverage of high flows. In Table A2.4, the mean concentration may be more representative of stormflow water quality for suspended sediment since it is biased towards the less frequent higher concentration samples. The mean values in Table A2.4 are closer to the median values presented in Table A2.1, especially for Christie Creek. The results for Field River in Table 3.1 (A2.1) and 3.2 (A2.4) are for different locations, one mid-catchment and one at the outlet, and these cannot be compared directly, although the differences receive more attention below. Sellicks Creek is known to be heavily incised in the section downstream of South Road and has a mean suspended sediment concentration 2.5 times the median value, approximately 50 mg/L.

The water quality of the southern creeks presented in Table 3.2 is broadly consistent with that of other creeks in Metropolitan Adelaide. In general these creeks have low total phosphorus relative to the rest of Metropolitan Adelaide. Only Willunga Creek stands out in this respect. The elevated total nitrogen (mean = 3.2 mg N/L), ammonia nitrogen (mean = 0.74 mg N/L) and total phosphorus (mean = 0.41 mg P/L) suggest contamination from domestic waste water and might deserve further investigation. Both the Field River and Christie Creek have elevated oxidised nitrogen relative to most other creeks and stormwater drains. Section 3.3 below discusses these findings in more detail. In Tables 3.1 and 3.2, as previously mentioned, two different sites for Field River are presented. Data from the composite flow proportional monitoring site downstream of Main South Road are presented in Table 3.1. In Table 3.2 ambient water quality at the catchment outlet at Hallett Cove is presented. The median NO_x concentration at Hallett Cove (1.52 mg N/L) is more than 15 times greater than that upstream at Main South Road (0.10 mg N/L), and the TKN is approximately half that at Main South Road. This is discussed in more detail in Section 3.4.

The suspended sediment concentrations in the southern creeks (Table 3.2) are lower than those measured elsewhere in the ACWS study. This is consistent with the fact that these data are from ambient monitoring which rarely captures elevated discharges associated with storm runoff. The data in this table do include storm event grab samples collected during ACWS IS1 and these are the higher values in the distribution of concentrations.

Table 3.1. Median concentrations of water quality determinants in stormwater from the major central zone sources (see Table A2.1 for full details). Flow is measured in ML/d and all concentrations are in mg/L.

Location	Period	Statistic	Flow	EC	SS	TN	TKN	NO _x	TP
Torrens	1996-2005	median	19.5	939	21.0	1.36	0.81	0.528	0.071
Brownhill	1996-2005	median	8.9	437	17.0	1.01	0.80	0.165	0.134
Sturt	1996-2005	median	6.3	1090	24.0	1.23	0.95	0.147	0.217
Ocean	2004	median	2.1	117	30.7	1.02	0.97	0.127	0.191
Field	2001-05	median	1.5	734	23.5	0.98	0.87	0.103	0.080
Christie	2001-05	median	0.7	2870	39.0	1.61	0.96	0.588	0.101

Table 3.2. Median concentrations of water quality determinants in water from the creeks of the Southern Metropolitan Zone from 1999 to 2004 including OCWMB ambient monitoring and IS1 stormwater sample analyses (see Table A2.3 for full details). All concentrations are in mg/L (note that the data for Field River is for the gauging site downstream of Main South Road).

Location	Statistic	TDS	SS	TN	TKN	NO _x	NH ₃ _N	TP	SRP
Field	median	1300	18.5	2.00	0.48	1.52	0.018	0.038	0.016
Christie	median	3100	16.5	1.40	0.56	0.74	0.024	0.032	0.016
Onka ON	median	1100	6.0	0.99	0.74	0.10	0.023	0.047	0.016
Onka Est	median	32500	17.3	0.71	0.64	0.03	0.068	0.047	0.012
Peddler	median	2300	8.0	0.90	0.88	0.03	0.024	0.101	0.050
Willunga	median	9050	18.0	1.22	1.23	0.02	0.054	0.256	0.112
Maslin	median	12000	13.0	1.07	1.06	0.01	0.032	0.077	0.014
Aldinga	median	3000	9.0	1.52	1.41	0.12	0.022	0.053	0.016
Sellicks	median	1300	19.0	0.51	0.50	0.01	0.011	0.037	0.022

Table 3.3 presents median concentrations of various water quality variables for the storm drains of the Patawalonga Ocean Catchment. A comparison between results for the Gulf St Vincent Water Pollution Studies (GSVPS) between 1973 and 1978 and data collected during ACWS IS1 in 2004. Full statistics are presented in Table A2.2. The variable flow in Table 3.3 is the median of daily modelled flows for both periods, the procedure for generating these flows is presented in Wilkinson (2005). The median values indicate several changes between the past and present concentrations. There has been a four-fold reduction in suspended solids concentrations, this was evidenced in photographs presented in Wilkinson et al. (2004). A three-fold reduction in total organic carbon has occurred. Reductions in nitrogen are also evident; the reduction mainly being due to the drop in NO_x concentration, a reduction in TKN was not seen at every site. The TKN concentration from the Edwards Street drain appears to have increased, this may be a consequence of the gross pollutant trap which is located a few metres back from the drain outlet. The drain was emptied in June 2004, and had been anaerobic producing a noticeable “landfill gas” smell. The emptying of the GPT did not have an obvious impact on subsequent nitrogen speciation or concentrations. The cause of the reductions in nitrogen concentrations is not clear. In general, the changes in water quality suggested by the data summarised in Tables 3.2 and A2.2 constitute an improvement in quality. There has been no apparent change in phosphorus concentrations, which might have been expected to be lower consistent with the reduction in suspended solids. The decline in pH is an indicator of deterioration in water quality, since this reduction might tend to increase heavy metal mobility. The reduction in pH may be a consequence of the reduction in suspended matter and hence pH buffering by readily reacting minerals. Since there are no major cation or anion concentrations this cannot be tested. pH decrease will also result from sulfide or Fe(II) oxidation which happens as anoxic waters aerate.

Table 3.3. Median concentrations of water quality determinants in stormwater from the Patawalonga Ocean Catchment (South Western Drainage Scheme) in 2004 and from 1973 to 1978 (see Table A2.2 for full details). Flow is measured in ML/d and all concentrations are in mg/L.

Location	Period	Statistic	Flow	SS	pH	TN	TKN	NOx	NH3_N	TP	SRP	TOC
Broadway	2004	median	0.56	34.2	6.33	0.78	0.73	0.07	0.171	0.19	0.064	6.86
Broadway	1973-78	median	0.43	88.0	7.60	1.34	1.04	0.28	0.060	0.14	0.064	22.00
Harrow	2004	median	1.90	24.0	6.26	0.99	0.93	0.10	0.067	0.22	0.078	8.02
Harrow	1973-78	median	1.48	112.0	7.60	1.32	0.98	0.44	0.060	0.19	0.069	22.50
Wattle	2004	median	1.44	23.3	6.25	1.05	1.00	0.15	0.082	0.20	0.041	6.29
Wattle	1973-78	median	0.95	120.0	7.30	1.91	1.61	0.36	0.040	0.20	0.141	27.00
Edwards	2004	median	2.21	40.2	6.09	1.01	0.96	0.05	0.074	0.19	0.052	7.48
Edwards	1973-78	median	2.91	110.0	8.00	1.18	0.62	0.48	0.080	0.19	0.085	17.00
Young	2004	median	2.98	45.8	6.27	0.58	0.50	0.07	0.060	0.16	0.061	6.68
Young	1973-78	median	2.12	270.0	8.10	1.59	1.18	0.93	0.055	0.15	0.055	19.00

As part of the IS1 analytical program volatile suspended solids, carbon speciation and phosphorus partition was investigated (Table 3.4). The median volatile suspended fraction was around 0.25 of the total suspended solids with a small number of high values at around 0.6 these tended to occur at low total suspended solids concentrations (Figure 3.1). Total organic carbon comprised around 44% of the total carbon with higher proportions of organic carbon occurring at the lower total carbon concentrations. Of the phosphorus, between 30 to 50% was in the dissolved form and most of the dissolved phosphorus was bioavailable reactive phosphorus.

Table 3.4. Additional water quality determinants not recorded in other programs.

All units mg/L.	Total Carbon	Inorganic Carbon	Total Organic Carbon	Total Phosphorus	Total Dissolved Phosphorus	Soluble Reactive Phosphorus	Suspended Solids	Volatile Suspended Solids
Valid n	64	64	64	64	64	64	61	61
Median	13.6	6.63	6.6	0.193	0.071	0.061	31.0	8.0
Mean	15.6	7.02	8.59	0.212	0.099	0.073	54.3	12.0
Minimum	6.46	2.17	0.97	0.055	0.013	0.013	0	0
Percentile 95	33.0	11.1	20.8	0.531	0.258	0.214	144	39.3
Std Deviation	7.43	3.13	5.5	0.123	0.081	0.068	68.3	13.5

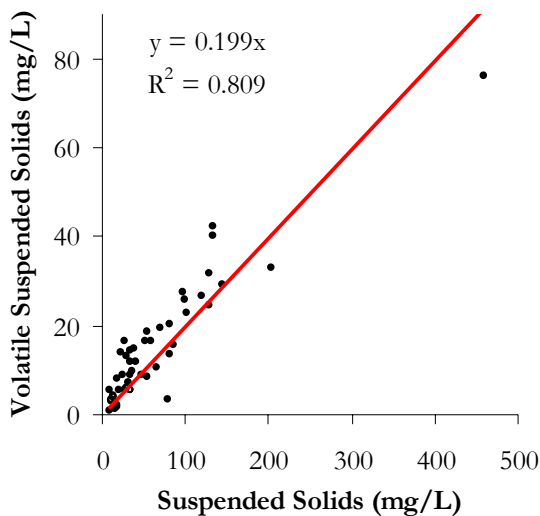


Figure 3.1. The relationship between suspended solids and volatile suspended solids in IS1 water quality grab samples.

In Table 3.5, the water quality of the Onkaparinga River at Old Noarlunga is presented for two periods, 1978-1980 and 1999 to 2003. The only apparent difference in these data is in the NO_x nitrogen concentration which is higher in the earlier data.

Table 3.5. Water quality of the Onkaparinga upstream of Old Noarlunga (This is the input to the estuarine system, all values in mg/L unless otherwise stated)

	Conductivity (mS/cm ²)	TDS	Turbidity (NTU)	SS	NH ₃ -N	TKN	NO _x -N	FRP	TP
1999-2003? Reference?									
Medians	2080	1100	2.8	5.5	0.02	0.73	0.13	0.02	0.05
Averages	6402	3794	6.2	12	0.07	1.10	0.15	0.03	0.10
n	51	51	51	44	51	49	35	30	51
Glatz (1985) June 78 -June 83									
Medians	2410	1320	1	3	0.02	0.56	0.32	0.01	0.03
Averages	2680	1470	2	9	0.05	1.31	0.44	0.01	0.07
n	18	18	16	13	13	18	18	16	17

The data presented in Tables 3.6 and 3.7 show far greater suspended solids concentrations in the northern stormwater sources than in the central and southern stormwaters. The data presented are almost all from earlier studies and may no longer be applicable, especially in the case of creeks entering Barker Inlet through wetland filtration zones. The data presented by Glatz (1989) Table 3.7 for the Gawler River indicates suspended solids concentrations similar to those in the central and southern creeks and drains. The other results suggest a much higher suspended sediment concentration and this might be expected in a largely agricultural catchment with a natural channel and a large supply of loose particulate matter in the channel.

Table 3.6. Geometric mean concentrations of water quality determinants in water from stormwater sources entering Barker Inlet from Little Para River south to Jenkins Road drain (Derived from digitised data presented in Hine *et al.* (1989), three or four samples were collected in April and May 1989). All concentrations are in mg/L.

Location	SS	TN	TKN	NO ₃ _N	TP
Dry Ck	1315.2	2.30	1.79	0.51	0.60
Little Para R	121.5	1.22	1.00	0.22	0.27
Jenkins Rd	33.1	2.77	2.40	0.37	19.26
Eastern Pde	79.0	2.63	2.40	0.24	1.51
South rd	86.1	2.49	2.01	0.48	0.61
Dunstan rd	257.2	2.35	1.98	0.37	0.81
HEP Drain	222.1	2.11	1.75	0.36	0.67
North Arm East	328.9	2.52	2.04	0.48	0.58
North Arm Rd	61.2	1.27	0.85	0.42	0.20

Table 3.7. Concentrations of water quality determinants in stormwater from the Gawler River various sources. Electrical conductivity is measured in µS/cm², all other values are in mg/L.

Period	Source	EC	SS	TKN	NO ₃ _N	TP
Jun 73 - Nov 74	Geometric Mean (n=6), EWS 3876		179		0.21	0.15
1978-83	Median (n=14), Glatz (1989)	1920 (n=54)	24 (n=9)	1.48	0.19	0.21
Aug-04	Geometric Mean (n=3), this study	1606	119	2.08	0.20	0.39

A brief review of the quality of recycled water quality shows that recycled water has greater concentrations of nutrients and lower suspended solids. These sources have varied widely in suspended solids concentration, Bolivar has historically always had a high suspended solids ranging from annual average values of around 150 mg/L to 35 mg/L. Glenelg WWTP has delivered around 12 mg/L suspended solids consistently since the mid 1970s; occasionally higher annual averages were recorded. Christies Beach WWTP has had very variable suspended solids concentration, at around 10 mg/L in the early years of operation, with a period of producing higher concentrations (25 mg/L) between the mid-nineteen eighties and mid-nineteen nineties. It currently discharges suspended solids at around 10 mg/L. Total nitrogen concentrations in stormwater (approximately 1 to 2 mg/L) are around 15 to 20 times lower than historical wastewater nitrogen concentrations (30 to 40 mg/L), and 5 times less than post-EIP WWTP nitrogen concentrations (approx. 10 mg/L). In general, stormwater nitrogen is dominated by TKN. Median total phosphorus concentrations in the central zone are around 0.2 mg/L (Tables 3.1 to 3.3) and around 20 times lower than WW concentrations (see Wilkinson et al., 2004). In the southern creeks (Table 3.2), total phosphorus appears to be lower than in the central zone stormwaters despite the differences in sampling regimes (the mean is also lower than the central zone median values).

These data demonstrate general similarities and patterns in stormwater quality. Exceptions become most apparent when looking at seasonal speciation of nitrogen (Section 3.3 below).

3.2 Interelemental relationships

Having presented mean water quality concentrations for the major creeks and streams, the relationships between the variables for all data are presented. Later, the seasonal and temporal variations are presented and finally load discharge relationships and seasonal loads.

Figure A3.1 to A3.13 present matrix scatter plots of water quality determinants and flow in the Metropolitan Adelaide creeks that have routine composite flow proportional sampling. This approach has been used to good effect in previous studies (e.g. Wilkinson et al., 1997). The frequency distribution of river flow and most water quality determinants tend to be log-normal. Log_{10} transformation facilitates the visualisation of data that would be skewed tightly to the graph origin if viewed in raw form. In addition, the log-normal nature of the data means that the arithmetic mean is not a good descriptor of the central tendency in the data, as it will tend to be too high. The geometric mean (where zero values are not common) and the median give the best indication of central tendency.

River Torrens at Holbrooks Road

Of the major central Metropolitan Creeks contributing stormwater to the Adelaide coastline, the relationships between flow, dissolved and suspended constituents in the River Torrens are the most complicated. The Sturt River and Brownhill Creek are largely canalised and offer the least potential for chemical and biological alteration of stormwater runoff. The River Torrens has the Torrens Lake in Central Adelaide and many standing pools in the reaches between the lake and the sea.

Major anions and cations were analysed for sporadically between 1997 and 2004 with 29 composite samples. The data demonstrate simple correlation between all of the analysed major ions (Figure A3.1) indicating dilution of more concentrated water by stormwater. There are not sufficient samples to provide a clear relationship with discharge, although this is more apparent for the more frequent conductivity (EC) determinations (Figure A3.2).

The contaminant metals Pb and Zn show a weak positive correlation with discharge related to the in flow of contaminated stormwater. The scatter in this relationship is enhanced by the fact that Pb and Zn concentrations have decreased in stormwater since the phasing out of leaded

petrol. Pb and Zn correlated with each other and Pb correlates positively with suspended solids due to its tendency to associate with solid phases, whereas zinc tends to occur in soluble forms. Suspended solids and total phosphorus show a weak positive correlation, and total phosphorus and TKN are weakly correlated. Nitrogen oxides (NO_x) demonstrate a complex relationship with flow and conductivity. This is likely to be a consequence of differing source waters, i.e. local stormwater from the catchment downstream of the Torrens Lake in Adelaide, and water from the lake where seasonal transformations will influence nitrogen speciation in water discharging into the downstream reaches of the river.

Brownhill Creek at Adelaide Airport

Figure A3.3 presents log₁₀ transformed flow, suspended solids and major cations and anions. There is evidence of a dilution effect as would be expected of the major ions, which correlate quite closely, although potassium exhibits a more complex “two-armed” relationship with the other major ions which suggests the presence of waters with distinct chemical composition as is more clearly observed in Christie Creek (see below).

TKN is the dominant form of nitrogen in stormwater from Brownhill Creek, although it is not known what proportion of this is ammonia nitrogen. TKN and total phosphorus are strongly correlated, and zinc correlates less strongly with TKN and total phosphorus (Figure A3.4). The presence of TKN, phosphorus and zinc may be indicative of constituents released from decaying organic matter and zinc from tyre wear products. Lead correlates with suspended solids. The oxides of nitrogen (NO_x) and copper show the least correlation with the other physical and chemical data displayed.

Sturt Creek at Anzac Highway

As for the River Torrens and Brownhill Creek, Sturt Creek has not had its major ionic components analysed frequently. The ionic concentrations tend to correlate and also indicate some evidence for source waters of varying magnesium, potassium and sodium constituents. A dilution effect is only weakly apparent and most strongly evident in the plot of flow and sulfate.

The contaminant and nutrient data are well scattered with similar weak relationships to those apparent in Brownhill Creek stormwater (Figure A3.5).

Field River at Main South Road

The strongest correlations between the water quality variables analysed for the Field River relate to dilution and suspended material (Figure A3.7). Turbidity is negatively correlated with total dissolved solids and the major cations and anions that comprise the total dissolved salts. Sodium and chloride are the dominant components of the total dissolved solids in Field River. Neither dissolved or suspended load (represented by turbidity in NTU in Figure A3.7) shows any relationship with flow in Field River, and flow is randomly scattered relative to all other analytes. Calcium, chloride and bicarbonate ions are strongly positively correlated. A weak positive correlation exists between TKN and total phosphorus (TP), as for other creeks, but these do not correlate with flow or dissolved matter, although a very weak relationship between TP and turbidity (NTU) is evident. Of the contaminant metals only lead shows a weak positive correlation with turbidity and TP, and a weaker negative relationship with dissolved solids. An examination of the temporal patterns of loads from Field River demonstrates a marked seasonal flushing effect (see later) which tends to blur any relationship with flow.

Christie Creek downstream of Galloway Road.

The major ion relationships with total dissolved solids (TDS) in Christie Creek are quite distinct from those in Field River, with all major cations and anions (Ca, Mg, K, Na, HCO₃, SO₄ and Cl) being strongly positively correlated (Figure A3.8). TDS and the major ions are negatively correlated with flow, demonstrating dilution by storm runoff. Nitrogen oxides are positively correlated with the major cations, as is pH (weakly). There is an unusual dog leg in the scatter

plot of calcium against the other major ions (Figure A3.8 and 3.2) this is also present in the NO_x plot. This feature in the data suggests the mixing of different source waters (e.g. ground/soilwater) of differing chemical composition. The transition appears to be associated with the increase in creek flow from dry weather flow to storm runoff fed flows (Figure 3.3). The changes in ionic composition associated with this transition would indicate that the dry weather flow is sustained by ground water high in sodium, chloride, magnesium, potassium and sulphate, but with a lower calcium concentration. Calcium richer water appears to come from storm runoff in response to rainfall, but is increasingly diluted as discharge becomes greater. The increase calcium would be expected due to the calcrete horizon of the near surface deposits in the Christie Creek catchment.

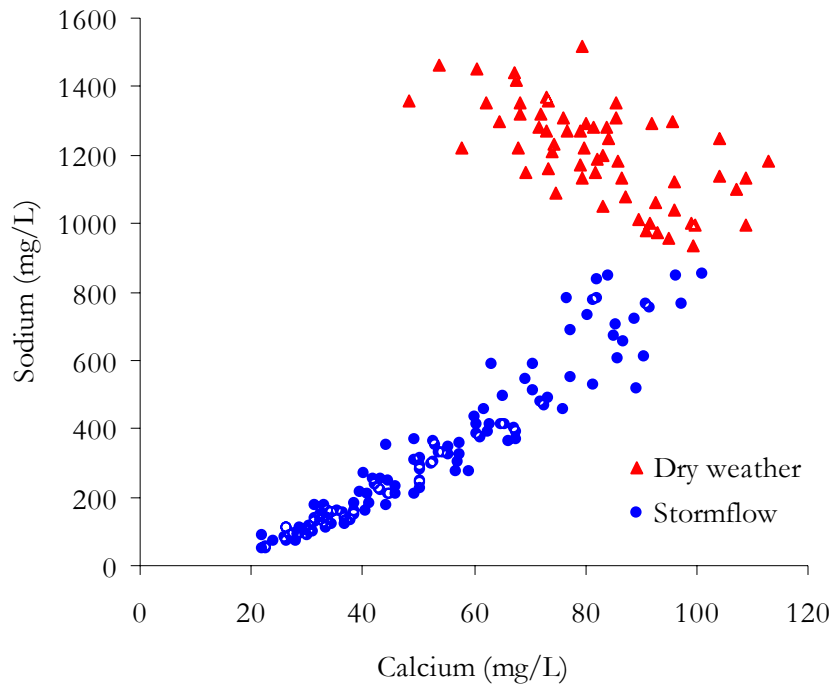


Figure 3.2. The relationship between sodium and calcium in Christie Creek flow proportional composite samples at the Galloway Road sampling location.

Total phosphorus and TKN exhibit the same weak positive correlation with each other and turbidity as seen in the Field River. Lead is also positively correlated with turbidity.

This investigation of inter-relationships between water quality variables using multiple (matrix) scatter plots has provided information that might not otherwise be apparent in time series or tabulated summary statistics of bulk data. In the case of Christie Creek these plots have highlighted evidence of a switching between a groundwater, enriched in magnesium relative to calcium and stormwater which is lower in magnesium than calcium. Highlighting these differences in source water has allowed the seasonal pattern in nitrogen concentrations and speciation in Christie Creek to be explained and contributing factors to be discussed (see Section 3.3 below).

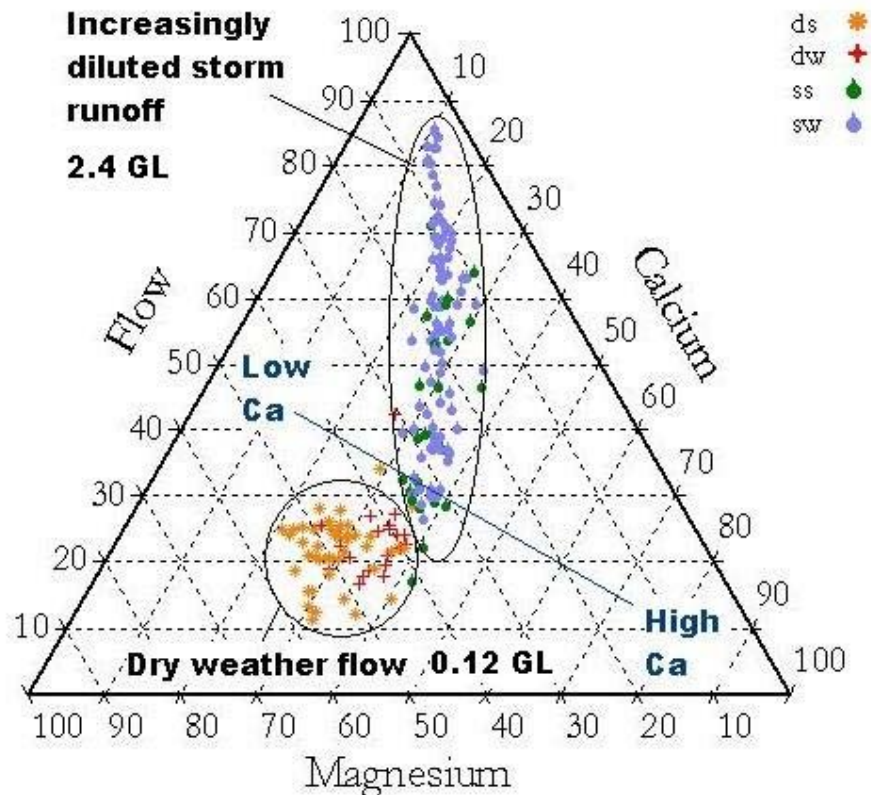


Figure 3.3. A ternary plot of calcium and magnesium against flow for Christie Creek. The data points are grouped according to runoff season and Ca:Mg ratio; ds = dry weather summer, dw = dry weather winter, ss = storm runoff summer, and sw = storm runoff winter (note: the “flow” variable is derived from $\log_{10}(\text{flow})$ scaled for maximum visual clarity).

