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Coastal
Waters
Study



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Audit of the quality and quantity of treated wastewater discharging from Wastewater Treatment Plants (WWTPs) into the marine environment



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Audit of the quality and quantity of treated wastewater discharging from Wastewater Treatment Plants (WWTPs) into the marine environment

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“... the ocean, because of its vast size, was regarded as a limitless oxidation lagoon in which dilution and microbiological agencies could effectively decontaminate the pollutants discharged to it. This view prevailed as late as a decade ago and even today there are some workers in public health sanitation who continue to regard the ocean as the ideal receiving water for the disposal of untreated sewage and other liquid and solid wastes. There have been many examples where local oceanographic morphometry has resulted in catastrophic episodes because of the inadequacy of the receiving body of seawater to deal with pollution. The problem which faces this Committee is, can this happen in Gulf St Vincent?”

Quote from “Gulf St Vincent Water Pollution Studies, Progress Report May 1973” Engineering and Water Supply Department of South Australia.

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Executive Summary

The ACWS has collated a wide range of historical and contemporary data on treated wastewater and digested sludge composition and volumes discharged to sea. Paper records have been entered and charts digitised and these data processed to recreate the full record of flows direct to the coastal zone. Currently the WWTPs are the best represented component of the inputs to the Adelaide coastal zone and these data represent a valuable resource and will significantly contribute to the possibility of running model hindcasts of seagrass impact scenarios.

Archival literature has provided a history of the Metropolitan Adelaide WWTP developments and more recent information indicates the current direction of wastewater treatment and disposal with the EIP improvements in quality and water reuse and aquifer storage and recharge (ASR) proposals.

WWTPs (including Port Adelaide WWTP) contributed around 43 % of the water discharged annually from land-based sources for the period April 2001 to April 2003.

Historically, winter flows accounted for 60 % of wastewater discharges while summer flows account for 40 %. Re-use schemes at Bolivar and Christies Beach have reduced the summer component of the annual flow to sea to around 36 % at each plant.

More than a quarter of the total (approximately 2000 gigalitres) of treated wastewater discharged by Bolivar, Christies Beach and Glenelg between 1945 and 2001 came from Bolivar in the last 20 years.

Since opening Bolivar has accounted for around 55 % of direct WWTP discharges to Gulf St Vincent. With the commissioning of the High Salinity WWTP this proportion has increased to 59 %, this figure includes an offset for increased usage of the Virginia Pipeline Scheme. Glenelg accounts for around 27 % and Christies Beach for the remainder.

The cessation of digested sludge discharge to sea in 1993 dramatically reduced the WWTP impacts on the Adelaide coast. Further improvements in treatment have reduced suspended solids from around 8500 T/y in the 1980s to 2000 T/y (current data).

The seagrass decline in the areas impacted by the Glenelg and Port Adelaide sludge outfalls differed greatly. This was due mainly to differences in the hydrodynamic environments in both locations. The Glenelg site was subject to stronger currents and mixing and dispersion of the sludge plume compared to the PAWWTP site off Point Malcolm where seagrass decline was believed to have been driven by light reduction resulting from daytime plumes of highly turbid sludge shading the sea floor.

The Virginia Pipeline Scheme and the Willunga Basin Transfer have started to reduce the volumes of flow to sea. The Virginia Pipeline Scheme reduced the flow to sea from Bolivar by an average of 11 GL (31% of the annual discharge) from 2000 to 2003. The Willunga Basin Transfer reduced the discharge to sea from Christies Beach WWTP by around 20% in 1999/2000 and 2002/3.

There have been major improvements in total nitrogen loads to the Gulf since the mid-1980s and mid-1990s. The annual nitrogen load from Bolivar peaked in 1996/7 at 1700 Tonnes. The total nitrogen load (excluding Port Adelaide WWTP) that season was around 2500 Tonnes, similar to the annual average load from 1986 to 1992 when the sludge outfalls were still in operation.

The EIP programme of SA Water had reduced the annual nitrogen load to 1000 Tonnes in the 2002/3 season. When the Bolivar High Salinity Plant is commissioned, the upgrades of Glenelg and Christies Beach have been optimised and with re-use from Bolivar to Virginia increased by 5 GL, the annual nitrogen load may fall to less than 700 Tonnes per year.

The reduction of ammoniacal nitrogen (NH₃_N) in treated wastewater has been an objective of improvements to the plants. Bolivar has seen an 8-fold reduction in NH₃_N since the late 1990s from 800 T/y to 100 T/y in 2002/3.