3.3 Seasonal Variation in Stormwater Quality

Nitrogen

Nitrogen has long been considered a key agent in the decline of seagrasses along the Adelaide coastline (e.g. Steffensen, 1989). Data from the five compositely sampled creeks that contribute directly to Adelaide Coastal Waters give a valuable insight into the seasonal variation of nitrogen concentrations and the speciation of nitrogen (Figures 3.4 to 3.10). Those creeks and channels that act largely as stormwater conduits, i.e. Sturt River and Brownhill Creek (and the upper Field River), exhibit similar patterns of concentration and speciation, with elevated summer concentrations and lower winter concentrations. The upper Field River sampled at in the northern branch downstream of Main South Road is included here to support the demonstrated differences between those creeks that act largely as drainage conduits, and those which have some active bio-chemical behaviour or other complicating factors that alter the nitrogen speciation, such as the Torrens, Christie Creek and Field River at its outlet.

In the creeks that act as stormwater conduits, the oxidised forms of nitrogen are the minority component of the nitrogen load. TKN is measured but not ammonial nitrogen, however, additional sampling undertaken by IS1 indicates that ammonia nitrogen only accounts for approximately 10% of TKN which implies that the majority of the nitrogen is organic nitrogen, predominantly from the breakdown of leaf matter.

The median total nitrogen concentrations in Brownhill Creek and Sturt River (and upper Field River; Main South Road) and the storm drains of the Patowalunga Ocean Catchment are similar at around 1.0 to 1.2 mg/L (Table 3.1 and Table A2.1). The River Torrens, Field River (at the Hallett Cove outlet) and Christie Creek differ in that they have significantly higher oxidised nitrogen concentrations (Table 3.2). Figures 3.4 to 3.10 demonstrate the seasonality of the nitrogen concentrations in these creeks by providing monthly mean values of TKN and NO_x. The bars in the charts are stacked, so that the maximum value on the y-axis represents the mean monthly total nitrogen. Figures 3.4 to 3.6 show that the seasonal nitrogen behaviour in Brownhill Creek and the Sturt River (and upper Field River) are essentially the same. In these creeks TKN is the dominant form of nitrogen and concentrations peak in the summer months. Nitrate (NO_x) is the minor component of the total nitrogen throughout the year, but is higher in the wet winter months as it is in the Torrens.

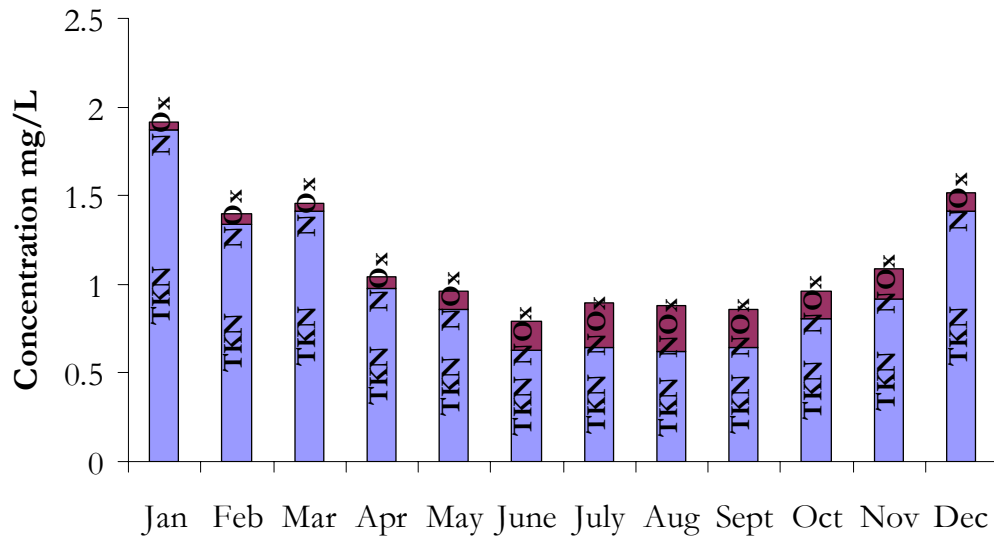


Figure 3.4. Monthly TKN concentrations 1997 to 2003 in Brownhill Creek at Adelaide Airport.

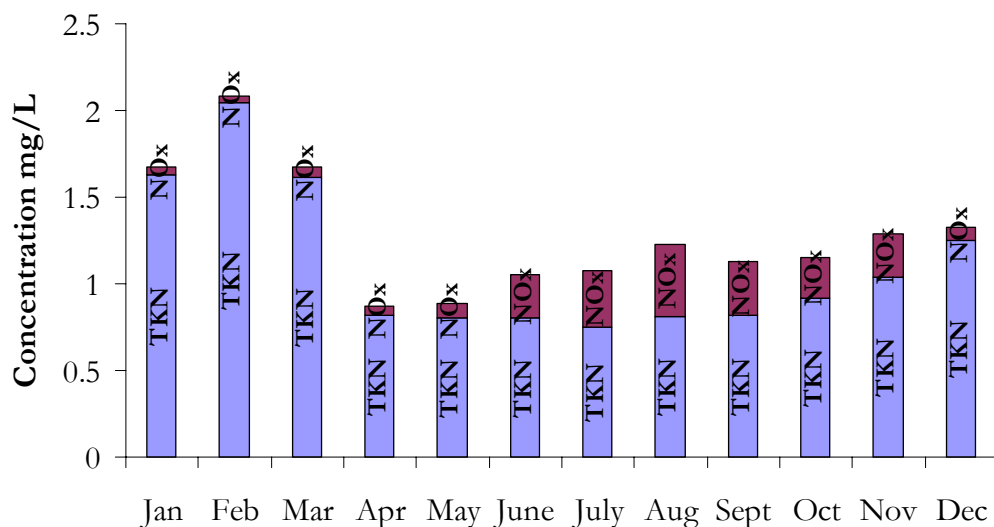


Figure 3.5. Monthly mean concentrations of TKN plus NO_x -N from 1996 to 2003 in the Sturt River at Anzac Highway.

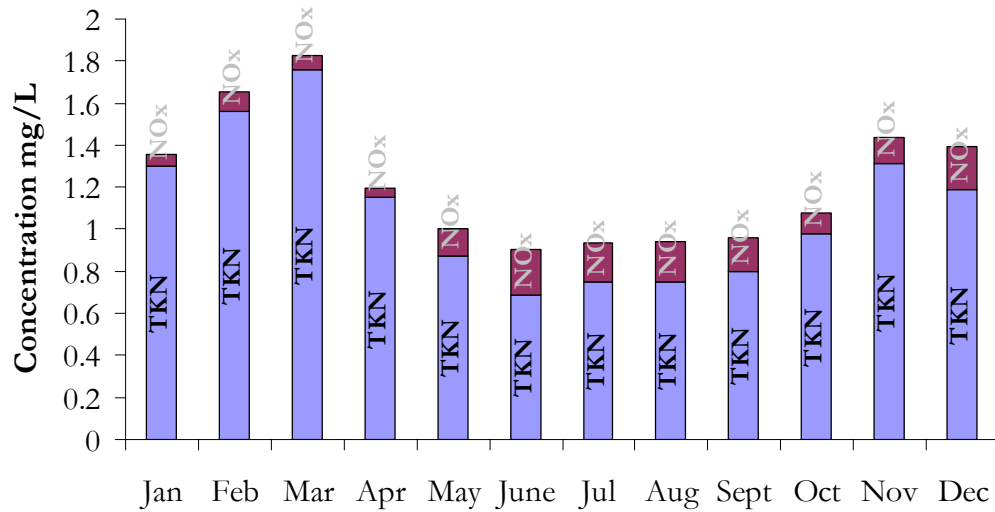


Figure 3.6. Monthly mean concentrations of TKN plus NO_x -N from February 2001 to April 2005 in Field River at Main South Road.

In the creeks that are not simply stormwater conduits, the nitrogen characteristics are somewhat different with elevated NO_x and lower TKN concentrations. These differences appear to be consistent with creek bed bio-chemical interactions and groundwater influences. Figure 3.7 presents monthly mean nitrogen concentrations in the River Torrens at Holbrooks Road. The seasonal variability of the nitrogen components is similar, i.e. TKN concentrations peak in the summer and are lower in the winter, and nitrate concentrations are highest in the winter months. The major difference is the scale of the nitrate concentration as a proportion of the total nitrogen. The cause of this difference in nitrogen speciation is most likely nitrogen transformations taking place in the Torrens Lake in Adelaide, and the various ponded parts of the lower channel, which act as storages of summer lower inflows which are relatively high in TKN. This TKN is mineralised in the lake and taken-up in algal blooms and other primary production through photosynthesis. In the winter months the decay (respiration) of these organisms and plant material releases the nitrogen back into the lake waters and is then flushed by winter flows downstream and on into the coastal zone. Consequently the seasonality of the maximum nitrogen concentrations from the River Torrens is reversed and in addition the nitrogen is in a more biologically available form.

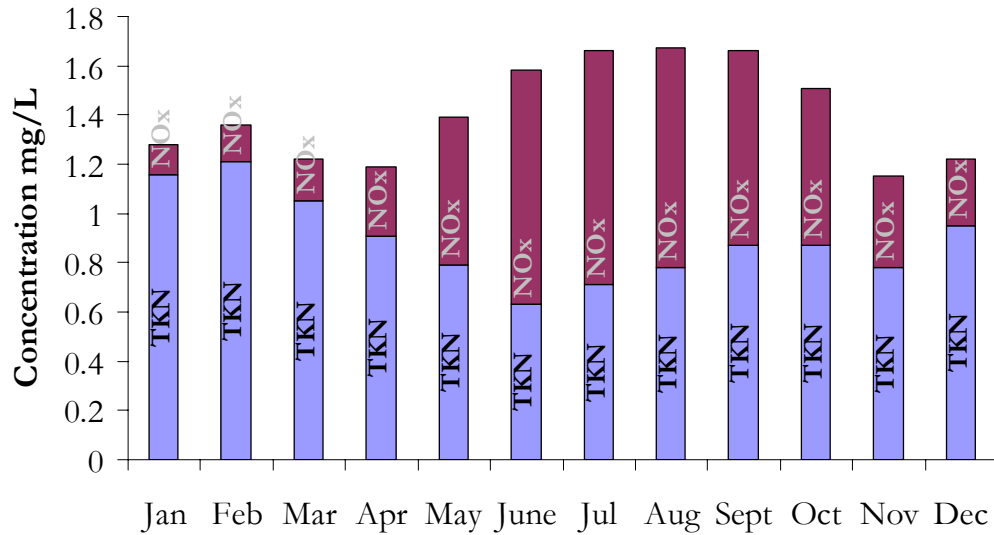


Figure 3.7. Monthly mean concentrations of TKN plus NO_x -N from 1995 to 2003 in the River Torrens at Holbrooks Road.

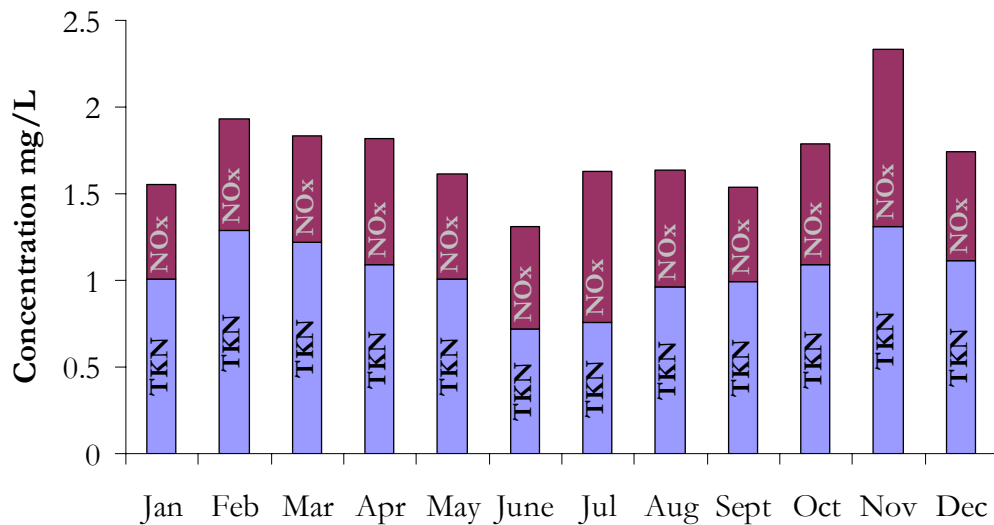


Figure 3.8. Monthly mean concentrations of TKN plus NO_x -N from February 2001 to April 2005 in Christie Creek downstream of Galloway Road.

The pattern of nitrogen speciation in Christie Creek differs again from the other creeks with flow proportional composite sampling (Figure 3.8). The pattern of TKN variation is similar to that in the Torrens, the nitrate concentrations, however, are consistently high throughout the year without major seasonal variation. A split of the water quality data on the basis of the chemical composition as indicated in Section 3.2 above and arbitrarily labelled as dry weather flow and storm runoff helps to demonstrate that the dry weather flow has greater nitrate concentrations than stormflow (Table 3.8), although the highest NO_x concentrations occur at the transition between groundwater to storm runoff. Samples that separate out as dry weather flow, occur in the height of summer at low river flow and have very high conductivity.

Table 3.8. Mean values of flow (ML/d), magnesium, calcium, TKN and NOx in Christie Creek dry weather flow and storm flow.

	n	Flow	Mg	Ca	TKN	NOx
Stormflow	147	10.5	39.4	51.3	1.07	0.58
Dry weather flow	61	0.5	124.6	82.5	0.99	0.82

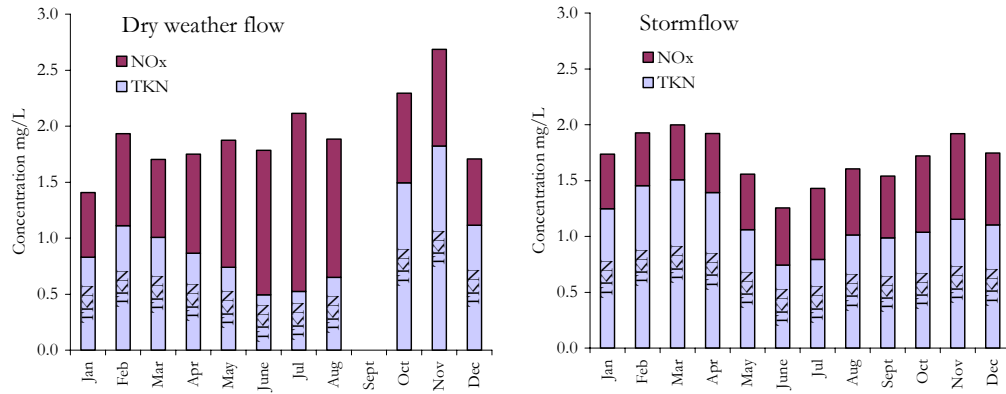


Figure 3.9. Seasonal mean nitrogen composition of dry weather and stormwater flows in Christie Creek.

The switching in chemical composition in Christie Creek indicated in Section 3.2 suggests a groundwater source feeding dry weather flow in the creek. During dry periods the creek is dry upstream of Kentwood Road (i.e. eastwards) apart from a few stagnant pools. From the reserve east of Brodie Road, which includes a wetland, the channel flows all year round. This location appears to be where the groundwater body is intercepted. The seasonal pattern in dry weather flow nitrogen speciation is similar to that in the Torrens, with higher winter nitrogen concentrations than in summer months. It is possible that the wetland moderates the nitrogen concentration by photosynthetic uptake in the summer and increased winter concentrations resulting from the lack of uptake and nitrogen release due to respiration.

The nitrogen concentrations at the Field River outlet at Hallett Cove (Figure 3.10) are similarly high in NO_x-N relative to TKN compared to the upstream site at Main South Road (Figure 3.6). At Hallett Cove, only grab sampling has been undertaken and there is no flow gauging station. Anecdotal evidence suggests that the creek flows at all times of the year and the area of cliffs to the south of Hallett Cove was known to indigenous peoples as a weeping wall (A. Scott pers comm.). Thus, assuming a groundwater feed to the lower reach of Field River, similar to that feeding Christie Creek, a new modelled flow has been generated that incorporates the Main South Road Flow plus storm and groundwater. The new modelled flow was used to estimate loads from Field River.

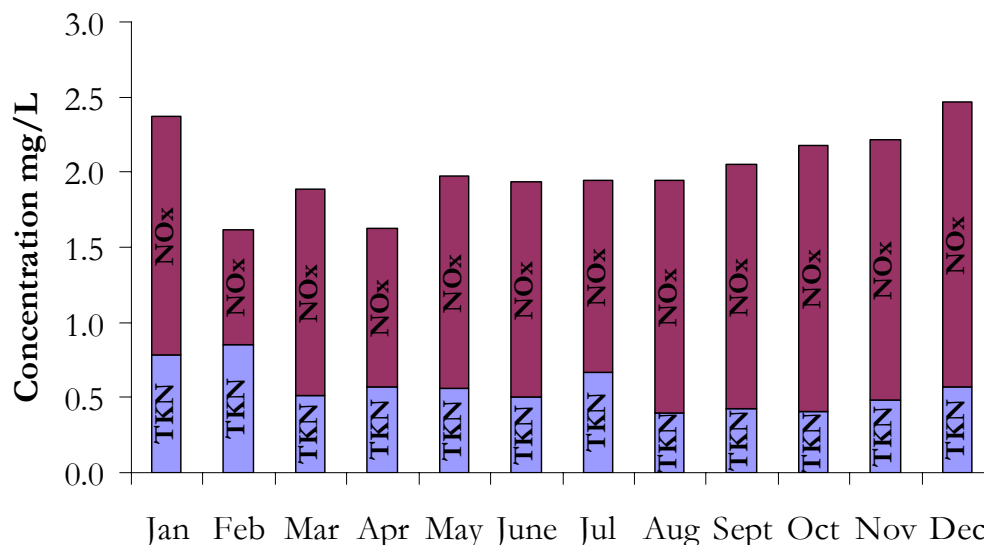


Figure 3.10. Monthly mean concentrations of TKN plus NO_x-N from April 1999 to October 2004 in Field River at Hallett Cove.

3.4 Copper, Lead and Zinc in stormwater

The ANZECC/ARMCANZ trigger concentrations for the protection of 95% of marine species from the toxic affects of copper, lead and zinc are 1.3, 4.4 and 15 µg/L, respectively. As indicated in the TR1 report (Wilkinson et al., 2004), the mean copper concentrations from WWTPs between 2000 and 2002 ranged between 20 and 40 times the 95% species protection trigger value. For lead, on average WWTP discharges were below the 95% trigger level, and for zinc, mean concentrations were between approximately 3.5 and 5.5 times the 95% trigger value.

Table 3.6. Mean and median concentrations of three heavy metals in flow proportional composite samples from the central metropolitan creeks (Note: figures presented are for acid digested samples and represent total concentrations)..

Location	Period	Statistic	Mean	Cu	Pb	Zn
			Flow MI/d			
Torrens	1996-2005	median	19.5	0.010	0.009	0.063
Torrens	1996-2005	mean	89.0	0.015	0.013	0.075
Brownhill	1996-2005	median	8.9	0.014	0.010	0.113
Brownhill	1996-2005	mean	32.8	0.018	0.015	0.130
Sturt	1994-2005	median	6.3	0.015	0.007	0.089
Sturt	1994-2005	mean	55.6	0.018	0.014	0.099
Field	2001-05	median	1.5	0.007	0.006	0.028
Field	2001-05	mean	9.40	0.012	0.008	0.037
Christie	2001-05	median	0.7	0.008	0.007	0.045
Christie	2001-05	mean	7.67	0.011	0.009	0.055

Table 3.7. Mean and median concentrations of three heavy metals in grab samples collected as part of the OCWMB ambient sampling program between 1999 and 2004 (Note: figures presented are for acid digested samples and represent total concentrations).

Location	Statistic	Cu	Pb	Zn
		mg/L		
Field	median	0.003	0.006	0.053
Field	mean	0.004	0.008	0.104
Christie	median	0.004	0.008	0.093
Christie	mean	0.007	0.009	0.097
Onka ON	median	0.006	0.002	0.079
Onka ON	mean	0.006	0.002	0.140
Onka Est	median	<0.001	<0.001	<0.010
Onka Est	mean	<0.001	<0.001	<0.010
Peddler	median	0.005	0.002	0.079
Peddler	mean	0.008	0.002	0.094
Willunga	median	0.004	0.003	0.158
Willunga	mean	0.005	0.003	0.619
Maslin	median	<0.001	0.001	0.257
Maslin	mean	<0.001	0.002	0.385
Aldinga	median	0.002	0.002	0.089
Aldinga	mean	0.003	0.003	0.108
Sellicks	median	0.003	0.003	0.092
Sellicks	mean	0.007	0.006	0.150

In stormwaters the situation is quite different (Table 3.6 and 3.7). Mean and median total concentrations of the three metals in most stormwaters exceed the 95% of species protection trigger value. The concentrations are highest in stormwaters from catchments with a greater degree of urbanisation. Median copper concentrations are between 2 and 10 times the 95% trigger value. Median lead concentrations in stormwaters from the urbanised catchments are between 1.5 and 4 times the 95% trigger values. In the largely rural southern creeks lead concentrations were less than the 95% trigger value. The figures for lead are slightly misleading, because the concentrations have fallen consistently since the mid-1990s (Figure 3.7, Table 3.8).

The median metals concentrations in the southern creeks (Table 3.7) from Peddler Creek south represent samples collected from 1999 onwards, well into the decline in lead concentrations observed in the more urbanised catchments further north. In the urbanised Field River and Christies Creek the grab sampled lead median concentrations are higher than the more rural urban sites and the values are consistent with the later years presented in Table 3.7 for the central metropolitan creeks.

Table 3.8 and Figure 3.7 demonstrate the decline in lead concentration in stormwater. The reduction in lead in air is well documented and is associated with the reduction in use of leaded petrol fuel products by motorists (Figures 3.9 and 3.9). The percentage reductions in the central metropolitan creeks range from a 65 % to 83 % reduction in median lead concentrations from the peak annual median for each site, for the available record. Figure 3.8 indicates a reduction in sales of leaded petrol from 1987 to 2002 of 92 %, and the reduction in airborne lead indicated in Figure 3.9 between 1995 and 2000 is 73.5 %.

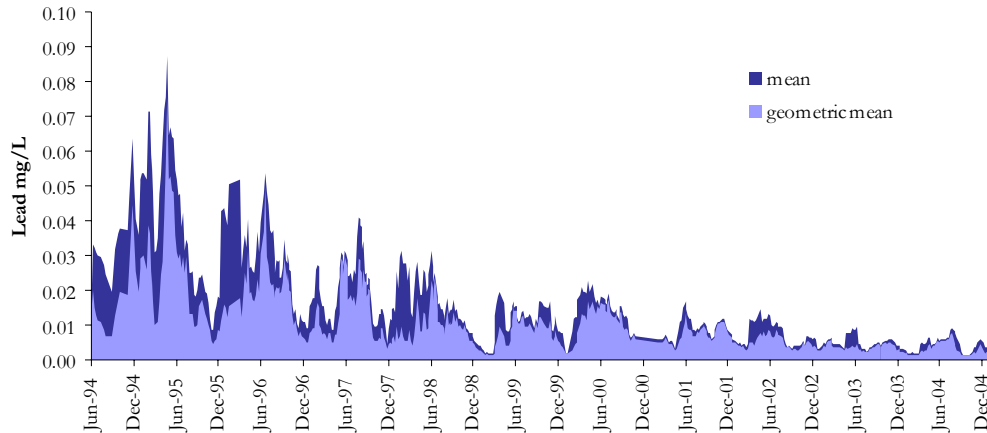


Figure 3.7. Two-month running average lead concentration in Sturt River at Anzac Highway since June 1994 (note that this is total lead in acid digested samples).

Zinc concentrations have also reduced, but to a lesser extent than lead (around 50 to 60 % of maximum annual median, Table 3.8). Copper levels in stormwater have varied from year to year but there has been no reduction in concentrations.

Table 3.8. Annual median concentrations of total lead and total zinc in flow proportional composite samples from the central metropolitan creeks. Percentage change in concentration is relative to the year of maximum concentrations for each site and variable.

Lead	Torrens			Brownhill			Sturt		
	Median, mg/L	% change	Count	Median, mg/L	%change	Count	Median, mg/L	% change	Count
1994							0.013	-35.0	25
1995							0.020	0.0	53
1996	0.016	0.0	37				0.018	-12.5	52
1997	0.013	-18.8	65	0.023	0.0	52	0.015	-25.0	68
1998	0.014	-12.5	57	0.017	-26.1	47	0.010	-50.0	61
1999	0.011	-31.3	57	0.012	-50.0	48	0.008	-62.5	60
2000	0.010	-37.5	43	0.011	-52.2	38	0.013	-37.5	46
2001	0.007	-57.5	50	0.009	-61.3	49	0.007	-62.8	50
2002	0.004	-73.1	52	0.007	-69.1	42	0.004	-79.0	55
2003	0.005	-68.1	50	0.005	-76.7	40	0.004	-80.3	50
2004	0.006	-65.0	27	0.006	-73.0	44	0.003	-83.0	52
Zinc	Median, mg/L	% change	Count	Median, mg/L	% change	Count	Median, mg/L	% change	Count
1996	0.101	-5.6	37	0.116	-34.1	25	0.109	-20.8	52
1997	0.101	-5.6	65	0.176	0.0	52	0.118	-13.9	68
1998	0.107	0.0	57	0.168	-4.5	47	0.137	0.0	61
1999	0.088	-17.8	57	0.132	-25.0	52	0.102	-25.5	59
2000	0.071	-33.6	53	0.107	-39.5	47	0.106	-22.6	42
2001	0.049	-54.2	50	0.083	-52.8	52	0.077	-43.8	48
2002	0.037	-65.4	52	0.090	-48.9	42	0.063	-54.0	52
2003	0.051	-52.3	50	0.082	-53.7	40	0.048	-65.3	50
2004	0.039	-64.0	52	0.096	-45.5	44	0.057	-58.8	52

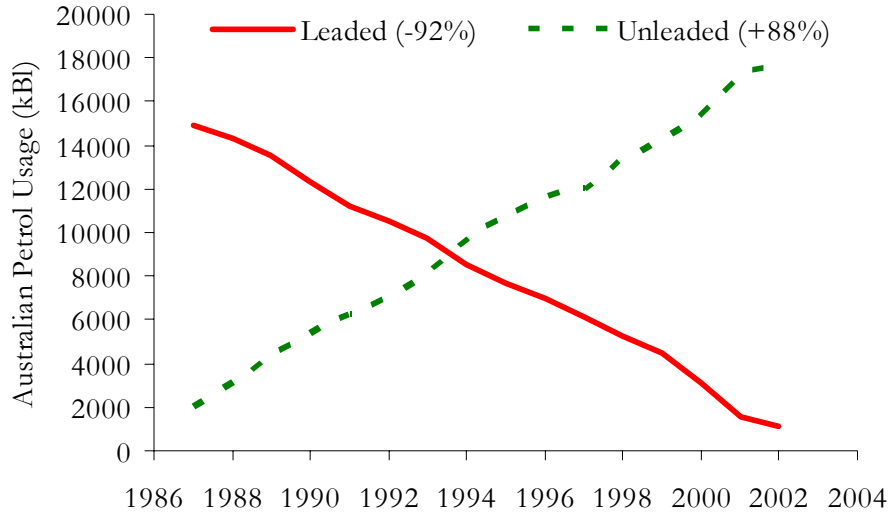


Figure 3.8. Reduction in sales of leaded petrol in Australia from 1987 to 2002 (data from Australian Institute of Petroleum).

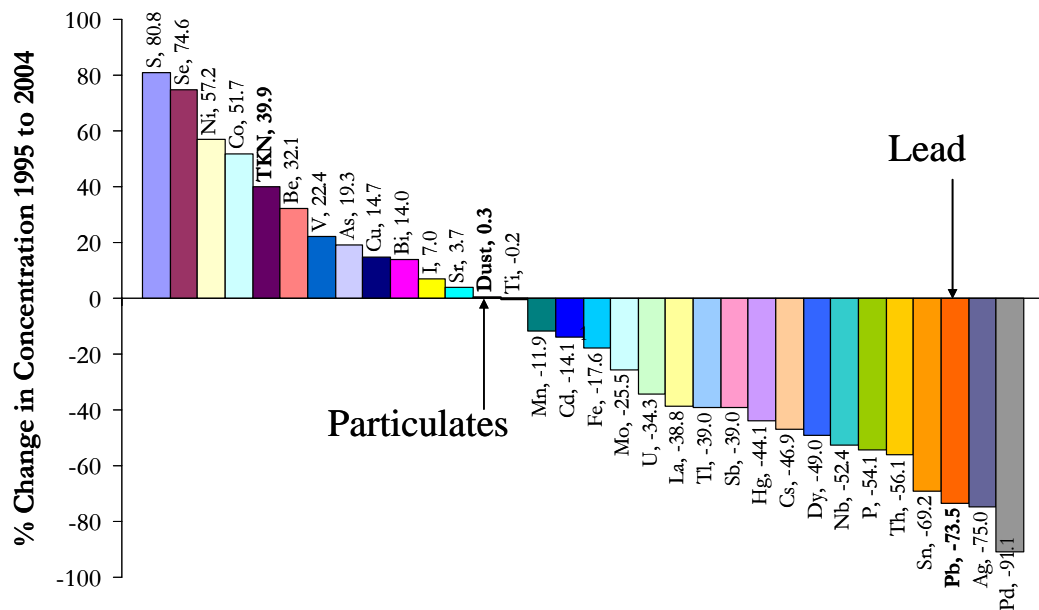


Figure 3.9. Changes in atmospheric concentrations of selected elements (bulked data for 1995 to the end of 1999 and for 2000 to 2004 are the two periods represented).

The median concentrations presented in Tables 3.7 and 3.8 demonstrate that lead concentrations are around the 4.4 $\mu\text{g/L}$ 95% species protection trigger values in recent years. In the early nineteen nineties median lead concentrations were between 4 and 6 times the 95% species protection trigger value. The highest annual median concentrations of zinc were between 7 and 11 times the 95% species protection trigger values in the mid nineteen nineties. Current median annual zinc concentrations are still in excess of the 95% trigger value (Table 3.8). Most notably Maslin Creek has a high zinc concentration relative to its lead and copper concentrations. Of the 16 samples analysed the maximum zinc value was 1.1 mg/L and there were three samples with concentrations in excess of 0.5 mg/L. These high values might warrant further investigation.

4. Load estimation for stormwaters

This chapter gives a brief overview of the load estimation techniques used for long-term and daily loads of suspended matter and nutrients in stormwater. The loads are presented in Chapter 5 and the load discharge relationships or observed loads have been used in the coastal circulation modelling undertaken by UWA.

There are dynamical differences in inputs of stormwater and recycled water. These differences are obvious, yet worth restating for readers with little knowledge in this field. Recycled water, i.e. effluents discharged by waste water treatment plants, discharges at a relatively steady rate throughout the year which means that these sources provide a continuous dose of contaminants to the receiving water – the coastal waters of Adelaide. Stormwater runoff is very different in nature, it is episodic, i.e. it is transient, effectively short duration spikes of water that flood over the near shore seawater.

4.1 Load-discharge relationships: ACWS and GSV

Load-discharge relationships for the stormwater inputs to the ACWS area were estimated for two purposes, firstly to provide estimates of loads for sites without flow proportional composite sampling, and secondly to enable modelling studies by providing time-series of loads to the coastal zone to investigate pollutant transport and dispersion.

Load discharge relationships for the Central Zone stormwater sources for a variety of dissolved and suspended constituents are presented below. The relationships have been estimated for the major creeks and the South Western suburbs storm drains (the Ocean Catchment of the Patawalonga). The relationships are used subsequently to estimate overall monthly loads of pollutants where actual observations are not available. Between 1972 and 1983 the Gulf St Vincent Water Pollution Studies program operated by the EW&S department (Steffensen, 1985) carried-out a similar, but longer running, programme of sampling to that of the ACWS Input Studies. These data have been collated and digitised and either measured or modelled flows have been used to re-create load/discharge relationships and estimate longer term loads. In recent times (since 1996) composite flow proportional sampling of Brownhill Creek, Sturt River and the River Torrens has provided detailed time series of concentrations and loads of pollutants, the load discharge relationships for these programs have been estimated, but actual data have been used to present monthly mean loads.

4.1.1 Flow-proportional composite data

The flow-proportional composite sampling based data do not provide instantaneous loads. The loads calculated are based on the total flow for the period between sampling multiplied by the concentration of the composite sample. To produce the load discharge relationships for the sites with this kind of monitoring the total volumes and loads were divided by the number of days over which each composite sample was collected (usually 7, but occasionally more or less). The load discharge relationship produced gives a mean daily load from a mean daily volume and can be used to estimate loads for a given flow condition (see Appendix IV), with the understanding that this is not an instantaneous load and is not directly comparable with a grab sample based load discharge relationship. In Chapter 5 the actual weekly loads were used to calculate the total loads of contaminants to the coastal zone (Section 4.2).

4.1.2 Grab sampled data

Two types of grab sample data have been used in the IS1 investigations. The first type is ambient sampling data collected by the OCWMB, which were approximately monthly samples collected throughout the year between 1999 and the end of 2004. Data are available for the main creeks from Field River in the north to Sellicks Creek in the south. The second type of data available was derived from event based grab sampling of the Patawalonga Ocean Catchment

(Holdfast Drains, or South Western Drainage Scheme). Due to the nature of these sources, they would either be flowing in response to a rainfall event, or dry (with the exception of the Broadway Drain which appears to intercept the shallow groundwater table). In addition, storm-event response samples were collected from the Southern Creeks and these data were used to augment the ambient data in generating load discharge relationships.

The second aspect of generating load discharge relationships for the grab sampled stormwater sources was the presence or absence of measured flows. Report TR10 (Wilkinson 2005) describes the modelling procedure used to derive flows for both gauged and ungauged sites. For the Southern Creeks the proportions of residential land area were used to adjust the observed flow in Pedler Creek to give estimated flow time series for the ungauged creeks (Section 5.6 below). With modelled flows the “numerical” flow declines in an exponential manner, i.e. it never reaches zero, which is unrealistic because at some low flow the creeks no longer break-out to sea. For this reason an arbitrary bottom cut off of 20,000 L/day was used for the load discharge relationships.

Appendix IV presents the tables of load discharge relationships and the graphical representations for each relationship. The mean R^2 of the aQ^b load discharge relationships was 0.845 for suspended solids, 0.923 for TKN, 0.854 for $\text{NO}_x\text{-N}$ and 0.923 for total phosphorus. The data are presented in log-log plots in order to demonstrate the full range of the data and demonstrate good straight-line relationships.

4.1.3 Sites with poor data coverage

The creeks and rivers of the Northern and Barker Inlet zone have sparse water quality data coverage. A small number of samples analyses exist for some of these sites spread over a long period and taken at irregular intervals, during which time changes in water quality relating to land-use practices, industrial activity and maturation of once newly developed residential areas mean that the comparability of these data as representative of current conditions is questionable. Recently commenced flow proportional monitoring of these sites will render accurate loads. Until these data become available the methodology adopted to estimate loads for these sites has been to take load discharge relationships for adjacent or similar locations and adjust them to the flow conditions at that site. This was achieved by assuming a mean concentration for each determinant at each site and the mean observed or modelled flow for the most recent ten years, then calculating the load for that mean flow. Since,

$$\text{Load} = aQ^b$$

the load is now known for the given mean discharge Q , and b is chosen as the mean of the slopes of the known relationships. The y-axis intercept (the notional concentration in mg/L at 1 ML/d), can then be calculated by rearranging for a :

$$a = \text{Load}/Q^b .$$

The calculated “ a ” values for Gawler River to the Port Catchment are presented in Table A4.2 in Appendix IV.

4.2 Load Calculation Methods

Using flow proportional composite data and gauged flows to calculate loads of contaminants to sea gives a relatively accurate and true time variable input. Where these data were available, they have been used to produce the loads used in Chapter 5 of the report. The load discharge relationships effectively remove the year to year, day to day variations in the relationship between concentration and flow by fitting a curve to all of the observed data. For the grab

sampled data this was unavoidable. Thus the year to year variation in load reflects the changes in flow only, because the load discharge relationship is fixed.

1. **Ocean catchment storm drains**

Loads of contaminants from these storm drains were estimated using modelled flow estimates as detailed in Wilkinson (2005) and stormwater samples collected between October 2003 and November 2004. Samples were analysed for a range of water quality determinants as detailed in Appendix V. Since these drains were very similar in nature and location the data for each of the drains was lumped together and flow weighted means for each event were calculated such that,

$$\bar{C} = \frac{\sum C_i \cdot Q_i}{\sum Q_i}$$

where C_i is the concentration at a site, and Q_i is the flow at that site at the time of sampling.

The instantaneous load is then $\bar{C} \cdot \sum Q_i$ which is the same as $\sum C_i \cdot Q_i$, and is plotted against $\sum Q_i$ on the x-axis and fitted with an appropriate regression equation as shown in Appendix IV.

2. **Southern Zone creeks**

Each of the Southern Creek Loads was estimated using the load discharge relationships presented in Appendix IV.

3. **Northern and Barker Inlet**

Basic annual loads for the Northern inputs were estimated using mean annual flow volume multiplied by the mean concentration. Estimated load discharge relationships were used for modelling coastal transport and dispersion. The estimated flows were those derived from the work presented in TR10 (Wilkinson, 2005). Reality checking of the basic annual loads was carried-out by comparing catchment yields as described in Section 5.5.

5. Loads of nutrients, suspended matter and heavy metals

The following sections provide an overview of the major inputs of nutrients to the ACWS coastal zone. These are presented as the overall loads from the four IS1 sub-programs, inputs from the WWTPs and stormwaters presented in geographical order, and seasonal variations in the major stormwater inputs from the central study zone.

5.1 Source by source mean annual loads

Table 5.1 and Figure 5.3 present estimates of flows and loads to the ACWS study zone from WWTP discharges and stormwater inputs. Table 5.1 gives a source by source inventory of inputs in a north to south progression, starting with the Gawler River and moving southwards to the grouped “Southern Creeks” (Peddler Creek to Sellicks Creek). This table includes flow as a percentage of the total, the contributing catchment area and the runoff yield in mm, all loads are in metric tonnes per year. The aim of the table is to give a “one-stop” overview of the key inputs. In addition to source by source estimates of load, the table provides overall totals, the split between WWTP and stormwater sources, and a zonation of sources into five main geographic blocks coinciding with distinct clusters sources. The zones are as follows (see Figure 5.1a):

1. Northern: Gawler River, Bolivar WWTP, Smith Creek, and Barker Inlet; all terrestrial discharges physically within the mouth of Barker Inlet.
2. Central: all major discharges from Largs to Marino.
3. Southern: All remaining major discharges from Marino to Sellicks Creek.
4. Central offshore zone.
5. Port Adelaide sludge outfall recovery zone.

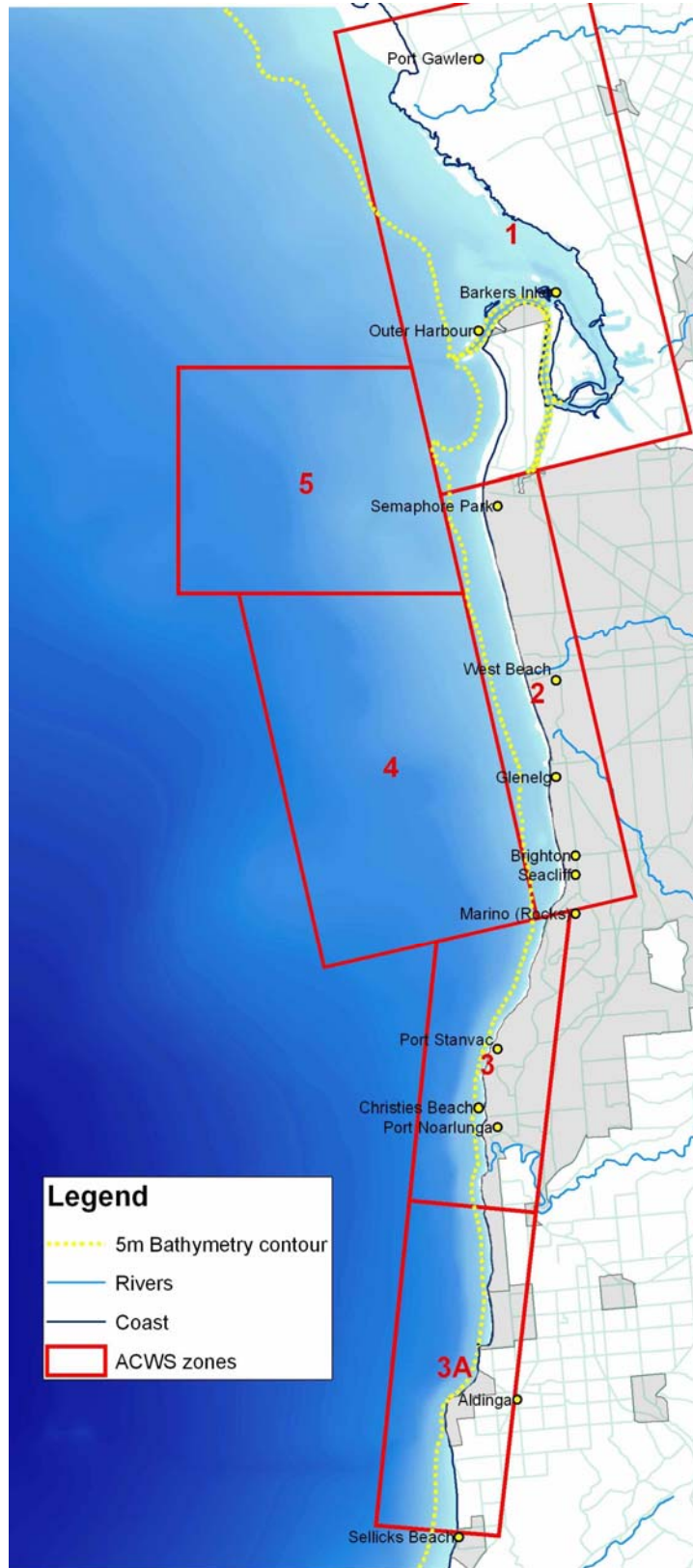


Figure 5.1a The ACWS study area with zonation relating to input groups.

Runoff yield estimates are also included for the total flow and for the four geographical clusters. These yield values include the WWTP flows, which rather than rendering the value meaningless it gives an indication of the impact of WWTP flows in increasing the effective yield of water derived from the terrestrial zone. For example, in the Central Zone where the degree of urbanisation is greatest, the combined stormwater runoff and WWTP discharge from Glenelg is equivalent to a yield (runoff per unit contributing catchment area) of 135 mm. Elsewhere within the ACWS study area the combined yield is around half that for the Central Zone or less. Apart from nitrogen into Barker Inlet from the Penrice Plant, the Central Zone from the Torrens to the Barcoo Outlet of the Patawalonga catchment, a 3-km strip, is a very concentrated zone of receiving water.

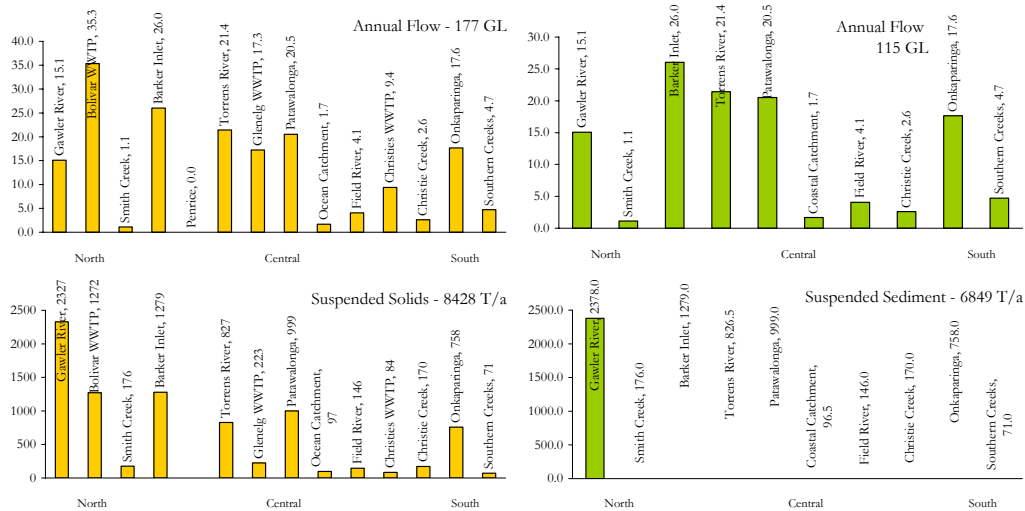


Figure 5.1. Bar charts of contemporary WWTP and stormwater and stormwater only annual flow volume and suspended solids.

It was shown in Section 1.1 that stormwater input of suspended material is dominant overall. Figure 5.1 shows the individual inputs. Bolivar contributes significantly to the suspended load at around 1300 tonnes a year. The nature of this load is quite different to that from the major creeks, in that the mineral component is almost absent, the load is composed of organically derived particles (it is predominantly a volatile suspended load). The right-hand plot in Figure 5.1 shows the stormwaters only. The suspended load from the major creeks (not surprisingly) dominates the input to the coastal zone. The load for the Gawler River is estimated and further investigation of the load from the Gawler River needs to be undertaken should a reliable estimate be required in the future. The loads from the Torrens River and Patawalonga are based on sound monitoring and measured flows and give a reliable indication of the major loads to the central zone. Section 5.2 presents seasonal variations in stormwater suspended and nutrient loads and provides estimates of yield. Yield estimates (kg/ha/y) provide a useful indication of whether a particular source is suffering accelerated erosion compared to other sites, or excessive nutrient transport (they can also highlight data discrepancies as was the case in Christie Creek (Wilkinson et al., 2005)).

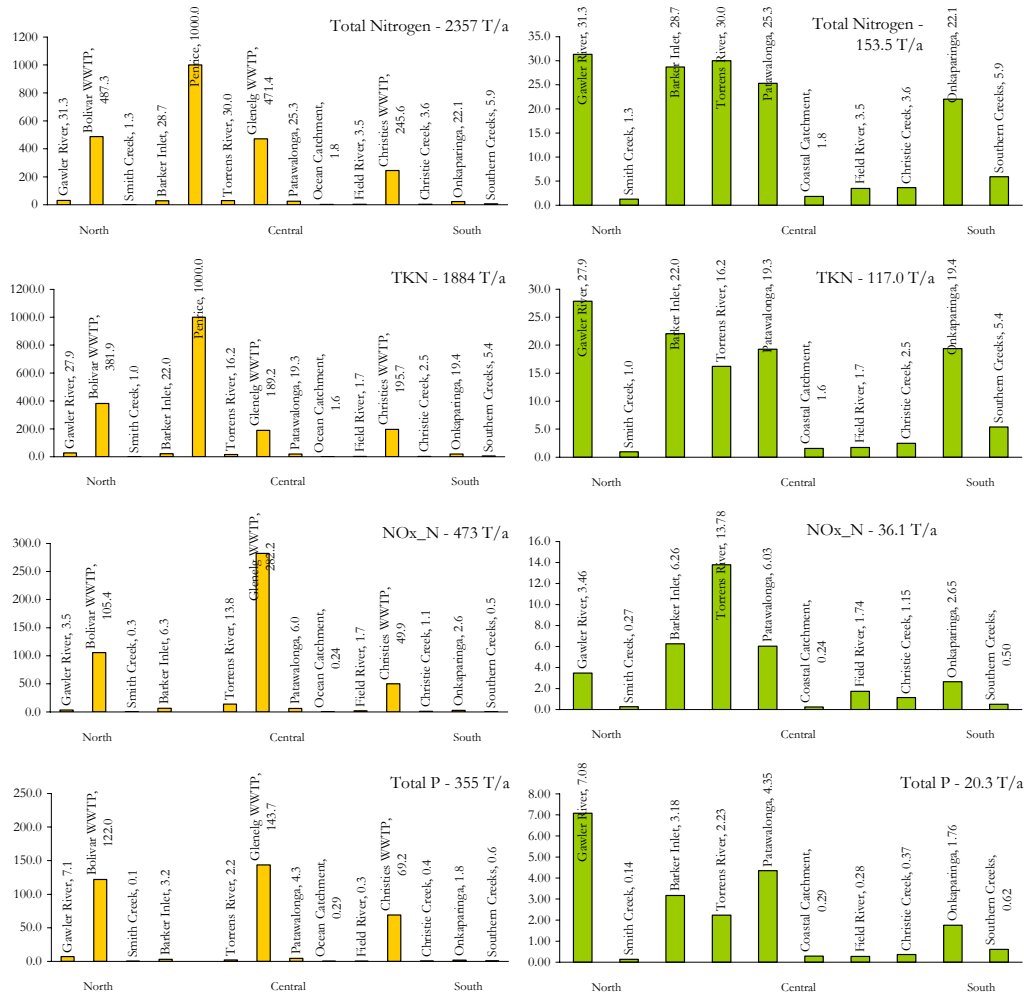


Figure 5.2: Bar charts of contemporary WWTP and stormwater, and stormwater-only annual nitrogen and phosphorus loads.