Heavy metal concentrations are much lower than in the 1970s, and loads have been reduced dramatically since the early 1990s with reductions in the total wastewater load of between 75 and 95 % for chromium, lead, nickel, iron, zinc, silver and cadmium. Loads of boron, aluminium, manganese and mercury have fallen between 34 and 53 %, molybdenum and arsenic between 18 and 14 %. The heavy metals with the greatest loads are zinc and copper. The mean treated wastewater concentrations of copper, cobalt, zinc and chromium from Bolivar, Glenelg and Christies Beach are all in excess of the ANZECC/ARMCANZ (2000) trigger level for the protection of 95 % of marine life. Of these metals, copper is between 20 and 43 times the trigger value and cobalt and zinc are only between 2 and 4 times the trigger value.

Organic contaminants such as pesticides, PCBs, PAHs, POPs etc. have only been detected sporadically. This lack of detection suggests that these substances, if present at all, are present at very low levels and are of low significance when set against the other bulk inputs of substances known to have a major impact on the system.

Given that wastewater inputs have excellent data provision relative to the other sources, project effort in ACWS IS1 needs to be concentrated on quantifying the aspects of the other inputs that are less well represented.

Studies of Seagrass Loss

Previous studies have lumped all seagrass loss episodes together when investigating the relationship between land based discharges and seagrass loss. The authors suggest that due to the differing nature of the degradation mechanisms and sources of contamination that certain episodes of seagrass loss should be considered in isolation. Namely, that the sewage sludge associated seagrass loss is anomalous to the general pattern of decline along the beach face zone. This area of loss should be subtracted from the total when investigating the impact of WWTP treated wastewater and stormwater discharges.

Borum (1985) indicated the importance of grazers in the ecosystem interactions associated with epiphyte growth. If contaminants in waste and stormwaters impact on grazers, this will exacerbate any affect from epiphyte growth. Previous investigations of seagrass decline along the Adelaide coast have made no reference to grazer populations. Has there been a loss of grazer populations and if so what impact might this have had on epiphyte build-up?

Sediment Resuspension

Previous studies have demonstrated the impact of light loss due to turbidity on the self-maintenance capacity of seagrass rhizomes. Adelaide coastal waters become turbid in response to wind wave activity, in the absence of recent storm or waste water inputs of suspended matter. It has been suggested that the weak coastal circulation is unlikely to disperse the fine material that is resuspended. To what extent is this material (already stored in the near shore zone) responsible for the weakening and decline of seagrass?

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1. Introduction

This report presents a history of the direct gulf discharging wastewater treatment plants (WWTPs). The Port Adelaide WWTP discharges indirectly via Barker Inlet and for the purposes of the project Barker Inlet is being treated as a single input. By the end of 2004 Port Adelaide waste water will be received at Bolivar and the treated water will discharge along with the existing Bolivar flow to sea. This report also provides a brief overview of Environment Improvement Programs (EIPs) that are reducing pollutant loadings from WWTP sources, thus improving both the quality and quantity of treated wastewater discharged to sea. Archive data that has been painstakingly assembled from paper records is presented. These records provide a detailed history of direct gulf WWTP discharges since 1945. These data will contribute to modelling and hindcasting activities within Sub-Task 2 of the ACWS project PPM2 *Physical Studies in the Adelaide Coastal Waters*. The report provides summaries of current and historical concentrations and loads of nutrients, suspended sediments and metals demonstrating the improvements that have been achieved since the early 1990's. Both treated wastewater and sludge data are reported and the scale of the sludge discharges in terms of loads of contaminants is shown. The relative contributions of pollutants from Bolivar, Glenelg and Christies Beach are provided using current and archived data. Information regarding organic contaminants such as pesticides, persistent organics and poly-aromatic hydrocarbons is presented. Finally the results of this broad analysis are summarised and the continuing direction of the input studies highlighted.

2. Historical Background

2.1 Glenelg

In 1903 an Act of Parliament authorised the provision of sewers for Glenelg, the construction of which commenced in 1904 (SADEC, 1975). The sewers transported treated wastewater north where it was pumped over the Patawalonga to septic tanks in the sand hills and finally discharging to the sea. The septic tanks were later abandoned and the sewage was irrigated over the sand hills, seeping away to the coastal zone (SADEC, 1975).