Figure 5.2 presents location by location nutrient loads for WWTP discharges and stormwater inputs. As indicated in Section 1.1 Penrice dominates the nitrogen input and individually in the northern, central and southern zones each of the WWTPs dominates local nitrogen loadings. The Glenelg WWTP singly accounts for 60 % of total WWTP and stormwater $\text{NO}_x\text{-N}$, and of the stormwaters the Torrens River dominates $\text{NO}_x\text{-N}$. Thus, the largest WWTP and stormwater inputs of $\text{NO}_x\text{-N}$ are discharged into the central zone. Stormwater TKN discharges are consistent from site to site with the contributing catchment.

WWTP discharges dominate phosphorus inputs in a similar way to total nitrogen, the phosphorus output from the Patawalonga is elevated at around $25 \text{ g/m}^2/\text{y}$ (in Sturt Creek) compared to other creeks which have around $15 \text{ g/m}^2/\text{y}$. A discussion of inter-annual variation and yields of pollutants is presented in Section 5.3.

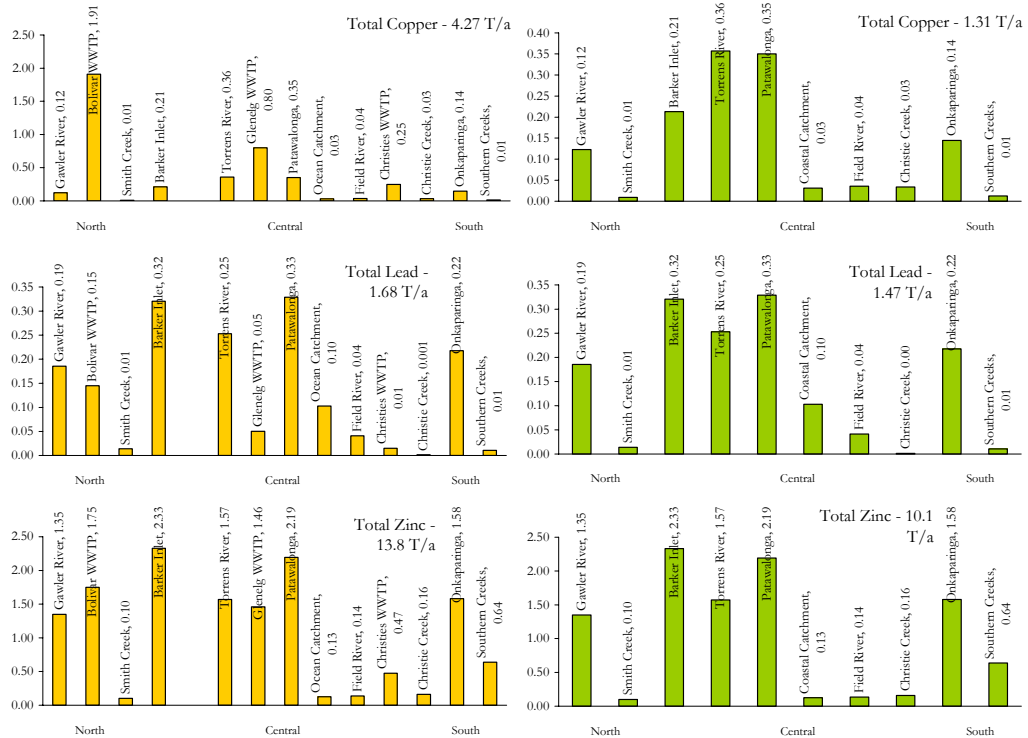


Figure 5.3. Bar charts of contemporary WWTP and stormwater and stormwater only annual copper, lead and zinc.

Figure 5.3 presents loads for the three dominant heavy metals in wastewater and stormwater; copper, lead and zinc. Bolivar WWTP clearly dominates the output of copper, at around 20% of the load. The Torrens and Patawalonga dominate the stormwater load of copper. As indicated in Section 1.1 stormwater dominates the input of lead and zinc overall, however, the individual loadings of zinc from both stormwaters and WWTPs are comparable in magnitude - even if the nature of the inputs is dramatically different, i.e. WWTPs discharge at a consistent rate, whereas stormflows are sporadic and of high individual magnitude. The nitrogen output of the Torrens River has been examined in terms of comparing peak loads to long term WWTP output (Section 5.2). This kind of analysis could be extended to stormwater metals to establish whether intense storm discharges of zinc are a serious transient threat to marine life.

5.2 Seasonal and temporal variation in stormwater and WWTP loads

ACWS Technical Report Number 1 detailed the variations in WWTP discharges seasonally and over longer time scales, i.e. the differences in loads of contaminants at certain key times were presented. This section of this report presents the seasonal variation in both WWTP and stormwater loads and also demonstrates the nature of short term variations in stormwater loads compared to WWTP discharges which tend to vary more gradually and in a more predictable way. Nutrient inputs and suspended load are considered separately since their behaviour and the way they impact the nearshore zone is quite different.

5.2.1 The nutrients – nitrogen

Nitrogen has long been held as the key limiting nutrient in the nearshore seagrass ecology of Gulf St Vincent (Steffensen et al., 1989). The following figures present recent data demonstrating the month by month variations in nitrogen load from Bolivar, Glenelg and Christies Beach WWTPs and from the Torrens and Patawalonga (Figure 5.4). It is important to

restate here that the nature of the Torrens and Patawalonga discharges are subtly different. The Torrens discharges high nitrate nitrogen, whereas the Patawalonga discharges largely organic nitrogen as quantified by TKN. In addition the Patawalonga discharges an elevated phosphorus load due to the Heathfield WWTP effluent. In Figure 5.4, the total equivalent daily load of nitrogen is represented, i.e. the data are stacked, they are added, one upon another. The strong seasonal variation in the Bolivar discharge is in part a consequence of the net gain or loss of volume due to evaporation from or rainfall into the tertiary lagoons. The discharges from Christies Beach and Glenelg are less seasonally variable, although as highlighted in TR1, Christies Beach now diverts at least 20% of its annual flow to the Maclaren Vale viticulturalists.

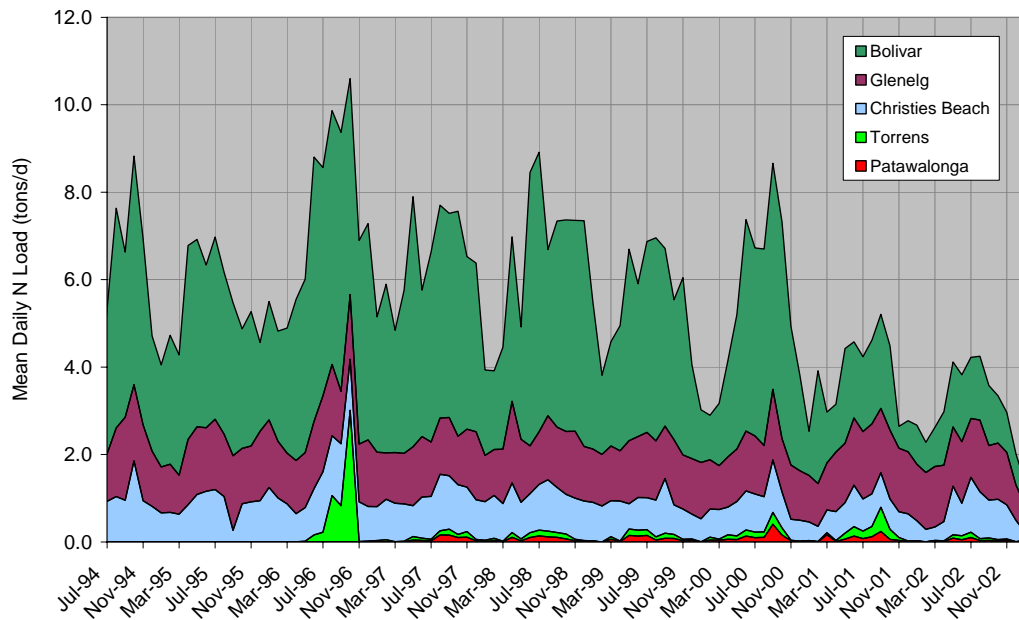


Figure 5.4. Mean daily loads of nitrogen discharged to sea. The data are stacked to show the total daily load and nitrogen discharges from the Torrens River and the Patawalonga system are included to demonstrate the relative magnitudes of these inputs.

The WWTP nutrient input dominates the loading to GSV, yet on occasions, such as the events of winter 1996, the stormwaters may in a short space of time discharge a load of nitrogen equivalent or greater than that from Glenelg WWTP (only data for Torrens River available). Improvements at the WWTPs have significantly reduced the overall load of nitrogen to GSV since the mid-nineteen nineties:

- Bolivar
 - **72 % reduction in N**, from 1350 to 380 t/y
 - **48 % reduction in P** since 1996/7
 - High salinity plant will add 100 tN/y and 77 tP/y
- Glenelg
 - **37 % reduction in N**
 - 70 % when IFAS fully optimised
 - P unchanged
- Christies Beach
 - **39 % reduction in N** since 1996/7
 - 64 % reduction in N when IFAS fully optimised
 - 23 % reduction in P since 1996/7

Figure 5.5 demonstrates the nature of the Torrens River nitrogen discharge. Since flow proportional monitoring commenced in 1996, 22.5 % of the total nitrogen load from the Torrens was discharged in a single week, 62 % of the total load was discharged in 10 % of the entire period of record, and 95 % of the total load has been discharged in 50 % of the monitoring period. The latter fact is not surprising and is a simple function of the wet winter/dry summer seasonality of stormwater. The other statistics clearly demonstrate the episodic nature of stormwater loads and the dramatic impact a small number of extreme events may have. The nitrogen concentration associated with each load value presented in Figure 5.5 is also given and that demonstrates that mean weekly nitrogen rarely rises above 3.5 mg/L except during the most extreme of events.

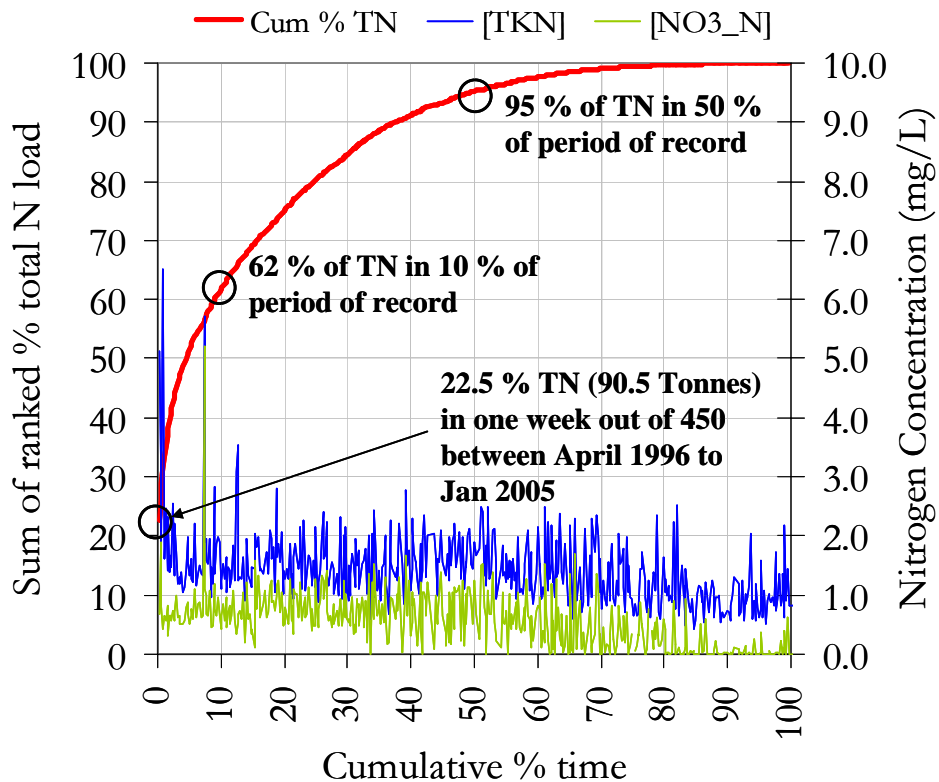


Figure 5.5. Load duration curve for nitrogen from Torrens River.

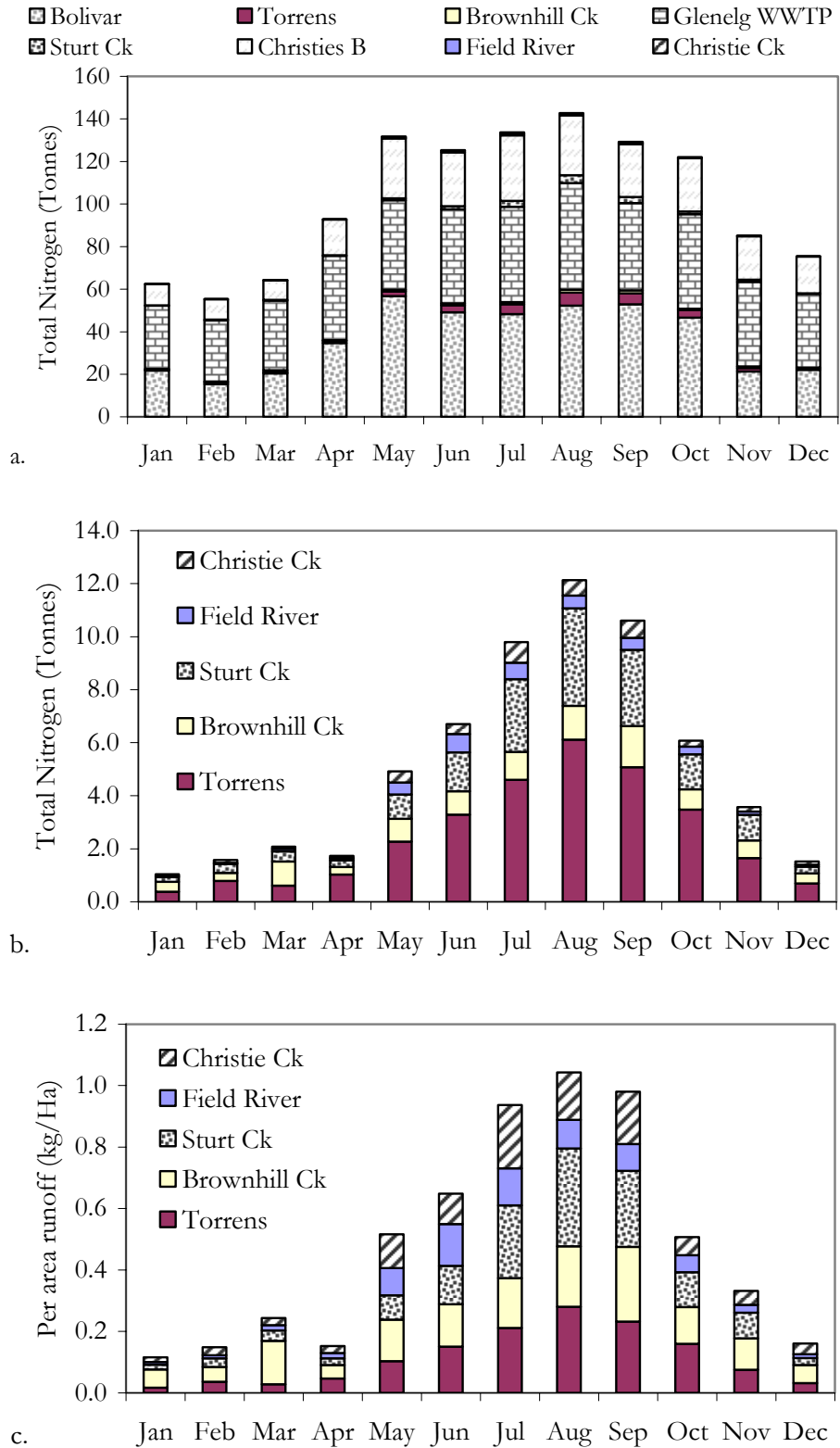
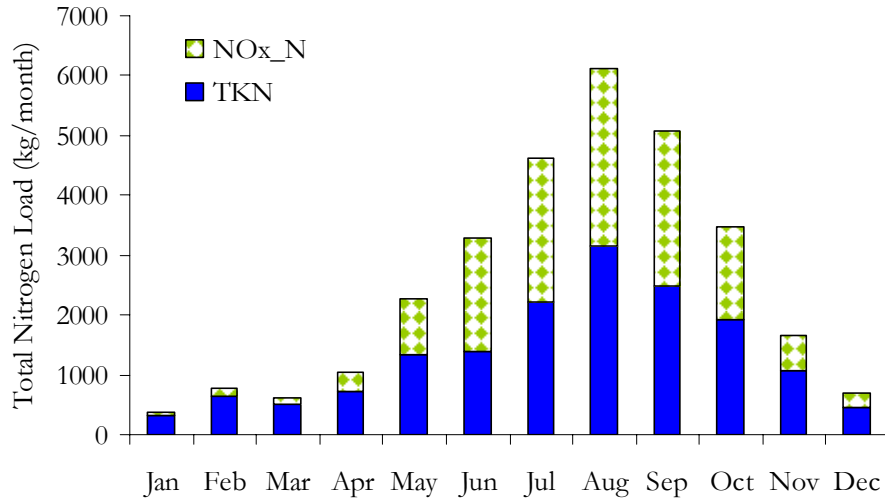
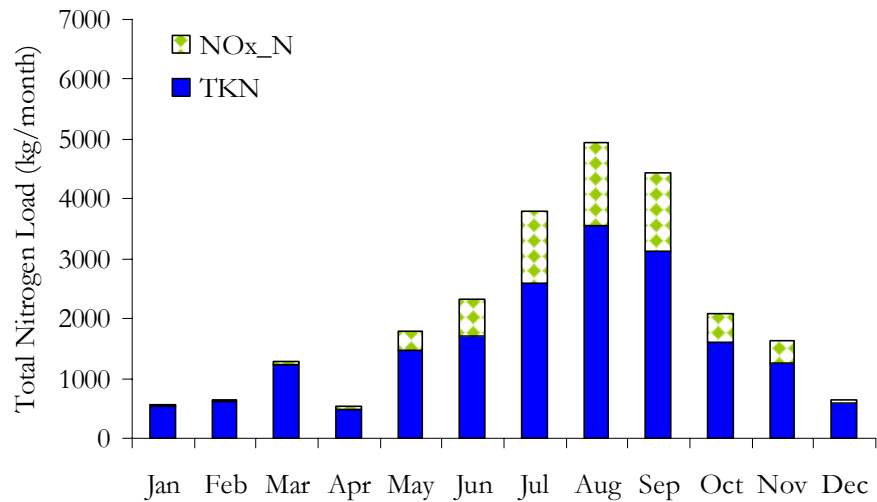


Figure 5.6. Monthly total nitrogen loads for a. WWTP and stormwater sources, b. stormwaters, with flow proportional monitoring, and c. nitrogen yield for the stormwaters in b.

Figure 5.6a shows total nitrogen loads for the major Metropolitan Adelaide WWTPs and the well instrumented stormwater sources, i.e. those with flow proportional monitoring. Clearly the WWTPs dominate nitrogen loadings to the coast as highlighted in Section 5.1 above. The WWTP nitrogen loads vary seasonally and are greatest in the winter months (May to October). During the summer Glenelg is the single greatest source of nitrogen, the loads from Christies Beach and Bolivar are reduced by the transfers to MacLaren Vale and Virginia, respectively. Variations in the component nitrogen species, i.e. ammonia nitrogen, TKN and oxidised nitrogen show similar seasonal behaviour so plots are not provided thus avoiding duplication. To reiterate from Wilkinson et al. (2005a), Bolivar and Glenelg discharge mainly nitrate nitrogen, Christies Beach continues to discharge ammonia nitrogen.



a.



b.

Figure 5.7. Total nitrogen from a. Torrens River and b. Patawalonga system, showing TKN and oxidised nitrogen composition.

The stormwaters predominantly discharge TKN, with the exceptions of the Torrens River, Field River and Christie Creek, which discharge up to 50% oxidised nitrogen (Figure 5.7a). In Figure 5.7b, the Patawalonga is shown demonstrating the lower oxidised nitrogen component (see Section 3.3).

Of the stormwater sources presented in Figure 5.6a, the Torrens River discharges the greatest quantity of nitrogen (50% split between TKN and oxidised nitrogen during the winter and predominantly TKN in the summer).

The nitrogen yield from stormwater sources or load of nitrogen derived per unit catchment area is fairly consistent between sites (Figure 5.7c), although the Sturt Creek and Torrens River yield the greatest nitrogen load during winter months.

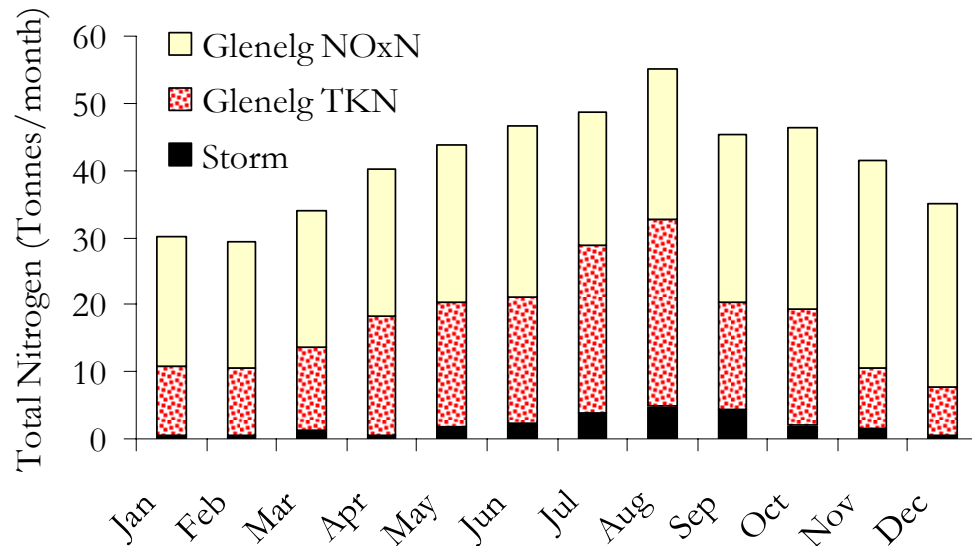


Figure 5.8. Total nitrogen loading to the Central Zone from WWTP and stormwater discharges (Glenelg WWTP, Torrens River and The Patawalonga System).

Figure 5.8 shows the summed nitrogen load for the ACWS central zone, the Glenelg discharge clearly overwhelms any stormwater input by a factor of 50 in the summer and at least 10 in the winter months.

5.2.2 The nutrients – phosphorus

The pattern of seasonal variation in phosphorus loads is similar to that for nitrogen and is dominated by WWTP inputs, however, the seasonality is not as strong (compare Figure 5.6a and 5.9a). Since the volume of flow from Bolivar and Christie Beach are reduced by the transfers for irrigation, the summer phosphorus concentration must result from elevated concentrations leaving the WWTPs. This is the case for Glenelg WWTP, where phosphorus concentrations are between 1 and 2 mg/L higher in summer. Phosphorus loads from stormwater do not increase as gradually as they do for nitrogen as the season progresses from autumn to winter. Phosphorus appears to increase in a number of steps (Figure 5.9b). These steps are a function of the nature of phosphorus which in natural waters tends to bind to clay minerals. In Appendix III, total phosphorus is shown to have a weak correlation with suspended sediment in all creeks. Nitrogen in natural waters is predominantly in the dissolved phase and thus behaves differently to phosphorus. Phosphorus is effectively flushed from the stormwater system increasingly with higher flows, thus phosphorus discharges are more extremely episodic than nitrogen. Figure 5.10 shows that over 80 % of total phosphorus was discharged from the River Torrens in only 10 % of the period of observation (from April 1996 to January 2005).

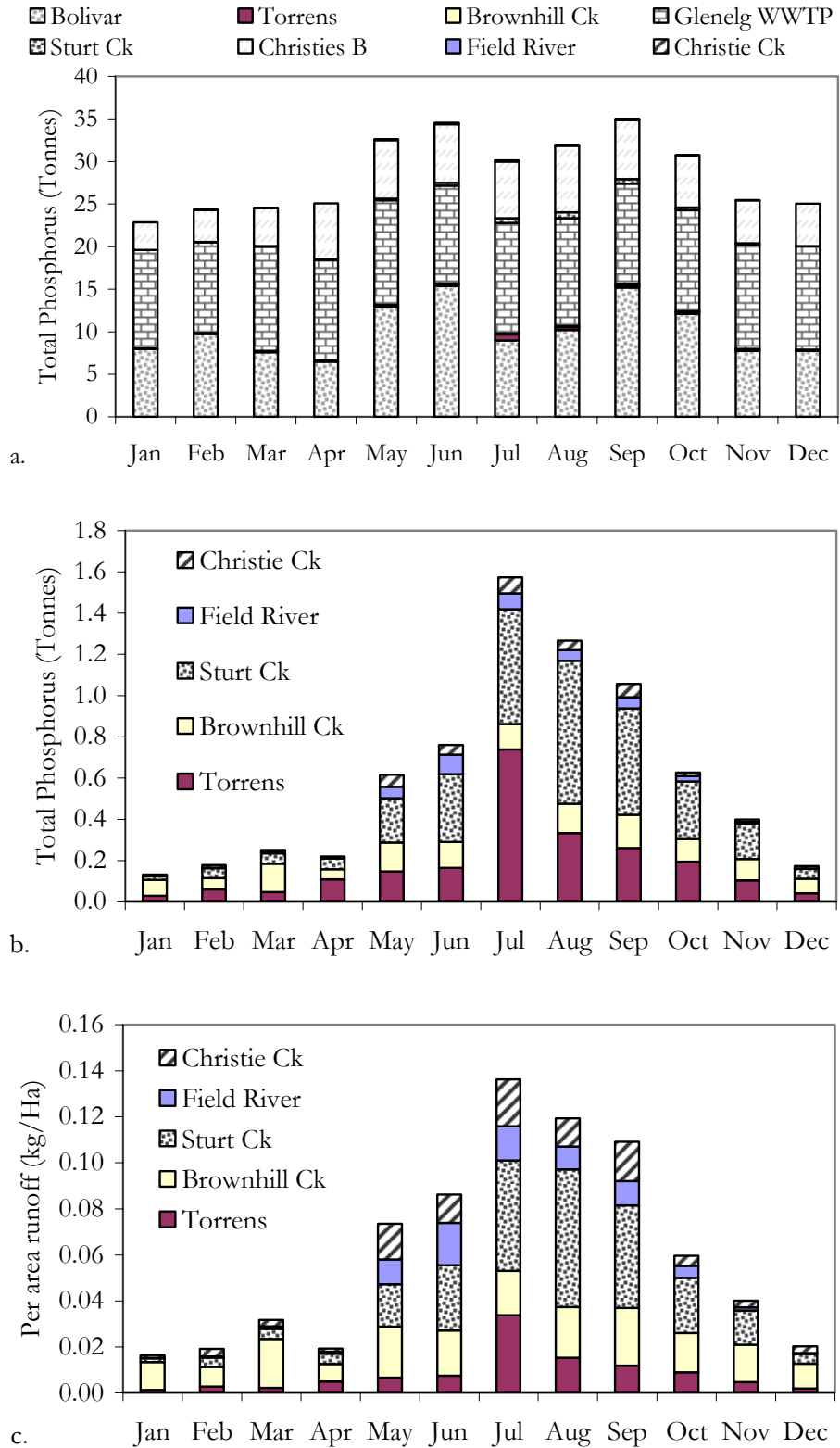


Figure 5.9. Monthly total phosphorus a. WWTTP and stormwater sources, b. stormwaters, with flow proportional monitoring, and c. phosphorus yield for the stormwaters in b.

During one week 39% of the total phosphorus was discharged, while only 22.5 % of the total nitrogen was discharged in the same event (Figure 5.5). The dotted line in Figure 5.10 shows the load duration curve for phosphorus without the extreme event of 1996, which demonstrates how much one extreme event can change the statistics.

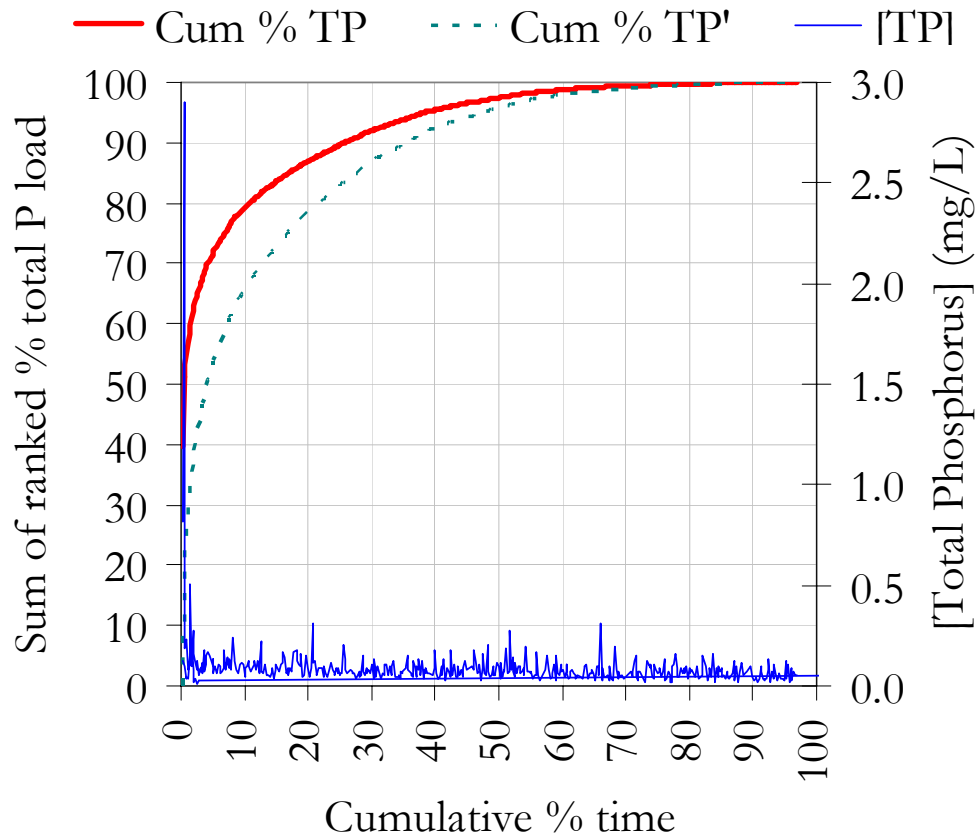


Figure 5.10. Load duration curve for total phosphorus from the River Torrens from April 1996 to January 2005.

Figure 5.9c presents the phosphorus yield for the flow proportionally monitored stormwater sources, showing that Sturt Creek yields a greater load of phosphorus than the other creeks. This is not surprising and it is well known that the Heathfield WWTP impacts on the phosphorus load in Sturt Creek. In relation to loads of phosphorus to the coastal zone, this input is negligible compared to that from Glenelg WWTP.

5.2.3 Suspended load

The suspended load from the Torrens River is even more episodic than the phosphorus load (Figures 5.11 and 5.12). Figure 5.12 shows the monthly sediment load from the three WWTPs and the Torrens and Patawalonga. The peak load from the Torrens was in excess of 20,000 tonnes of suspended load in 1 week, this was 63% of the total load between April 1996 and January 2005. Of the total suspended load from the River Torrens, 90% was discharged in 10% of the monitoring period. Bolivar discharges a high load of suspended matter, but as previously stated, this is predominantly algal and other biogenic particles derived from within the tertiary lagoons.

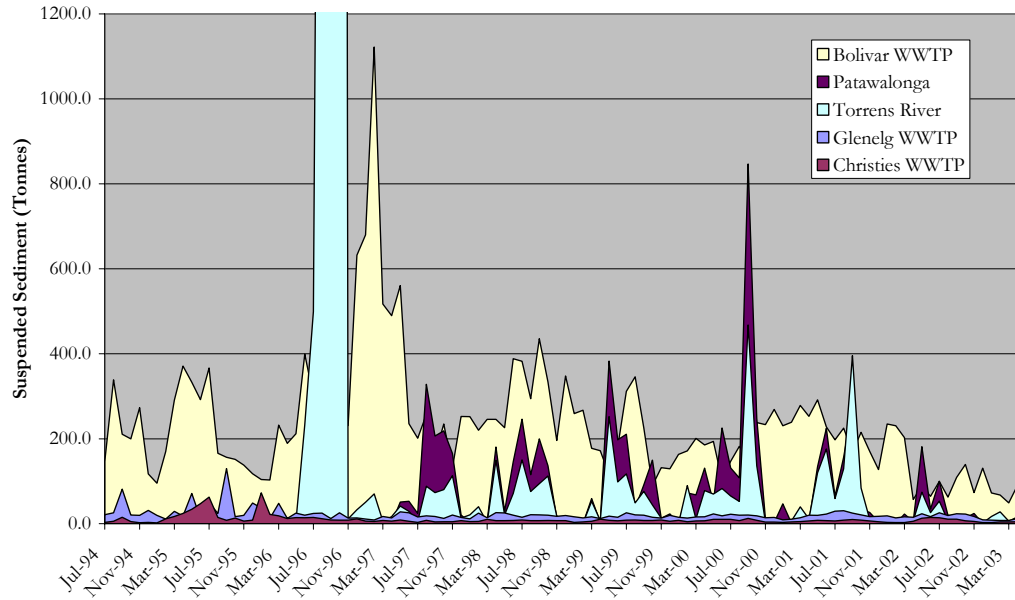


Figure 5.11. Monthly suspended load discharged from the WWTPs, the Torrens River and Patawalonga system.

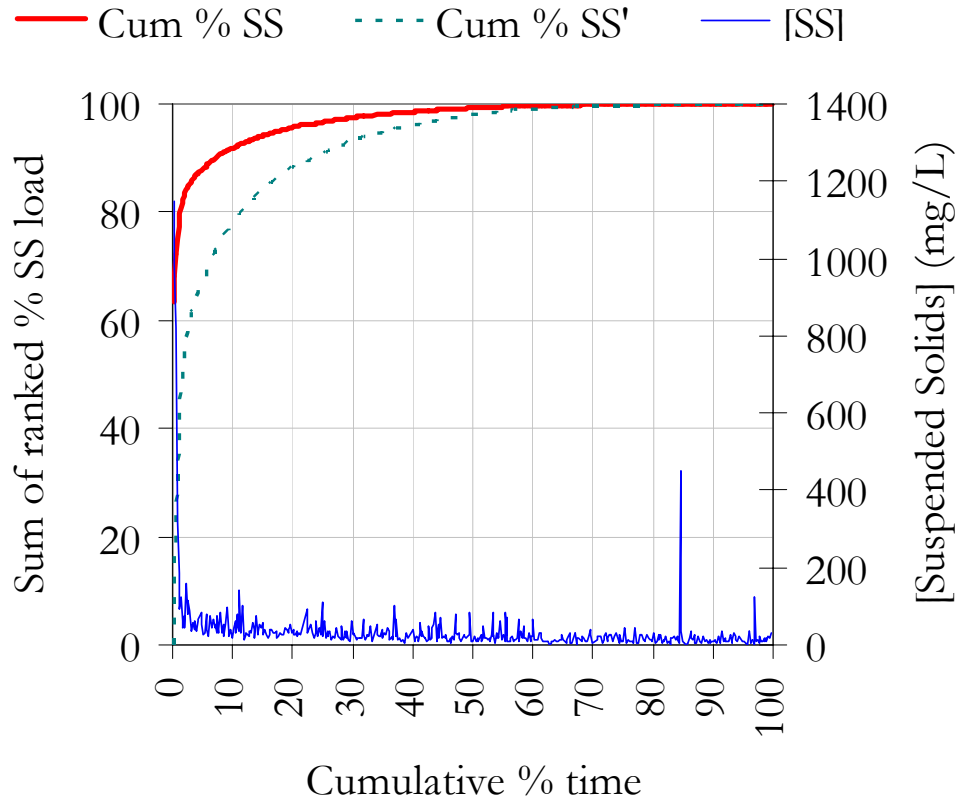


Figure 5.12. Load duration curve for suspended load from the Torrens River between April 1996 and January 2005.

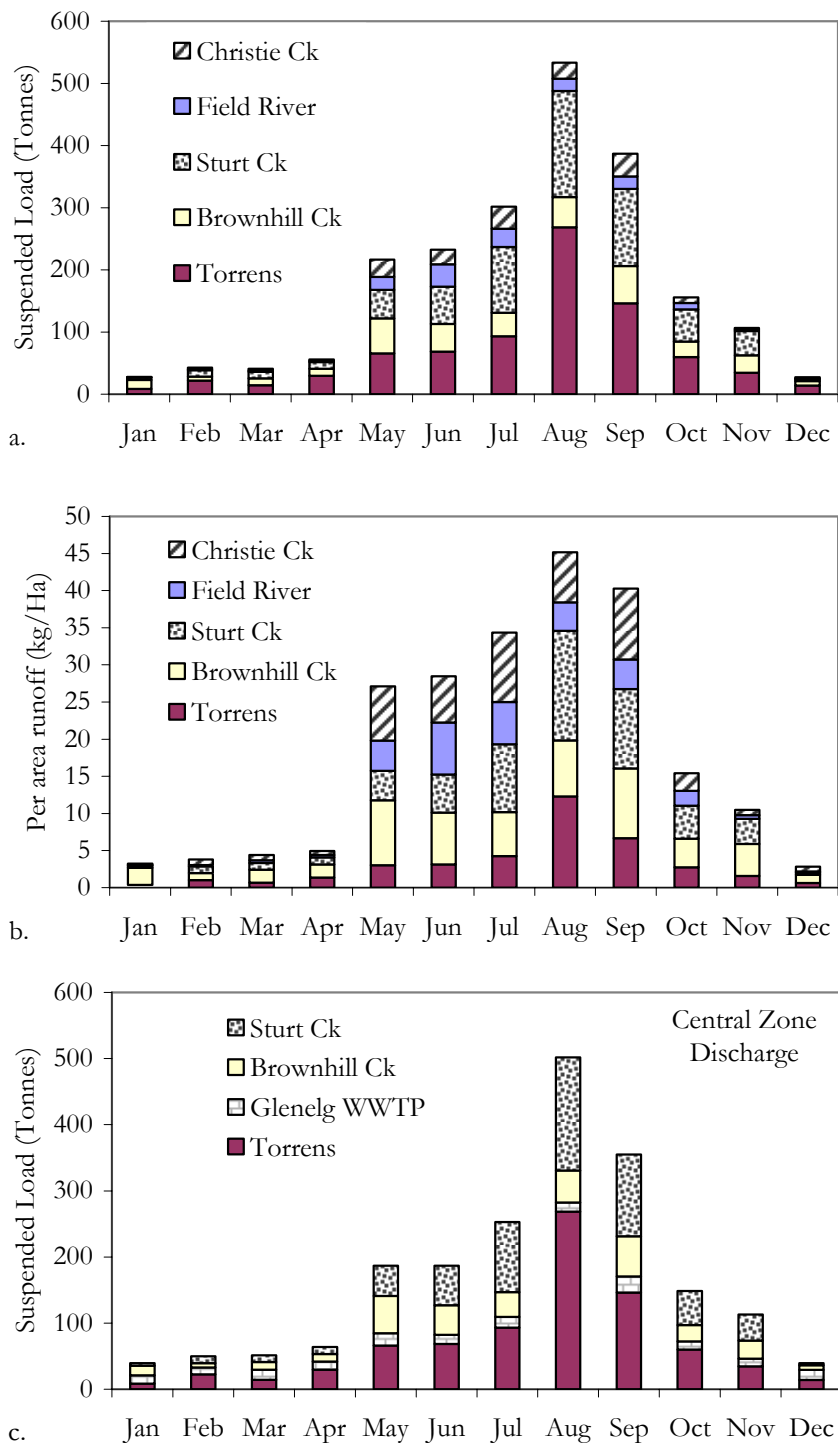


Figure 5.13. Monthly suspended load from a. stormwater sources with flow proportional monitoring, b. sediment yield for the stormwaters in a., and c. Central Zone stormwaters and WWTP sources.

As shown in Section 1.1, the stormwaters dominate the delivery of turbidity causing suspended matter to the coastal zone. In Figure 5.13, only the suspended load from Glenelg WWTP is shown in relation to the stormwater inputs to the Central Zone. The data presented are the monthly mean loads for between 8 and 9 years of observation. In a similar way to phosphorus (the delivery which is linked to particulate matter), the stormwater suspended load from the River Torrens, Brownhill Creek and Sturt Creek doubles from April to May and doubles again from July to August. These are dramatic increases in load far in excess of the increase in discharge volume (i.e. the volume of water discharged) Figure 5.14 and again reflect the nature of the suspended sediment concentration (C) response to flow (Q) which follows a power law relationship; $C=aQ^b$. In the flow-proportional monitoring data used in this study this relationship is lost as a consequence of the long sampling period of one week, the dynamic variations in river response occur over periods of hours. Examination of the statistics for mean suspended solids concentrations shows that in August, September and November the suspended solids concentration is around 40% greater than other times of the year, which clearly accounts for the elevation in suspended load.

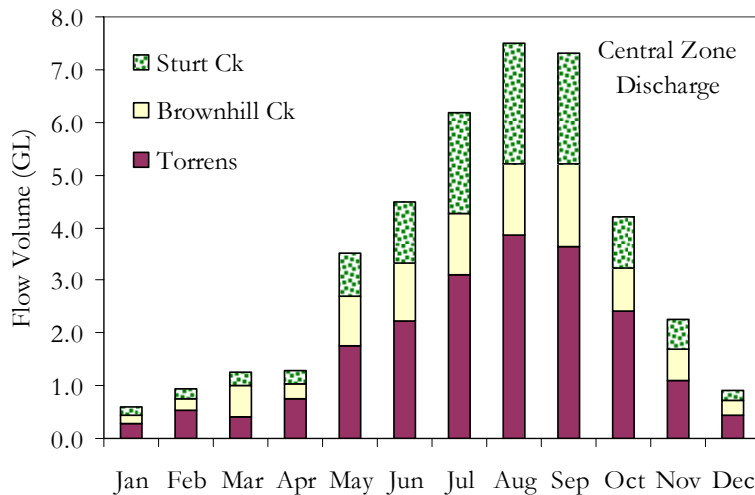


Figure 5.14. Seasonal variation in mean monthly flow from the major Central Zone stormwaters.

The sediment yield from the flow proportionally monitored catchments varies from site to site. Christie Creek has a higher yield than Field River. There is much evidence of erosion along the Christie Creek channel, and yet the sediment yield is consistent with the other creeks excepting August. In August, both Sturt Creek and the River Torrens yield their largest sediment loads, this may be due to greater erosion in the upper reaches of the connected stormwater catchment, where soil wetting and peak winter rains generate maximum erosion. The August suspended sediment load of approximately 500 tonnes in 7 GL of water is the equivalent of 230 ML/day with a suspended solids concentration of 70 mg/L (a turbidity of around 70 NTU). With a mid-winter clear-sky irradiance (I_0) of 650 W/m^2 (calculated by Kirks method, Kirk (1983)), which gives a mean daily irradiance of 178 W/m^2 , which is approximately $780 \mu\text{molquanta/m}^2/\text{sec}$ in the visible range. Using a light attenuation coefficient of $\eta=0.22[\text{SS}]^{0.78}$ (Pompey et al., 1992), the average sea-bottom light intensities given by $I_z= I_0 e^{-\eta z}$ that would be expected are presented below (Figure 5.15). Figure 5.15 is presented as an illustration of the dramatic affect turbidity has on sea-bed light intensity and it indicates a mean or average condition rather than the diurnal and longer term fluctuations that occur as storm events come and go. Direct observations of the variation in seabed light intensity have been made off the Adelaide coast by other study groups within the ACWS.

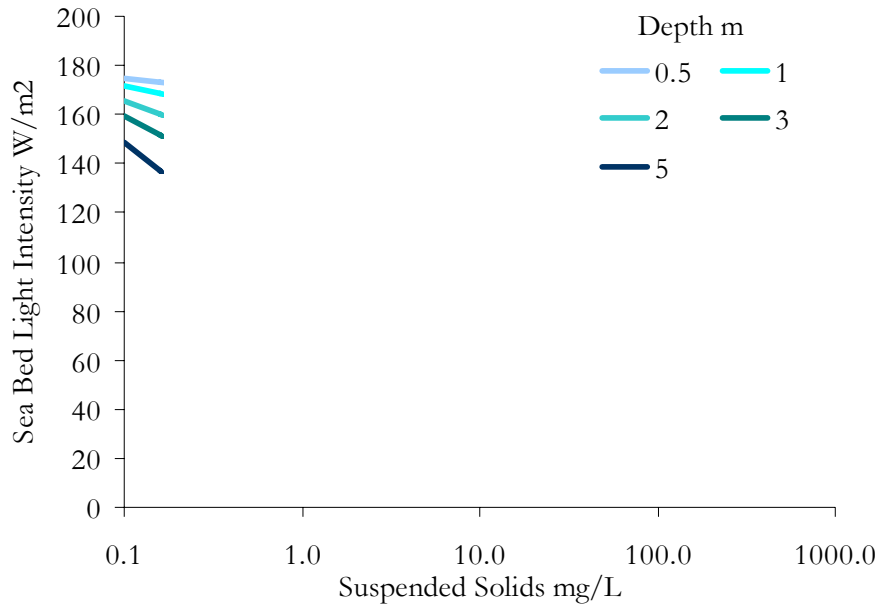


Figure 5.15. Seabed light intensity against suspended solids concentration for a variety of water depths.

5.2.4 Copper, lead and zinc

As stated earlier in this report, copper lead and zinc are the most abundant and commonly detected heavy-metals in storm- and wastewater discharged to the coastal zone. A brief summary of the main sources and seasonal variations in these metals is presented below.

The WWTPs contribute a major component of the copper load to the Adelaide coast (Figure 5.16a). The winter stormwater load is a significant proportion of the winter copper load. Of the stormwater copper yields from the flow-proportionally monitored creeks, Field River and Christie Creek contribute the least and Brownhill Creek the greatest copper yield (Figure 5.16b). The yield from the Torrens River and Sturt Creek are almost identical. Copper yield from the Torrens River, and to a lesser extent Sturt Creek, rise throughout the winter period and drops-off suddenly in October. The cause of this increase has not been investigated.

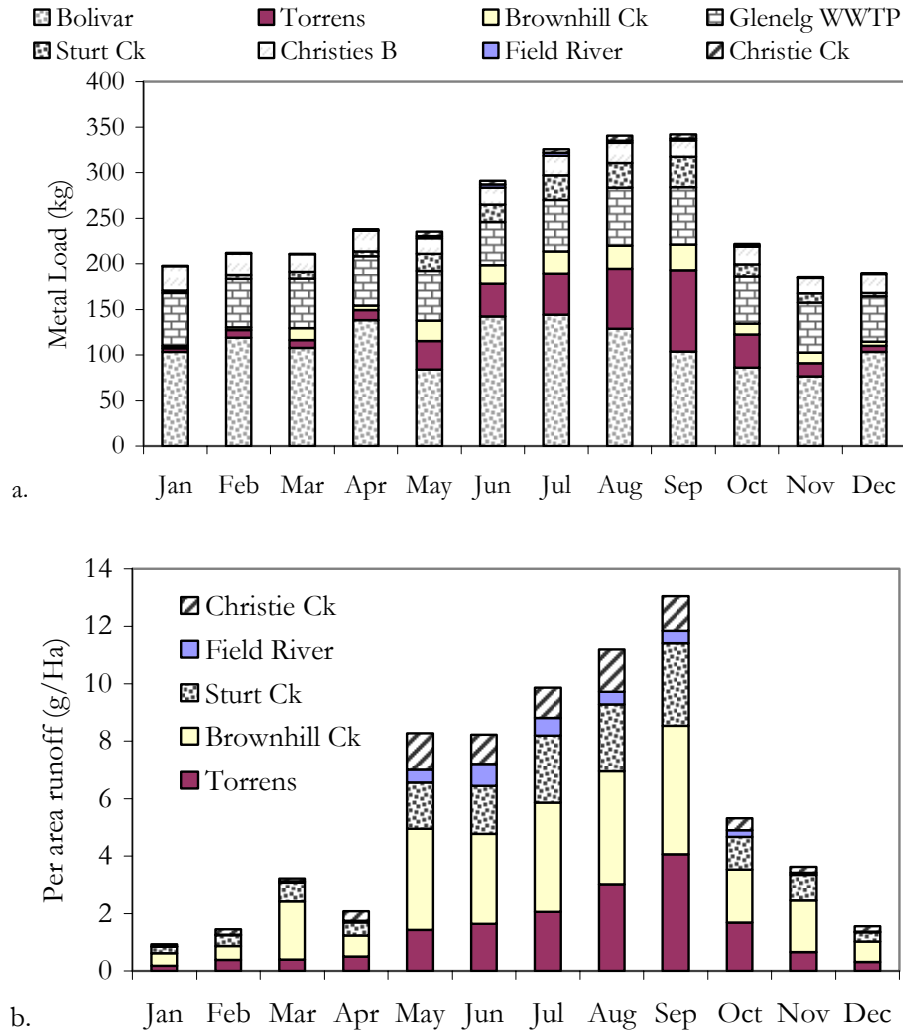


Figure 5.16. Monthly mean WWTP and stormwater a. copper loads, and b. yields from stormwater (note: units of yield = g/Ha).

Stormwaters dominate lead loads (Figure 5.17a). May to the end of September are the peak months for lead discharges. The lead yields from the Torrens River, Field River and Christie Creek are similar. Both Brownhill Creek and Sturt Creek have higher lead yields than the other creeks during the winter months and Brownhill Creek yields greater lead loads throughout the year relative to the other creeks.

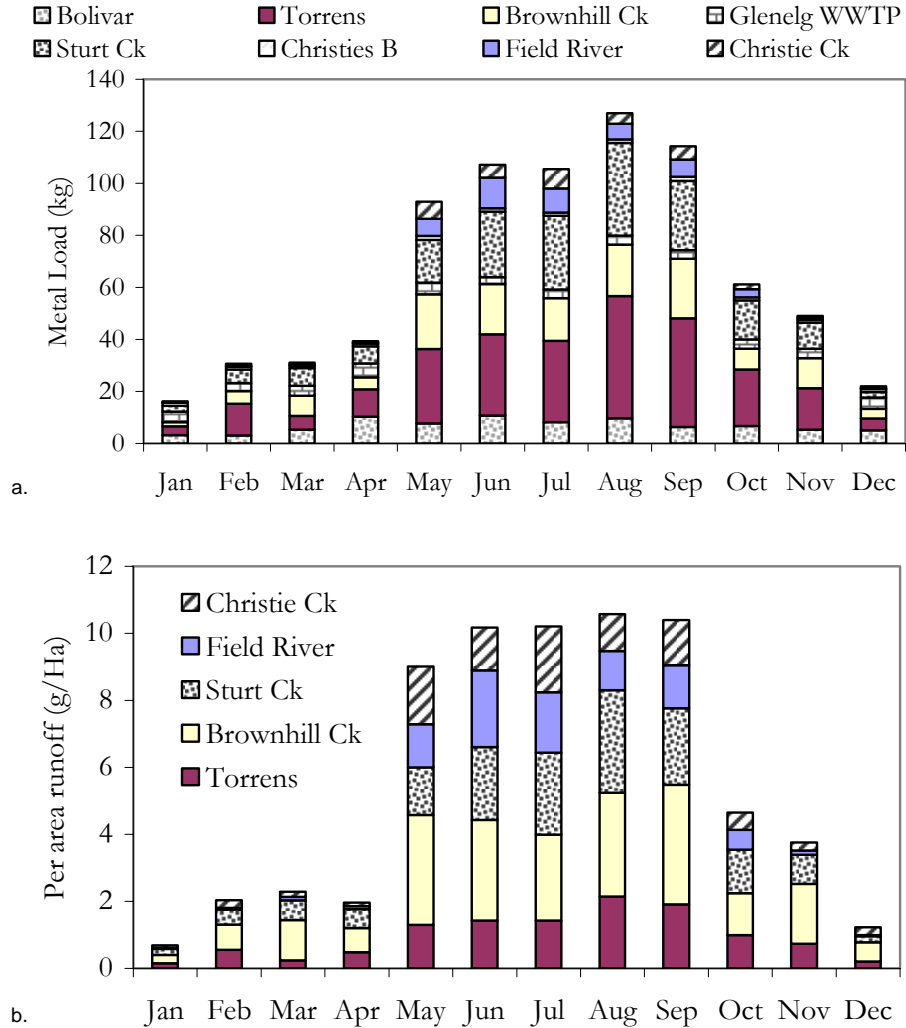


Figure 5.17. Monthly mean WWTP and stormwater a. lead loads, and b. yields from stormwater.

Lead and zinc in stormwater were shown to be correlated in their concentrations (Appendix III). WWTP discharges are not a major source for lead, yet they contribute around 27% of zinc from point source discharges (Section 1.1). The WWTP load of zinc remains consistent throughout the year (Figure 5.18a). The monthly stormwater zinc yield rises in proportion to discharge volume in a similar manner to copper (Figure 5.18b). The yield of zinc is greatest from Brownhill Creek (see Table 5.2 for more detail).

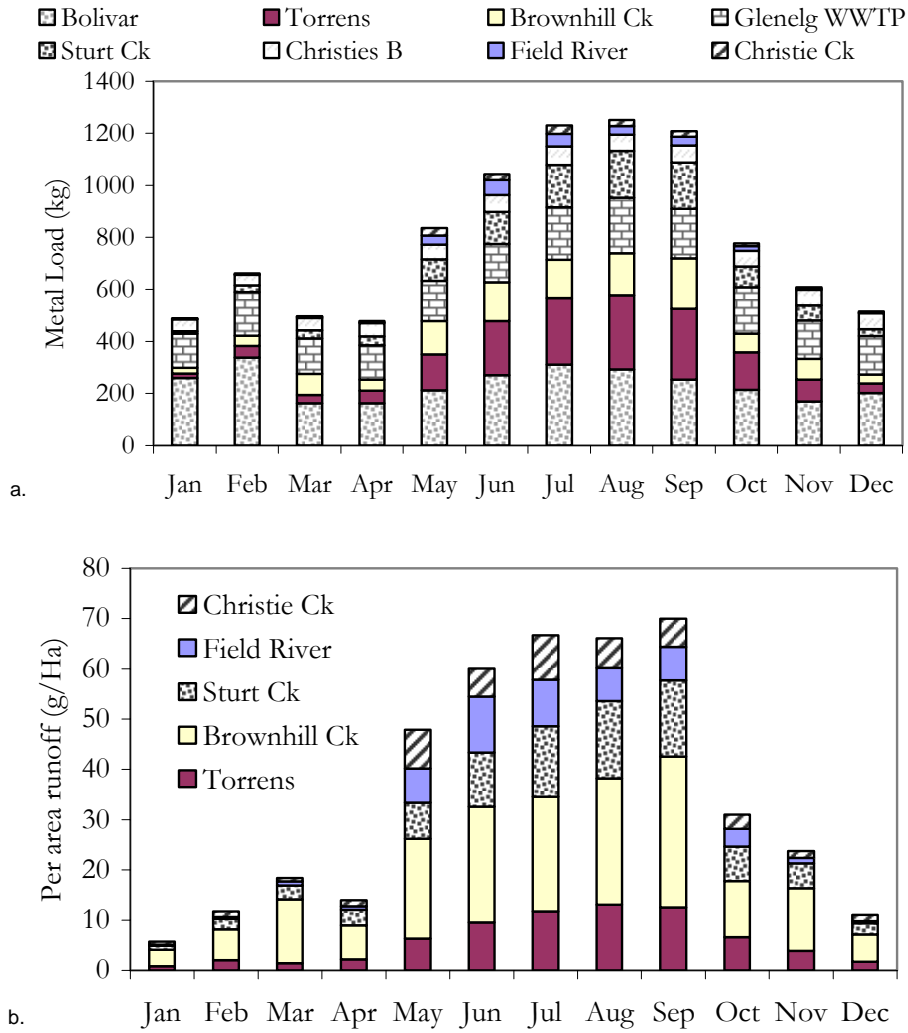


Figure 5.18. Monthly mean WWTP and stormwater a. zinc loads, and b. yields from stormwater.

5.3 Inter-annual variation in stormwater loads and yields

Tables 5.1 and 5.2 are provided to give an indication of the year to year variation in contaminant loads and yields from stormwaters. Only those sites with flow proportional monitoring are presented (with the exception of Field River), to ensure maximum consistency and accuracy in data and hence in the comparisons made. The tables provide flow yield in mm (ML/km²), this gives context to the flow values in relation typical annual rainfall of 400 to 700 mm in the Adelaide Plain and around the Hills face.

Table 5.1. Annual loads and yields for suspended sediment and nutrients from flow proportionally monitored stormwater sources

	Flow GL	Susp Sed Tonnes	TN kg	TKN kg	NOxN kg	TP kg	P:SS g/kg	TN:TP	Yield mm	TN kg/Ha/yr	TKN kg/Ha/yr	NOxN kg/Ha/yr	TP kg/Ha/yr	
Torrens River: Holbrooks Road	1996	50.56	25356	158237	116307	41931	18424	0.7	8.6	231	7.24	5.32	1.92	0.84
	1997	12.68	453	22084	13640	8443	1234	2.7	17.9	58	1.01	0.62	0.39	0.06
	1998	16.61	723	26229	14460	11769	2102	2.9	12.5	76	1.20	0.66	0.54	0.10
	1999	18.20	673	25843	14030	11813	1753	2.6	14.7	83	1.18	0.64	0.54	0.08
	2000	25.08	1025	33787	16902	16885	1787	1.7	18.9	115	1.55	0.77	0.77	0.08
	2001	36.62	986	51626	25839	25787	2505	2.5	20.6	168	2.36	1.18	1.18	0.11
	2002	10.12	202	14196	7534	6661	588	2.9	24.1	46	0.65	0.34	0.30	0.03
	2003	23.06	577	27517	15021	12496	1569	2.7	17.5	106	1.26	0.69	0.57	0.07
	2004	25.28	1998	41684	24214	17470	6556	3.3	6.4	116	1.91	1.11	0.80	0.30
Brownhill Creek: Adelaide Airport	1997	5.74	355	6362	5330	1032	933	2.6	6.8	89	0.99	0.83	0.16	0.15
	1998	7.09	444	8029	6565	1465	1184	2.7	6.8	110	1.25	1.02	0.23	0.18
	1999	9.21	544	9720	8100	1620	1465	2.7	6.6	143	1.51	1.26	0.25	0.23
	2000	11.76	405	11025	8047	2978	1309	3.2	8.4	183	1.72	1.25	0.46	0.20
	2001	14.74	412	16192	12782	3410	2071	5.0	7.8	230	2.52	1.99	0.53	0.32
	2002	4.67	140	4911	4071	841	764	5.5	6.4	73	0.77	0.63	0.13	0.12
	2003	8.60	162	8023	6261	1762	1082	6.7	7.4	134	1.25	0.98	0.27	0.17
	2004	8.83	287	7994	6233	1761	1136	4.0	7.0	138	1.25	0.97	0.27	0.18
Sturt Creek: Anzac Hwy	1996	28.63	2904	41023	34453	6570	6828	2.4	6.0	247	3.54	2.97	0.57	0.59
	1997	10.11	662	15410	11303	4107	3035	4.6	5.1	87	1.33	0.97	0.35	0.26
	1998	9.80	481	13579	10141	3438	2943	6.1	4.6	84	1.17	0.87	0.30	0.25
	1999	10.63	532	14548	10674	3875	2819	5.3	5.2	92	1.25	0.92	0.33	0.24
	2000	15.94	1095	21562	15541	6021	3992	3.6	5.4	137	1.86	1.34	0.52	0.34
	2001	8.98	487	12536	8036	4501	2162	4.4	5.8	77	1.08	0.69	0.39	0.19
	2002	5.70	268	7837	5890	1948	1808	6.8	4.3	49	0.68	0.51	0.17	0.16
	2003	14.20	517	19853	14540	5313	3830	7.4	5.2	122	1.71	1.25	0.46	0.33
	2004	13.60	977	16399	12465	3935	3546	3.6	4.6	117	1.41	1.07	0.34	0.31
Field River: Hallett Cove	1999	3.17	109	2802	1352	1449	284	2.6	9.9	60	0.82	0.40	0.42	0.08
	2000	5.75	213	4464	2336	2128	559	2.6	8.0	108	1.31	0.68	0.62	0.16
	2001	5.00	182	4021	2059	1963	479	2.6	8.4	94	1.18	0.60	0.57	0.14
	2002	2.69	88	2491	1172	1319	229	2.6	10.9	50	0.73	0.34	0.39	0.07
	2003	3.83	127	3401	1651	1750	329	2.6	10.3	72	0.99	0.48	0.51	0.10
	2004	4.36	154	3585	1815	1769	404	2.6	8.9	82	1.05	0.53	0.52	0.12
Christie Creek: Galloway Rd.	2001	2.95	273	4852	3264	1588	447	1.6	10.9	78	1.28	0.86	0.42	0.12
	2002	2.39	165	3676	2480	1196	385	2.3	9.6	63	0.97	0.66	0.32	0.10
	2003	2.17	143	3206	2359	848	365	2.5	8.8	57	0.85	0.62	0.22	0.10
	2004	1.94	103	2826	1876	950	263	2.6	10.8	51	0.75	0.50	0.25	0.07

Table 5.1 demonstrates that while stormwater contaminant discharges may vary by a factor of around 2 from year to year, exceptional years such as 1996 (and probably 2005) can deliver dramatically elevated loads in a very short space of time (as shown in Section 5.2 above).

Phosphorus and lead tend to be associated with particulates, especially clay particles in stormwater, so the phosphorus/suspended solids (TP:SS) and lead/suspended solids (Pb:SS) ratios (in g/kg) have been provided. For phosphorus, this value was consistently around 2.5 to 2.7 g/kg for the Torrens, Field River and Christie Creek. For Brownhill Creek and Sturt Creek the TP:SS ratio is more variable and higher. The TN:TP ratio is also informative in this respect since it is known that Sturt Creek transports an elevated phosphorus load and consequently has

a lower TN:TP ratio. The Torrens has higher TN:TP ratio and a low P yield compared to the other sites. This may be due to the biological utilisation of P in the Torrens Lake. The low TN:TP ratio in Brownhill Creek and Sturt Creek may also be due to the low potential for biological uptake caused by the high proportion of canalised channel.

Table 5.2 presents data for copper, lead and zinc. Lead and zinc decline in load and yield with time (excepting Field River where a time invariant load/discharge relationship was used). The yield estimates for metals can be used in conjunction with dry deposition data to indicate how much of the point source load is derived from atmospheric deposition.

Table 5.2. Annual loads and yields for suspended sediment and heavy metals from flow proportionally monitored stormwater sources.

		Flow	Susp Sed	Cu	Pb	Zn	Pb:SS	Pb:Flow	Yield	Susp Sed	Cu	Pb	Zn
		GL	Tonnes	kg	kg	kg	g/kg	ug/L	mm	kg/Ha/yr	g/Ha/yr	g/Ha/yr	g/Ha/yr
Torrens River: Holbrooks Road	1996	50.56	25356	1985	6084	11903	0.240	120.3	231	1160.4	90.8	278.4	544.8
	1997	12.68	453	153	288	1505	0.636	22.7	58	20.7	7.0	13.2	68.9
	1998	16.61	723	186	337	2219	0.466	20.3	76	33.1	8.5	15.4	101.5
	1999	18.20	673	546	314	1535	0.467	17.3	83	30.8	25.0	14.4	70.3
	2000	25.08	1025	664	277	1823	0.270	11.0	115	46.9	30.4	12.7	83.4
	2001	36.62	986	867	323	2228	0.328	8.8	168	45.1	39.7	14.8	102.0
	2002	10.12	202	89	67	551	0.330	6.6	46	9.3	4.1	3.1	25.2
	2003	23.06	577	141	158	1384	0.274	6.9	106	26.4	6.4	7.2	63.3
	2004	25.28	1998	259	270	1505	0.135	10.7	116	91.4	11.8	12.3	68.9
Brownhill Creek: Adelaide Airport	1997	5.74	355	116	206	1112	0.580	35.9	89	55.2	18.0	32.0	173.2
	1998	7.09	444	92	223	1351	0.501	31.4	110	69.2	14.4	34.7	210.4
	1999	9.21	544	310	182	1404	0.335	19.8	143	84.7	48.3	28.4	218.7
	2000	11.76	405	325	138	1261	0.341	11.8	183	63.1	50.6	21.5	196.4
	2001	14.74	412	293	158	1512	0.384	10.7	230	64.2	45.6	24.7	235.4
	2002	4.67	140	50	67	558	0.479	14.4	73	21.8	7.9	10.4	87.0
	2003	8.60	162	74	58	853	0.357	6.7	134	25.2	11.6	9.0	132.8
	2004	8.83	287	100	81	915	0.283	9.2	138	44.7	15.6	12.7	142.5
Sturt Creek: Anzac Hwy	1996	28.63	2904	644	2854	5127	0.983	99.7	247	250.3	55.5	246.0	442.0
	1997	10.11	662	162	264	1340	0.399	26.2	87	57.1	13.9	22.8	115.5
	1998	9.80	481	113	221	1481	0.459	22.5	84	41.5	9.8	19.0	127.7
	1999	10.63	532	313	169	1087	0.318	15.9	92	45.9	27.0	14.6	93.7
	2000	15.94	1095	422	208	1305	0.190	13.1	137	94.4	36.3	17.9	112.5
	2001	8.98	487	170	99	719	0.203	11.0	77	42.0	14.6	8.5	61.9
	2002	5.70	268	65	69	533	0.257	12.1	49	23.1	5.6	5.9	45.9
	2003	14.20	517	103	68	473	0.131	4.8	122	44.5	8.9	5.8	40.8
2004	13.60	977	126	99	670	0.101	7.2	117	84.2	10.8	8.5	57.8	
Field River: Hallett Cove											5326		
	1999	3.17	109	12.3	33.6	185	0.308	10.6	60	20.4	2.3	6.3	34.8
	2000	5.75	213	22.9	68.8	345	0.323	12.0	108	40.0	4.3	12.9	64.8
	2001	5.00	182	19.8	58.6	299	0.321	11.7	94	34.2	3.7	11.0	56.0
	2002	2.69	88.1	10.3	26.1	155	0.296	9.7	50	16.5	1.9	4.9	29.1
	2003	3.83	127	14.7	37.7	222	0.297	9.8	72	23.8	2.8	7.1	41.7
2004	4.36	154	17.1	48.3	258	0.313	11.1	82	29.0	3.2	9.1	48.4	
Christie Creek: Galloway Rd.	2001	2.95	273	60.6	46.2	217	0.170	15.7	78	72.1	16.0	12.2	57.5
	2002	2.39	165	19.2	37.9	168	0.229	15.9	63	43.8	5.1	10.0	44.3
	2003	2.17	143	17.1	32.4	146	0.226	14.9	57	37.9	4.5	8.6	38.6
	2004	1.94	103	17.0	19.5	108	0.190	10.0	51	27.1	4.5	5.2	28.6

As previously mentioned lead and zinc have declined consistently in stormwater since the mid-nineteen nineties, copper loads show no trend and remain consistent with variations in rainfall/flow.

Longer term variations in loads estimated using contemporary and historical data are presented below (Section 5.4).

5.4 Contemporary and Historical Pollutant Loads

Contemporary and historical loads of contaminants entering the Adelaide Coastal Zone from WWTP and stormwater discharges are presented below. The contemporary loads are estimated using regular monitoring data and a combination of measured or modelled river flows. Where flow-proportional sampling was available, loads were calculated directly from the observed flows and concentrations. Where only grab sampled concentrations and modelled flows were available, as was the case for the historical loads, either load/discharge relationships were derived or mean concentration/mean flow loads were estimated (Chapter 4 gives more details of load estimation methods).

A reliable comparison of these estimates of load is not possible. The results should be seen as indicative only. The sources of uncertainty in the estimates are many, and in some cases the number of samples on which an estimate is made is small, while in others the nature of the sampling introduces bias towards low or high flow events. Comparison of pollutant yields may be misleading because they represent runoff from the whole catchment, and not the runoff contributing area. The load per km² per mm runoff (kg/km²/mm) (pollutant yield/runoff yield) gives a better indicator of the load or quality of runoff from the area which is active in producing runoff. This approximates the mean concentration and can also be calculated by load/flow. Where flow-proportional sampling is available and loads have been estimated directly from the flows and concentrations, or a well populated load/discharge relationship has been used, the mean concentration and the load/flow will give approximately the same result, e.g. River Torrens, Ocean Catchment, Field River and Christie Creek (Table 5.3).

Table 5.3. Checking and comparison of sediment yields and runoff from the major ACWS stormwater sources (current).

Current	Flow GL	Area km ²	Runoff Yield (mm)	Susp Sediment Original Estimate (T/a) Note: suspect figures underlined	SS Load assuming an appropriate yield (ignores runoff yield)	Mean x Flow (T/a)	SS Yield (original est.) (T/km ²)	SS yield/runoff yield (kg/km ² /mm = kg/GL = mg/L)	Observed Mean Concentration (mg/L)	Derived Mean Concentration (mg/L)
Gawler River	15.1	883.3	17.0	<u>2327</u>	2421		2.63	154.5		
Smith Creek	1.1	205.6	5.5	<u>20</u>	563	<u>176</u>	0.10	17.9		<u>155.0</u>
Barker Inlet	26.0	439.4	59.2	<u>466</u>	1662	<u>1279</u>	1.06	17.9		<u>49.1</u>
Torrens River	21.4	218.5	98.0	<u>827</u>		766	<u>3.78</u>	<u>38.6</u>	35.8	
Patawalonga	20.5	212.4	96.5	<u>999</u>			<u>4.70</u>	<u>48.7</u>		
Ocean Catchment	1.7	21.0	79.8	<u>97</u>		91	<u>4.59</u>	<u>57.5</u>	54.4	
Field River	4.1	53.3	76.6	<u>146</u>		147	<u>2.74</u>	<u>35.8</u>	36.1	
Christie Creek	2.6	37.8	69.2	<u>170</u>		148	<u>4.50</u>	<u>65.0</u>	<u>56.8</u>	
Onkaparinga	17.6	200.1	88.2	<u>210</u>	548	<u>752</u>	1.05	11.9	18.1	42.6
Southern Creeks	4.4	221.4	19.8	78	607	<u>128</u>	0.35	<u>17.7</u>	17.9	29.1

For Smith Creek and most of Barker Inlet, no recent concentration data was available to estimate loads. An initial mean concentration of 17.9 mg/L suspended solids was used based on the overall area weighted mean of the Southern Creeks. This yielded suspiciously low load values and in order to increase these loads the mean of the Torrens to Christie Creek concentrations was used for Barker Inlet (assuming some stabilisation in source areas and similarity), and for Smith Creek the mean for Gawler River was used. The estimates for the Onkaparinga River and Southern Creeks were also suspect and the mean concentrations were derived predominantly from ambient samples collected during quiescent (or dry weather conditions). In order to reduce this bias, a new mean, deliberately biased towards stormflow

samples, was used. This was done using all IS1 stormflow sample results for each site and then n times the mean of the ambient sampling data, where n is the number of IS1 stormwater samples. For example, if 5 stormwater samples were collected the new mean would be:

$$NewMean = \frac{\sum_{i=1}^n IS1samples + n.ambient}{2n}$$

The same approach was used to investigate the suspended solids loads for 1975-85 (Table 5.4).

Table 5.4. Checking and comparison of sediment yields and runoff from the major ACWS stormwater sources (1975-85).

1975 - 85	Flow GL	Area km ²	Runoff Yield (mm)	Susp Sediment Original Estimate (T/a) Note: suspect figures underlined	SS Load assuming an appropriate yield (ignores runoff yield)	Mean x Flow (T/a)	SS Yield (original est.)(T/km ²)	SS yield/runoff yield (kg/km ² /mm = kg/GL = mg/L)	Observed Mean Concentration (mg/L)	Derived Mean Concentration (mg/L)
Gawler River	16.9	883.3	19.1	<u>2689</u>	<u>6883</u>		3.04	159.5	159.5	
Smith Creek	1.7	205.6	8.1	<u>265</u>	<u>1602</u>	204	1.29	159.5		123
Barker Inlet	21.0	439.4	47.8	<u>16384</u>	<u>3424</u>	2584	37.29	779.9	779.9	123
Torrens River	18.6	218.5	85.2	801		<u>980</u>	3.67	43.0	52.7	
Patawalonga	13.4	212.4	63.2	1655		<u>2079</u>	7.79	123.4	155.0	
Ocean Catchment	1.4	21.0	67.4	<u>657</u>	<u>164</u>	235	31.28	463.7	459.0	166
Field River	2.3	53.3	42.7	<u>3</u>	195	82	0.06	1.5	1.5	36.1
Christie Creek	2.0	37.8	52.7	<u>3</u>	295	<u>113</u>	0.08	1.5	1.5	56.8
Onkaparinga	13.7	200.1	68.5	507	<u>733</u>	<u>590</u>	2.54	37.0	37.0	43
Southern Creeks	5.8	227.3	25.4	<u>4</u>	<u>833</u>	103	0.02	0.8		17.9

The original Barker Inlet suspended solids load was estimated from the mean of sample concentrations from the late 1980s, this was the only data available. The resulting suspended solids load was not consistent with that for the other large discharges, so an adjusted load was estimated on the higher of the two values of yield/mm runoff (concentration) estimated for the Torrens and Patawalonga. This was 123 mg/L, this gave a more realistic load, the validity of which remains uncertain. Suspended sediment loads for Christie Creek and Field River were initially made on the basis of a very few samples collected during Gulf St. Vincent Water Pollution Studies. These gave very low estimates of SS yield and new loads were estimated using the sediment yield for the Torrens River and for comparison the contemporary mean SS concentrations. Since flow modelling suggests that flows in the Field River and Christie Creek were lower due to lesser impervious areas it was argued that the sediment loads might also be less or of a similar magnitude. At the same time around 20,000 to 30,000 new dwellings were constructed in the Christie Creek catchment between 1970 and 1989, thus major land disturbance and soil erosion would have been likely during this period. This evidence of major development in Christie Creek prompted the choice of the higher sediment yield (from the Patawalonga) to estimate the suspended load.

Table 5.5. Contemporary loads of contaminants from WWTP and stormwater sources, as presented in Section 1.1.

Loads (T/a)	Flow GL	% tot	Area km ²	Yield (mm)	Susp Sed	Total N	TKN	NO _x -N	Total P	NH ₃ -N	Total Cu	Total Pb	Total Zn
Current													
Gawler River	15.1	8.5	883.3	17.0	2327	31.3	27.9	3.5	7.1	2.4	0.12	0.19	1.35
Bolivar WWTP	35.3	20.0		1272	487.3		381.9	105.4	122.0	171.2	1.91	0.15	1.75
Smith Creek	1.1	0.6	205.6	5.5	176	1.3	1.0	0.3	0.1	0.1	0.01	0.01	0.10
Barker Inlet	26.0	14.7	439.4	59.2	1279	28.7	22.0	6.3	3.2	1.4	0.21	0.32	2.33
Penrice	0.0				1000.0	1000.0	1000.0						
Islington SF													
Port Adelaide WWTP													
Torrens River	21.4	12.1	218.5	98.0	827	30.0	16.2	13.8	2.2		0.36	0.25	1.57
Gleneig WWTP	17.3	9.8		223	223	471.4	189.2	282.2	143.7	144.5	0.80	0.05	1.46
Patawalonga	20.5	11.6	212.4	96.5	999	25.3	19.3	6.0	4.3		0.35	0.33	2.19
Ocean Catchment	1.7	0.9	21.0	79.8	97	1.8	1.6	0.24	0.29	0.12	0.03	0.10	0.13
Field River	4.1	2.3	53.3	76.6	146	3.5	1.7	1.7	0.3		0.04	0.04	0.14
Christies WWTP	9.4	5.3		84	84	245.6	195.7	49.9	69.2	188.3	0.25	0.01	0.47
Christie Creek	2.6	1.5	37.8	69.2	170	3.6	2.5	1.1	0.4		0.03	0.001	0.16
Onkaparinga	17.6	10.0	200.1	88.2	758	22.1	19.4	2.6	1.8	1.2	0.14	0.22	1.58
Southern Creeks	4.7	2.7	221.4	21.4	71	5.9	5.4	0.5	0.6	0.1	0.01	0.01	0.64
TOTAL	176.9	100.0	2492.8	70.9	8428	2357.7	1883.7	473.6	355.2	1509.3	4.27	1.69	13.87
WWTP	62.0	35.0		1579	1204.2		766.8	437.5	335.0	504.0	2.96	0.21	3.69
Stormwater	114.9	65.0	2492.8	46.1	6849	153.5	117.0	36.1	20.3	5.3	1.31	1.48	10.19
Northern	51.5	29.1	1088.9	47.3	3775	519.8	410.7	109.1	129.2	173.7	2.0	0.3	3.2
Barker Inlet	26.0	14.7	439.4	59.2	1279	1028.7	1022.0	6.3	3.2	1001.4	0.2	0.3	2.3
Central	60.8	34.4	451.9	134.6	2145	528.5	226.2	302.3	150.5	144.6	1.5	0.7	5.3
Southern	38.5	21.7	512.6	75.0	1229	280.7	224.7	56.0	72.3	189.6	0.5	0.3	3.0

Table 5.6. Historical loads of contaminants from WWTP and stormwater sources for the period 1975-85.

Loads (T/a)	Flow CL	% tot	Area km ²		Yield (mm)		Susp Sed		TKN	NO _x -N		NH ₃ -N	Total Pb		Total Zn
			Area km ²	Yield (mm)	Susp Sed	Total N	Total N	Total Cu		Total Pb					
Gawler River	16.9	10.8	883.3	19.1	2689	35.6	31.2	4.4	6.6	2.7	0.46	0.17	0.67		
Bohlar WWTP	26.1	16.8		2137	824.2	783.9	40.3	537.1	213.7	40.3	2.40	1.26	9.40		
Smith Creek	1.7	1.1	205.6	8.1	204	3.5	3.1	0.4	0.6	0.3	0.04	0.02	0.07		
Banker Inlet	21.0	13.5	439.4	47.8	2584	53.6	43.8	9.8	32.4	1.14	5.72	8.19			
<i>Penrice</i>	0.0					1000.0				1000.0					
Port Adelaide WWTP	12.0	7.7			290	401.6	264.0	140.3	96.1	1.08	0.12	3.36			
Port Adelaide Sludge	0.1				1721	81.8	81.6	0.2	19.4	33.0	2.23	0.45	3.61		
Torrens River	18.6	12.0	218.5	85.2	801	51.4	16.8	34.6	2.0	2	0.99	1.60	3.07		
Gleneig WWTP	16.7	10.7			125	595.3	261.8	333.5	133.4	2.35	0.73	10.89			
Gleneig Sludge	0.2	0.1			2230	213.2	213.2	50.9	50.9	104.7	4.97	0.71	7.33		
Patawalonga	13.4	8.6	212.4	63.2	1655	31.9	22.2	9.7	1.8	2	0.71	1.15	2.21		
Ocean Catchment	1.4	0.9	21.0	67.4	235	3.2	1.9	1.29	0.50		0.17	0.38	0.88		
Field River	2.3	1.5	53.3	42.7	82	5.0	3.0	2.0	0.9		0.03	0.40	0.05		
Christies WWTP	3.8	2.4			34	192.9	58.5	134.4	30.0	979.7	0.52	0.09	1.72		
Christie Creek	2.0	1.3	37.8	52.7	295	4.4	2.6	1.7	0.8		0.05	0.11	0.10		
Onkaparinga	13.7	8.8	200.1	68.5	507	43.2	36.7	6.4	1.2		0.25	0.26	0.65		
Southern Creeks	5.8	3.7	227.3	25.4	103	12.7	7.6	5.0	2.2		0.03	0.11	0.19		
TOTAL	155.6	100.0	2498.7	62.3	15692	3553.3	2831.9	1220.8	592.5	2165.1	17.4	13.3	52.4		
WWTP	58.9	37.8			6536.2	2309.0	1663.0	1145.5	543.6	1157.7	13.5	3.3	36.3		
Stormwater	96.7	62.2	2498.7	38.7	9155	244.3	168.9	75.4	48.9	7.4	3.9	9.9	16.1		
Northern	44.6	28.7	1088.9	41.0	5030	863.3	818.2	541.9	220.9	43.3	2.9	1.4	10.1		
Barker Inlet	33.0	21.2	439.4	75.2	2874	1455.2	1307.8	150.1	128.5	1000.0	2.2	5.8	11.6		
Central	50.4	32.4	451.9	111.6	6766.9	976.7	597.5	379.3	208.0	142.1	11.4	5.0	28.0		
Southern	27.5	17.7	518.5	53.1	1021	258.1	108.5	149.6	35.1	0.0	0.9	1.0	2.7		

Tables 5.5 and 5.6 give the full summary of loads for the two periods being compared. Figure 5.19 and 5.20 present these data in graphical form. The figures give a quick visual indication of the changes in volumes and loads.

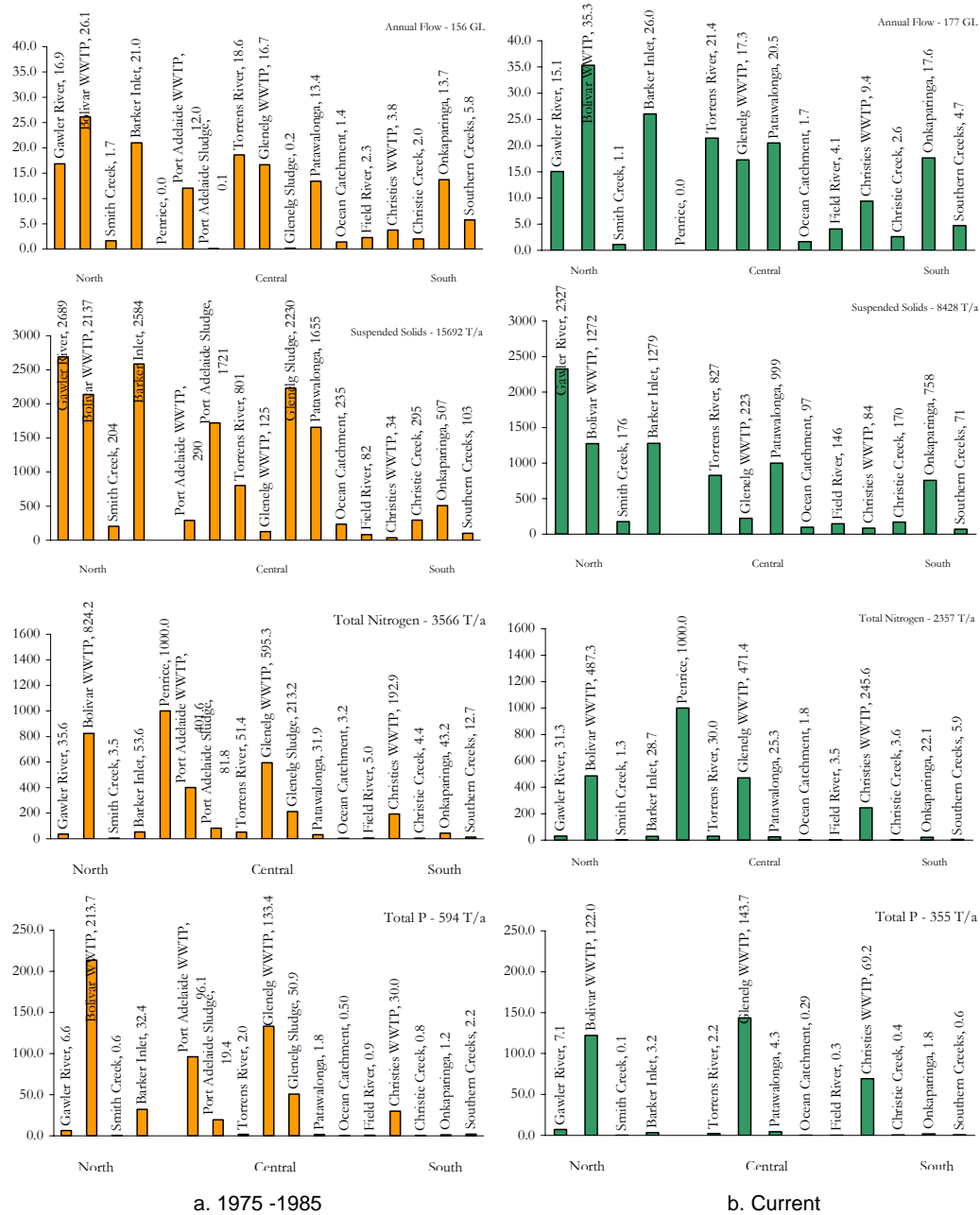


Figure 5.19. Volumes and contaminant loads of WWTP and stormwater discharges to the coastal zone.

Figure 5.19 presents flow volume, suspended load and nutrients for two ten year epochs, 1975 to 95 and 1995 to 2005, respectively. The changes in flow volumes are relatively insignificant between the two periods. The most apparent differences are the reduction in number of sources and reductions in loads (Table 5.5), and from both WWTP sources (as reported in Wilkinson et al., 2004), and stormwaters. The EIP improvements, closure of Port Adelaide WWTP, and closure of the two sludge outfalls have in combination significantly reduced contaminant loads to the Adelaide coastline. The output from Christies Beach WWTP has increased in all

components, as a consequence of the major growth in residential development that has taken place south of O'Halloran Hill since 1970.

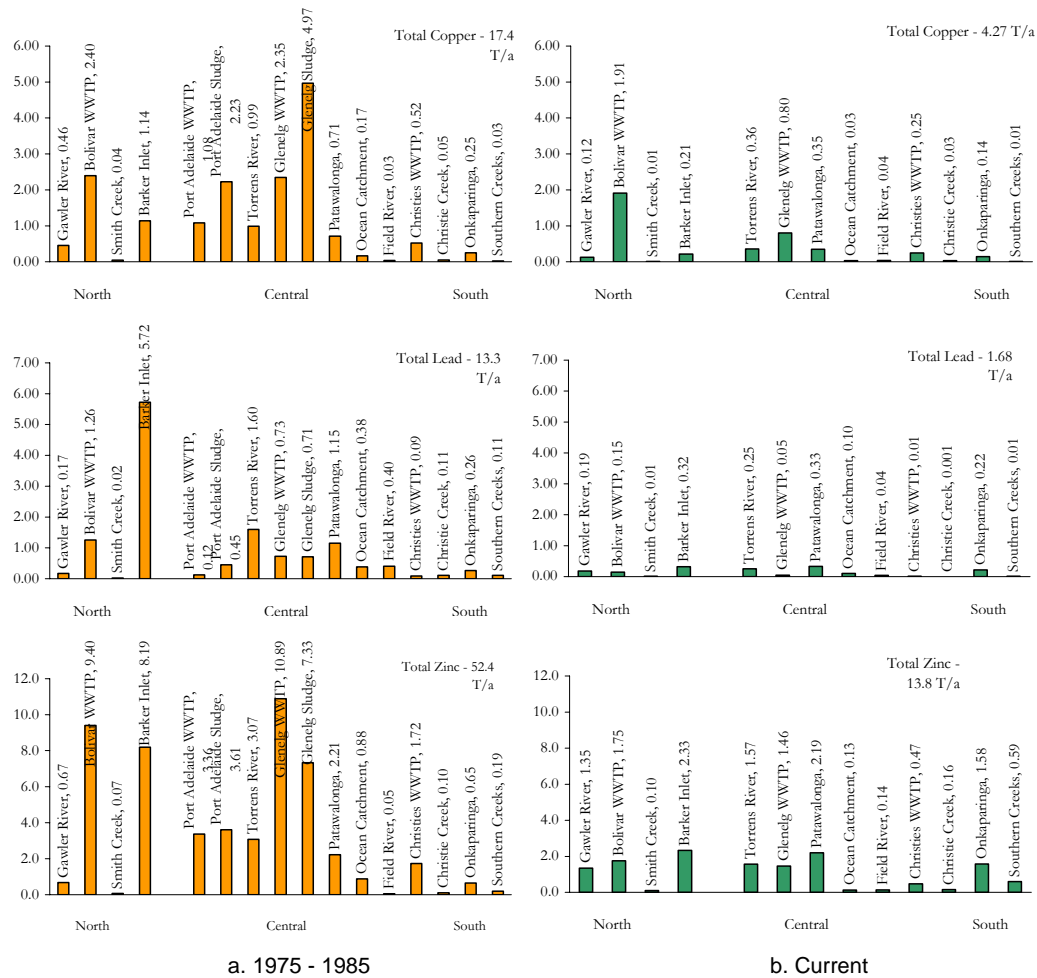


Figure 5.20. Volumes and contaminant loads of WWTP and stormwater discharges to the coastal zone.

The storm drains of the Patawalonga ocean catchment (Holdfast Bay), are on the few locations where directly comparable data have been collected for the two periods (see Table A2.3). Table 5.5 demonstrates a 50 % reduction in stormwater nitrate (NO_x-N), a 30% reduction in TKN. These overall reductions in stormwater nitrogen loadings are consistent with the changes in nitrogen concentrations and speciation in the stormwater of the ocean catchment (Table A2.3). The overall reduction in nitrate nitrogen of 61%, and 40% reductions in total phosphorus and suspended load are significant and may account for the slowing sea-grass decline since the mid-nineteen eighties (see Section 6).

The reduction in metal loads to the Adelaide Coast is particularly striking (Figure 5.20, Table 5.5), reductions of between 70 and almost 90%. These improvements are due to the reductions in WWTP discharge loadings and reductions in stormwater loads. The WWTP reductions have been achieved through the elimination of the sludge outfalls and major reductions in metals concentrations as detailed in Wilkinson et al. (2004) (ACWS TR1). The stormwater metal reductions are less than for WWTPs as a proportion (excepting lead, down 85%, Table 5.5). Copper is also down 36% in stormwater, since the flow volume for 1975-85 was lower than

1995-2005 (-22%), and copper load is influenced by flow (Table 5.2) the observed “reduction” might merely be a consequence of the lower flow.

What can be concluded from this brief comparison of past and current pollutant inputs to the Adelaide coastal zone is that the overall loadings have fallen significantly as summarised in Table 5.5 below.

Table 5.7. WWTP and stormwater loads by input zone with percentage change in totals, overall, by source type and by input zone.

	Flow GL	% tot	Area km ²	Yield (mm)	Susp Sed	Totals	TKN	NOKN	TotalP	NH ₃ -N	Total Cu	Total Pb	Total Zn
1975-85 Loads	TOTAL	155.6	2498.7	62.3	15692	3553.3	2831.9	1220.8	592.5	2165.1	17.4	13.3	52.4
	WWTP	58.9	37.8		6536	2309.0	1663.0	1145.5	543.6	1157.7	13.5	3.3	36.3
	Stormwater	96.7	62.2	2498.7	38.7	9155	244.3	75.4	48.9	7.4	3.9	9.9	16.1
	Northern	44.6	28.7	1088.9	41.0	5030	863.3	541.9	220.9	43.3	2.9	1.4	10.1
	Banker Inlet	33.0	21.2	439.4	75.2	2874	1455.2	1307.8	150.1	128.5	2.2	5.8	11.6
	Central	50.4	32.4	451.9	111.6	6767	976.7	379.3	208.0	142.1	11.4	5.0	28.0
Southern	27.5	17.7	518.5	53.1	1021	258.1	108.5	35.1	0.0	0.9	1.0	2.7	
Current Loads	TOTAL	176.9	2492.8	70.9	8428	2357.7	1883.7	473.6	355.2	1509.3	4.27	1.69	13.87
	WWTP	62.0	35.0		1579	1204.2	766.8	437.5	335.0	504.0	2.96	0.21	3.69
	Stormwater	114.9	65.0	2492.8	46.1	6849	153.5	117.0	20.3	5.3	1.31	1.48	10.19
	Northern	51.5	29.1	1088.9	47.3	3775	519.8	109.1	129.2	173.7	2.0	0.3	3.2
	Banker Inlet	26.0	14.7	439.4	59.2	1279	1028.7	6.3	3.2	1001.4	0.2	0.3	2.3
	Central	60.8	34.4	451.9	134.6	2145	528.5	302.3	150.5	144.6	1.5	0.7	5.3
Southern	38.5	21.7	512.6	75.0	1229	280.7	56.0	72.3	189.6	0.5	0.3	3.0	
Percentage Change	TOTAL	13.6	0.0	-0.2	13.9	-46	-33.6	-61.2	-40.0	-30.3	-75.50	-87.30	-73.53
	WWTP	5.3	-7.4		-76	-47.8	-53.9	-61.8	-38.4	-56.5	-78.17	-93.72	-89.85
	Stormwater	18.8	4.5	-0.2	19.0	-25	-37.2	-30.8	-52.1	-28.6	-66.14	-85.14	-36.69
	Northern	15.4	1.6	0.0	15.4	-25	-39.8	-49.8	-41.5	301.1	-29.6	-76.1	-68.4
	Banker Inlet	-21.2	-30.7	0.0	-21.2	-55	-29.3	-21.8	-97.5	0.1	-90.4	-94.5	-79.8
	Central	20.7	6.2	0.0	20.7	-68	-45.9	-62.1	-27.6	1.8	-86.5	-85.4	-80.9
Southern	39.7	22.9	-1.1	41.3	20	8.8	107.1	106.1	0.0	-46.5	-70.5	10.1	

5.5 Estimated loads and hydrology of southern creeks

The Southern Creeks (Peddler Creek to Sellicks Creek) have not been heavily documented in this study. Sections 3.1 and 3.4 present median concentrations of key contaminants from these creeks. In the main load budgets of inputs to the Adelaide Coastline the Southern Creeks are lumped together as a single input to the system (Section 5.1). In this section individual flow estimates and loads are presented. These estimates are based on modelled flows, assumptions about the influence of developed areas on hydrology and on grab sample data combined with modelled flows to give load discharge relationships. These derived load discharge relationships were then used to estimate loadings to the southern coastal zone.

The southern creeks are ephemeral in nature. In general, they only flow during the winter months. The runoff yield from these catchments is low and is only enhanced from residential areas. Of the Southern Creeks, only Peddler Creek has hydrometric gauging stations. The gauging station at Stump Hill Road AW503543 provided the data for this study. This site accounts for 78.4% of the catchment area and is within 5 km of the coast. Downstream of Stump Hill Road are the residential areas of Moana and Seaford Rise. The other residential area in the catchment is McLaren Vale. These areas account for 12.47% of the total 107.39 km² Peddler Creek catchment area. Moana and Seaford Rise are within 2 km of the sea shore and on the basis of orographic effects their influence on the total annual flow has been assumed to be small. Thus the Stump Hill gauge is assumed to account for the majority of flow to sea in the Peddler Creek catchment.

Flow at Stump Hill Road was estimated as described in TR10 (Wilkinson, 2005), using a simple three box semi-mechanistic rainfall runoff model. These flows were then adjusted for the other creeks according to the ratio of developed or urban area in each ungauged catchment to that in the Peddler Creek catchment (Table 5.6). Previously the Peddler Creek flows had been adjusted solely on the basis of catchment area only, however, this ignored the importance of urbanised area and impervious surfaces in contributing runoff.

Table 5.8. Adjustment to flow estimates for Southern Creeks based on implied impervious area (estimated mean 1985-2005).

	Peddler	Maslin	Willunga	Aldinga	Sellicks	
Flow	2768.8					ML
Area	107.39	33.93	30.28	49.2	6.54	km ²
Yield	25.8					mm
Grey land	13.39	3.90	5.54	11.24	0.78	km ²
Ave rain	617	617	617	617	617	mm
VRC	0.0418					ratio
DCP	4.64					%
% urb	12.47	11.49	18.28	22.84	11.89	%
DCP/%urb	0.372					
DCP from %urb	4.64	4.28	6.81	8.50	4.43	
Yield/DCP	5.56					
Yield from DCP		23.77	37.83	47.26	24.60	mm
Estimated Flow	2768.8	806.6	1145.5	2325.1	160.9	ML
Flow adjustment	1	0.291	0.414	0.840	0.058	

This methodology may need some further development and testing, however, it does provide the potential to compare approximate catchment flows and loads. It is also worth noting that the flows are estimated for the whole catchment and include stormwater producing areas that may

have surface drains that emerge along the coast at a different location. The water quality and loads estimated are based-on grab samples collected at the main outlet of each catchment. In this way the loads give an indication of the likely total load from the catchment but not necessarily from the point of sampling.

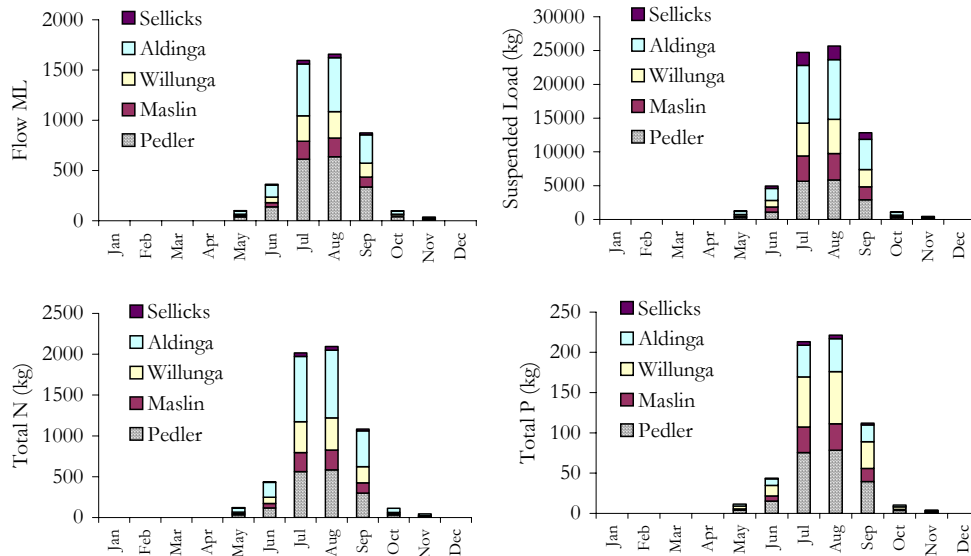


Figure 5.21. Seasonal variation in flow volume and monthly loads for the Southern Creeks (estimated means 1995-2005).

Table 5.9. Estimated loads and yields of contaminants for the Southern Creeks (1995-2005).

Mean Annual Totals	Flow ML	SS kg	TN kg	TKN kg	NO _x -N kg	NH ₄ -N kg	TP kg	
Onkaparinga	18297	128174	19290	16075	3215	6564	1125	
Peddler	1818	16106	1636	1459	176	54	218	
Maslin	530	10795	691	683	7	15	91	
Willunga	752	14022	1091	1036	54	24	181	
Aldinga	1527	24608	2371	2118	253	31	114	
Sellicks	106	5480	121	111	9	3	12	
Southern Creeks Total	GL 4.73	tonnes 71.01	tonnes 5.91	tonnes 5.41	tonnes 0.50	tonnes 0.13	tonnes 0.62	
Yields	Area (ha)	mm	kg/ha	kg/ha	kg/ha	g/ha	g/ha	g/ha
Peddler	10739	16.9	1.50	0.15	0.14	16.4	5.0	20.3
Maslin	3393	15.6	3.18	0.20	0.20	2.2	4.4	26.8
Willunga	3028	24.8	4.63	0.36	0.34	18.0	7.9	59.9
Aldinga	4920	31.0	5.00	0.48	0.43	51.5	6.3	23.3
Sellicks	654	16.2	8.38	0.18	0.17	14.5	4.6	18.3

Figure 5.21 presents monthly total flows and loads for the five catchments between 1999 and the end of 2004. Table 5.6 presents the estimated hydrology. Note that Aldinga Creek is estimated to discharge almost as much water as Peddler Creek and the estimated loads of pollutants are higher from Aldinga Creek except for total Phosphorus which is proportionally higher from Willunga Creek. Willunga Creek was highlighted in Section 3.1 as having higher

nitrogen and phosphorus concentrations than the other creeks. The loads of concentrations suggest that only phosphorus is elevated and that Aldinga is elevated in nitrogen yield. That Willunga Creek has elevated concentrations but not loads (except TP) suggests that there is dilution with rising flow, this might add evidence to the assertion that septic tank drainage is resulting in elevated low flow concentrations (see Figure A3.11). Since phosphorus tends to be particulate associated, it is likely that phosphorus would accumulate in settled particulate matter during low flows and be flushed (resuspended into the flowing water) during storm runoff episodes. With Aldinga Creek, it is likely that the elevated yields of N and P are a consequence of storm-runoff driven processes, i.e. they are rainfall driven and not a consequence of baseflow inputs.

In Table 5.9 estimates of loads and yields for the five southern creeks and includes loads for the Onkaparinga River at Old Noarlunga are presented. Note that because of the low runoff yield the yield of contaminants is significantly lower than from the more residentially developed catchments further north.

6. Zone 2 events, inputs, and near-shore seagrass dieback

The investigation that led to the estimation of past stormwater flows, as reported in ACWS TR10 (Wilkinson, 2005), prompted an examination of the relationship between population, inputs and sea-grass decline (see also Appendix V, which provides figures for inputs from north of Zone 2). Figure 6.1 provides a vivid indication of the rapid development of Metropolitan Adelaide (the areas shaded red are built-up land).

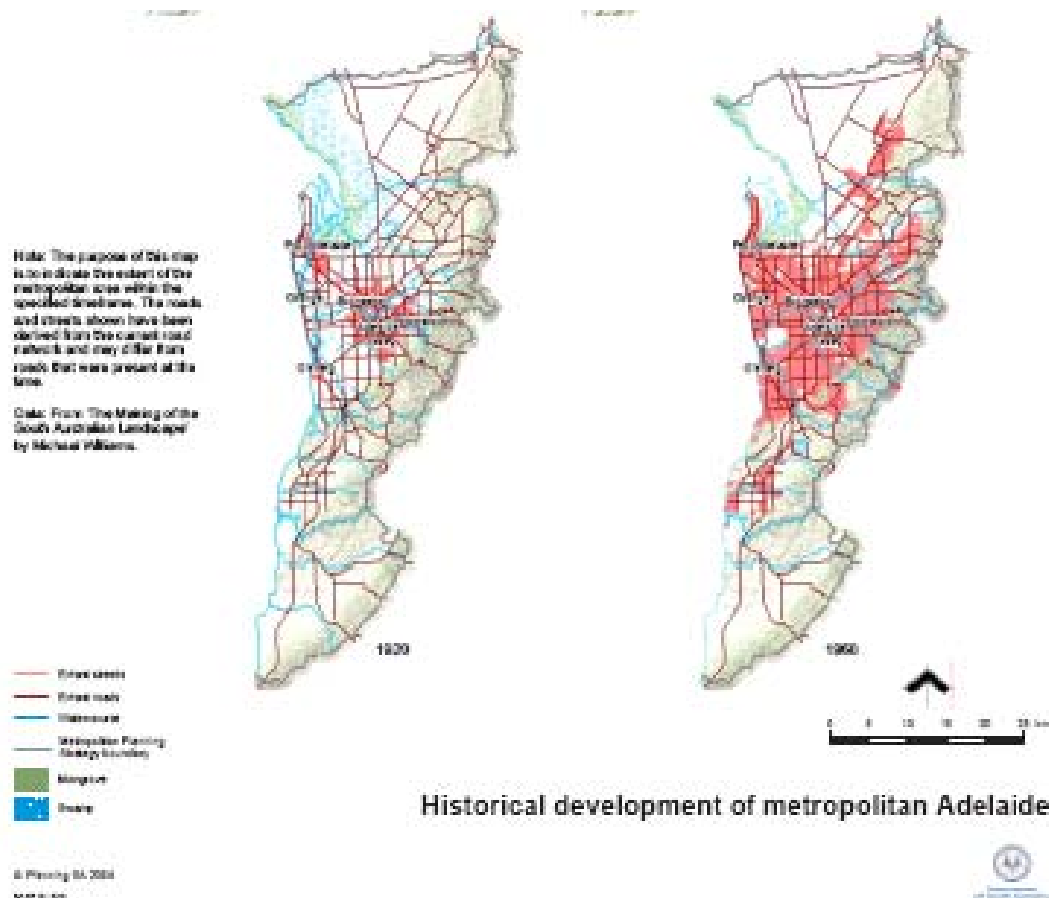


Figure 6.1. Two snap-shots of urban development in Metropolitan Adelaide, 1920 (left) and 1969 (the areas shaded red indicate the built-up area).

Population and the change in the relationship between population and the number of persons per dwelling was used as a surrogate for the rate of residential development in Metropolitan Adelaide between 1930 and the present day (Wilkinson 2005). This information was used to estimate the changes in runoff associated with increasing cover of impervious surfaces and rapid stormwater drainage in order to indicate the likely change in stormwater flow patterns over time. These data have been summarised in Chapter 1 of this report and indicate increasing summer flows to sea through time. The other important changes are highlighted below.

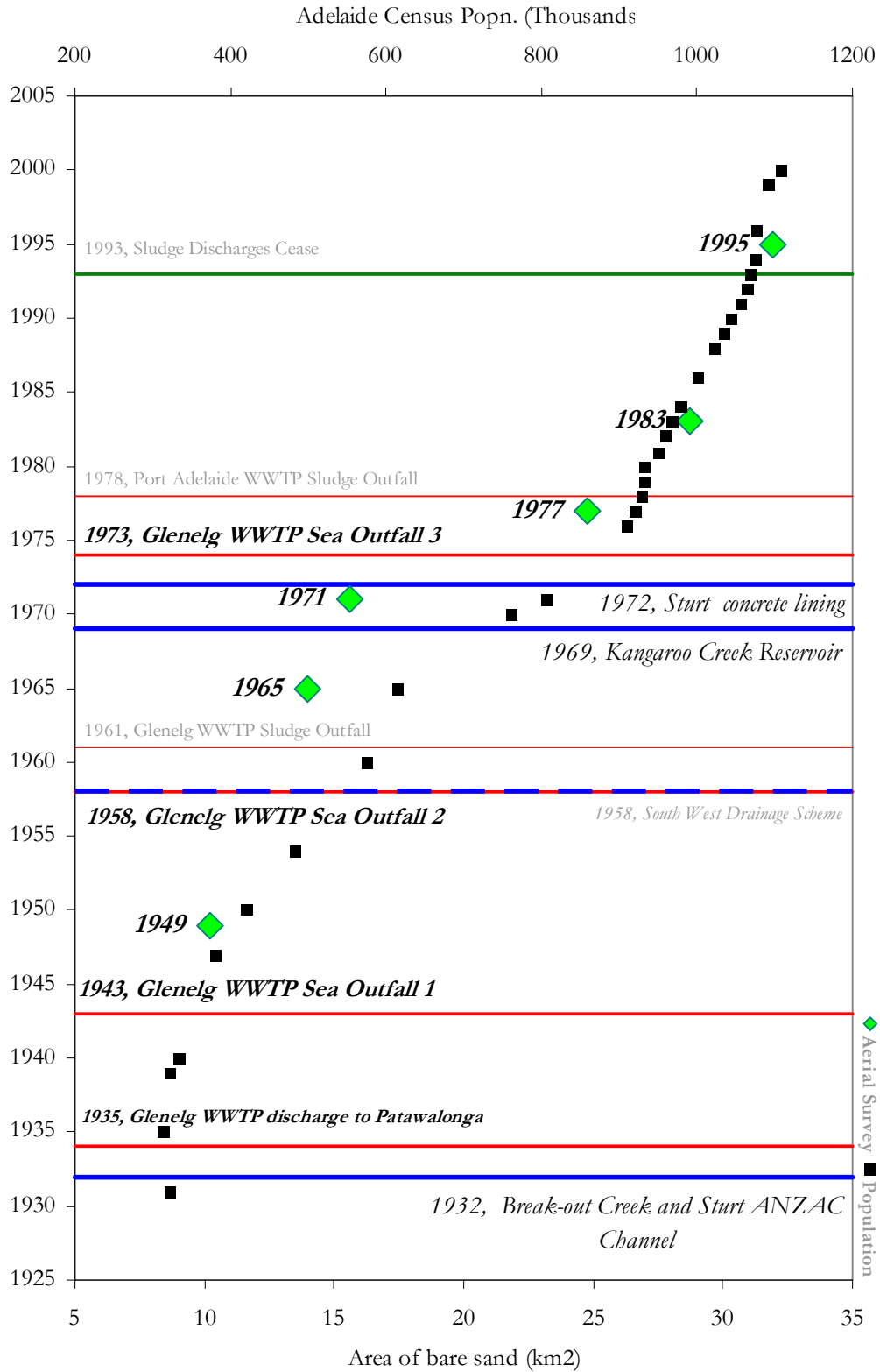


Figure 6.2. The aerially mapped increase in bare sand area off the Metropolitan Adelaide Coastline and the increase in population, with coast impacting events marked as vertical lines.

Figure 6.2 was derived from the bare sand cover data (EPA, 1998) produced from digitised aerial photography of the coastline and Bureau of Statistics census data for Metropolitan Adelaide. The figure shows the increase in bare sand following the rise in population with a lag of around 8 years. EPA (1998) presented a similar diagram. Figure 6.1 takes that original diagram further by correcting certain dates of coastline impacting events and adding a range of other events of significant magnitude. Significant events not previously presented in relation to Zone 2 seagrass decline include the lining of the Sturt Creek between Anzac Highway and Sturt Road. This operation extended the lined section of the channel from approximately 2.5 km to 7.5 km and was completed in 1972. A chronology of events and inputs is provided below.

The series of key events that are displayed in the time line of Figure 6.1 have added-up to be a series of major changes in the condition of the near-shore waters of Zone 2 (as summarised in Table 6.1 and Figure 6.1). The system can be considered as being treated like a large scale experiment where major step changes in inputs are carried-out and the response observed in terms of the retreating sea-grass line. Table 6.2 summarises the changes in system state giving the average or ramp end-points of components of the inputs (annual flow volume, suspended load, and total nitrogen) for each time period.

1. **Up to 1932:** *System might be considered to be in dynamic equilibrium.* Oligotrophic waters with minimal direct freshwater episodic turbid inputs, i.e. minimal shading.
2. **1932/5:** *Step increase in turbidity and nutrients.* Extreme episodic turbid inflows from Torrens Breakout Creek and episodic elevated N/P from the Patawalonga due to the Glenelg WWTP discharge.
3. **1943:** *Step increase in nutrient load, ramping steadily up.* The first of three Glenelg WWTP direct sea outfalls discharging at 5m depth, giving continuous near-shore nutrient enrichment. Continued regular extreme turbid events from the Torrens (also delivering a large N load, Figure 6.2).
4. **1969:** *Step reduction in turbid load.* The reduction of extreme turbid events and a likely reduction in the extent of turbid plumes due to the reduction in liquid volume, and consequent contraction of the area of seagrass that encounter regular shading. Ever increasing N load, more frequent summer turbid events.
5. **2003:** *Step reduction in nitrogen load.* the EIP improvement to Glenelg WWTP reduced the nutrient load from around 600 tonnes per year to 380 tonnes per year, with a potential reduction to 180 tonnes if the treatment process can be optimised.

Clearly the major increase in seagrass decline that is apparent between 1971 and 1977 followed a number of significant coastline impacting events in from 1963 (Figures 6.2, 6.3 and 6.4). Figure 6.3 presents the input data for flows, suspended load and nitrogen load to Zone 2 since 1948. The sharp increase in bare sand area between 1971 and 1977 either follows or coincides with three major events: completion of Kangaroo Creek reservoir, the extension of the concrete lining of Sturt Creek and the commissioning of the third Glenelg WWTP outfall. Figure 6.4 from the Gulf St Vincent Water Pollution Studies (Steffensen, 1985) demonstrates the expansion of bare sand in greater temporal detail by providing snapshots not presented in the EPA aerial photography. This early work is useful in that it also clearly demonstrates the relationship between the areas of loss and sources; there is a clear separation between the central zone of loss and that associated with the Port Adelaide River, i.e. north of Largs. In Figure 6.4 the progression of the line of seagrass loss is consistent with the pattern of mixing and dispersion that might be expected from the north-south tidal motion of the coastal waters. It gives the appearance of a hydraulic mixing or fluid dynamic experiment.

Table 6.1. Chronology of changing inputs to Zone 2 in terms of flow volumes, turbidity and nutrient load.

Period	Inflows	Turbidity	Nutrients
Until 1890	Minimal direct freshwater inflows	Negligible if any	Negligible (Oligotrophic Waters)
1890 to 1932/5	Weir built at the Patawalonga outlet (average 8 GL/y)	Occasional turbid outflows with extreme events (900 tonnes/y?)	Negligible (up to average 7 tonnes N/y)
1932/5 to 1943	Break-out Creek directs Torrens to nearshore (average 55 GL/y)	Regular extreme turbid events 7600 tonnes/y	Glenelg WWTP discharge to Patawalonga; episodic enrichment by stormwater flushing, high N/P load from Torrens (average 150 [+7] tonnes N/y)
1943 to 1969	First direct Glenelg WWTP sea outfall in 1943, second in 1958 (rising from 3.0 to 16.0 GL/y), plus stormwater (+55 GL/y)	Regular extreme turbid events 7600 tonnes/y	Direct continuous enrichment of near-shore waters (rising from 100 to 560 tonnes N/y), episodic high N loadings from Torrens (average 157 tonnes N/y)
1969 to 2003	Kangaroo Creek Reservoir completed; Torrens flows radically curtailed from 47 GL/y to 21 GL/y . Summer flows increase from 0.7 GL/y to 2.5 GL/y. (average annual flow 55 GL/y)	Contraction of major turbid events, expansion of (minor) summer turbid events 2700 tonnes/y	Growth of Glenelg WWTP discharge and nutrient output, third sea outfall 1973 (560 tonnes N/y), plus stormwater (70 tonnes N/y)
2003 onwards	As 1969 to 2003 (average annual flow 55 GL/y)	As 1969 to 2003 2700 tonnes/y	37 % Reduction in Total N from Glenelg WWTP (up to 70 % reduction possible) (375 tonnes N/y), plus stormwater (70 tonnes N/y)

This changing history of flows, suspended load and nutrients sets a context against which to consider the response of the different species of sea-grasses and the nature of those responses. It is known that these plants can adjust their physiological response to suit the light conditions they encounter and it is possible that they can modulate their metabolism of nutrients. These factors and the many other factors that influence seagrass survival are being investigated by the ACWS EP1 team and that task will report these findings in relation to the changing input states encountered since European intervention.

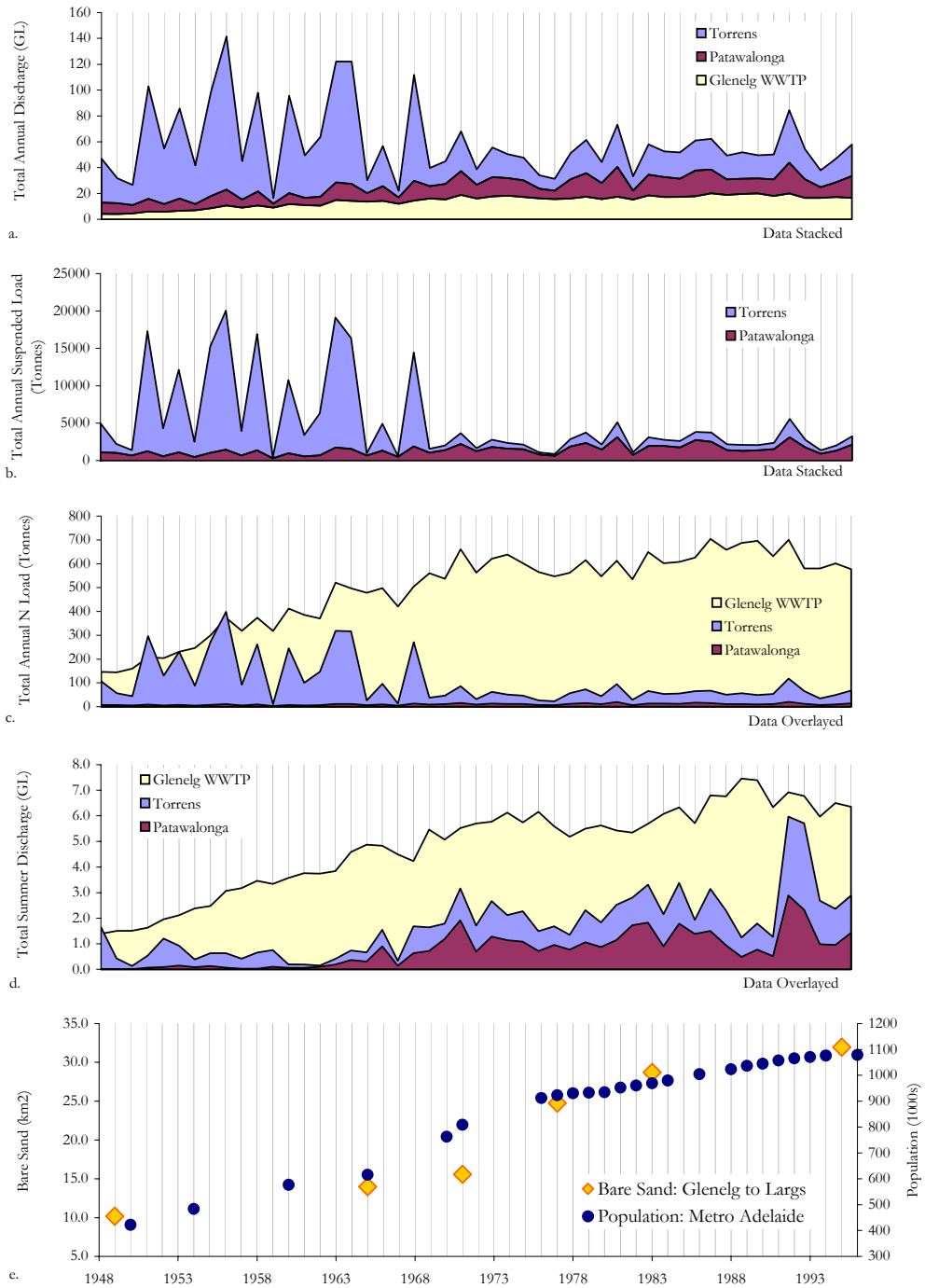


Figure 6.4. Central zone inputs between 1948 and 1996, showing a. total annual flow, b. total suspended load, c. total nitrogen, and d. total flows December to end April.

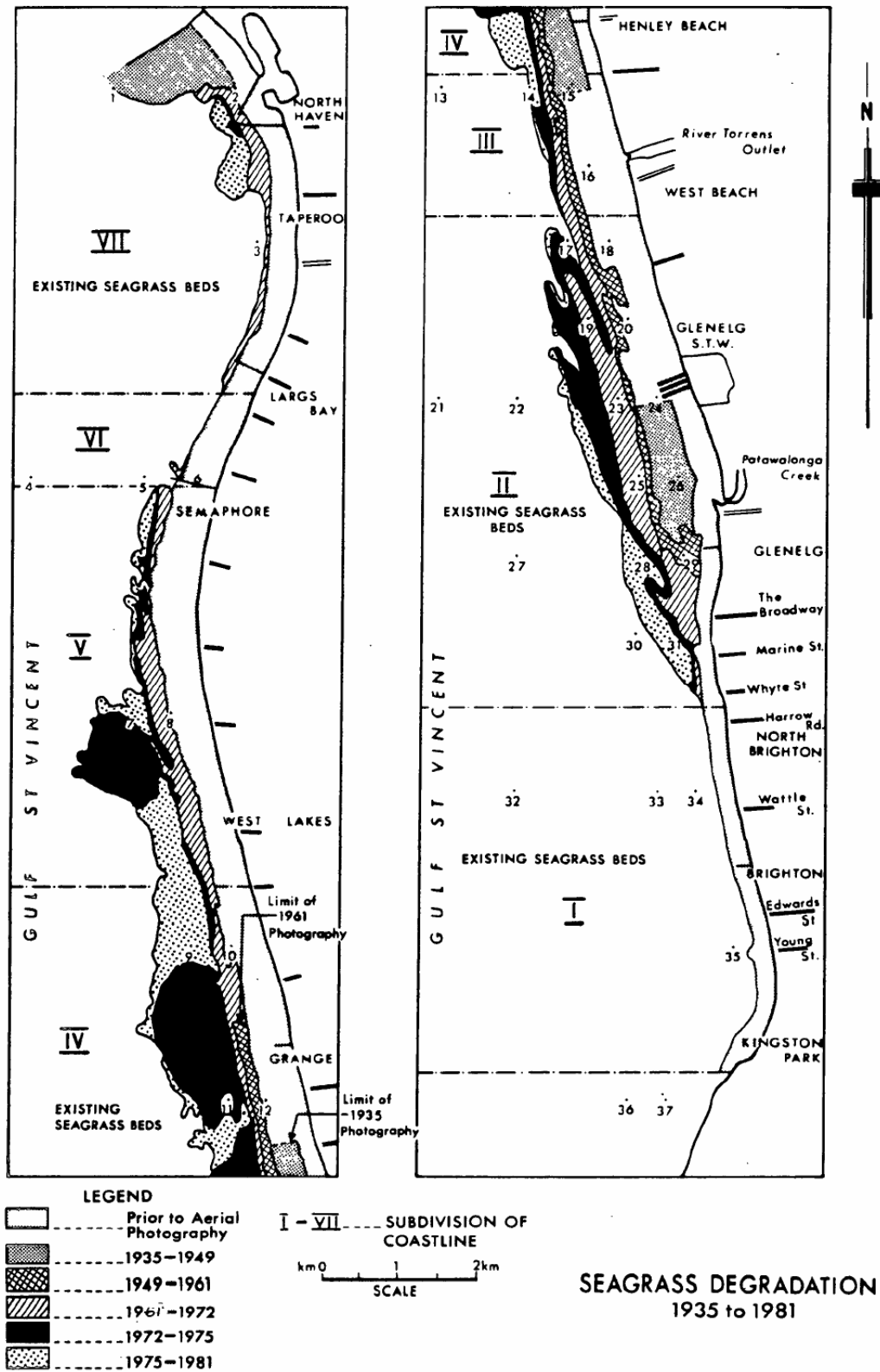


Figure 6.4. The expansion in area of near-shore bare sand in Zone 2 as observed during the Gulf St. Vincent Water Pollution Studies (Steffensen, 1985).

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