The WWTP at Glenelg was the first activated sludge plant in Australia and was commissioned in December 1932 (SADEC, 1975). The treated wastewater from the plant ran into filter beds from which it seeped, on the landward side through swamps and in streamlets across Military Road into lagoons and meadows where the nutrient enrichment supported “abundant vegetation”, and on the seaward side to the beach and sea (Johnston, 1934). The landward seepage eventually entered the Patawalonga, occasionally overflowing directly into the river. On the seaward side the situation was described as “..very unsatisfactory..” and the affected stretch of beach, some several hundred metres, was considered “ very much defaced as a result of the discharge...” (Johnston, 1934). The seepage entered the beach zone beneath a rich growth of vegetation and formed a number of streamlets, the area was heavily discoloured by algal growth giving a reddish or yellowish-brown hue (Johnston, 1934). As a result of this situation it was proposed that the treated wastewater be piped into the Patawalonga (Johnston, 1934).

The first direct piped discharge from the Glenelg Works to the Adelaide coastal zone was not commenced until January 1943 (EWS, 1973), when a 0.6 m diameter outfall was constructed, discharging 350 m off-shore at a depth of 4 – 5 m (Lewis, 1975) (see Figure 1 for outfall location).

In 1956 construction of a second activated sludge plant (Plant B) was commenced. The new plant increased the works capacity from a population of 40,000 to 100,000 (Dinesh, pers. comm.) The 0.81 m diameter plant outfall was commissioned in 1958 and discharges adjacent to the original outfall (Lewis, 1975) (see Figure 1).

Digested sludge at Glenelg was disposed of by drying on sand beds which covered an area of 0.465 Ha in 1933, increasing to 0.790 Ha in 1943 and by 1949 the beds covered 2.32 Ha (EWS, 1971). Half of the drying area was lost in 1961 due to expansion of the treatment plant (EWS, 1971), and the first of two sludge off shore outfalls was installed at Glenelg, discharging digested sewage sludge from a 3.2 km pipe (CCE, 1985) (see Figure 1). Sewage sludge comprises approximately 1 to 2 % of the total volume of sewage treated and the sludge itself comprises around 2 % solids (EWS, 1991b). The rate of sludge discharged from the Glenelg outfall was recorded in monthly data abstracts giving the mean rate of discharge in thousands of litres per day (m^3 per day) from May 1962 up to the date of the final cessation of pumping to sea in 1993. The source for these data is archived paper records held at the SA Water depot at Thebarton.

Operation of the first experimental sludge outfall at Glenelg was problematic, repairs having to be made on five occasions between September 1962 and 1963. In 1967 construction of the third (Plant C) activated sludge train commenced, shortly afterwards the sludge outfall was upgraded and extended in 1968. The Plant C outfall has 0.91 m diameter and was commissioned in 1973 (Lewis, 1975), the total works capacity being increased to serve a population of 250,000 (Dinesh, pers. comm.).

The increased diameter upgraded sludge outfall extended an additional 0.7 km offshore to 3.9 km offshore (CCE, 1985). The initial discharge regime involved night-time pumping for around five to six hours on ebb tide (SADEC, 1975) at a rate of 17 litres/sec (CCE, 1985).

Changes in seagrass species density around the Glenelg sludge outfall were observed as early as 1968 (SADEC, 1975). Prior to the opening of the first sludge outfall in 1961, EWS surveys

showed *Amphibolis antarctica* to be the dominant species (SADEC, 1975). When the same area was re-surveyed in 1975, the species composition was dominated by *Posidonia*; *Amphibolis* having largely been displaced (SADEC, 1975). In March 1973 a 20 m² patch of bare sand at the sludge outfall was recorded surrounded by dense *Posidonia* growth. By May 1974 there was only sparse and stunted *Posidonia* growth within a 50 m radius of the outfall (SADEC, 1975). Sludge particulate matter was observed in the vicinity of the outfall, lying in a thin layer on bare sand and in pockets among the seagrass root system (SADEC, 1975). The SADEC (1975) report also includes two-dimensional plots of contours of heavy metal concentrations of samples collected around the Glenelg outfall. The impact of the sludge outfall was regarded as minimal and this encouraged plans for the Port Adelaide sludge outfall which was commissioned in 1978 (see below for detailed history). The Port Adelaide sludge discharge was found to have a major impact seagrass on health and in 1991, EWS reported plans to cease sludge discharges from both Glenelg and Port Adelaide and pump sludge to Bolivar via a 37 km pipeline (EWS, 1991a). The estimated daily total sludge requiring pumping to Bolivar was around 1.2 ML/d in 1988.

The sludge discharges to sea finally ceased in late 1993 (October). An EPA report of 1998 indicates that the discharges ceased in 1992. The 1993 date has been confirmed by Sickerdick (pers. comm.).

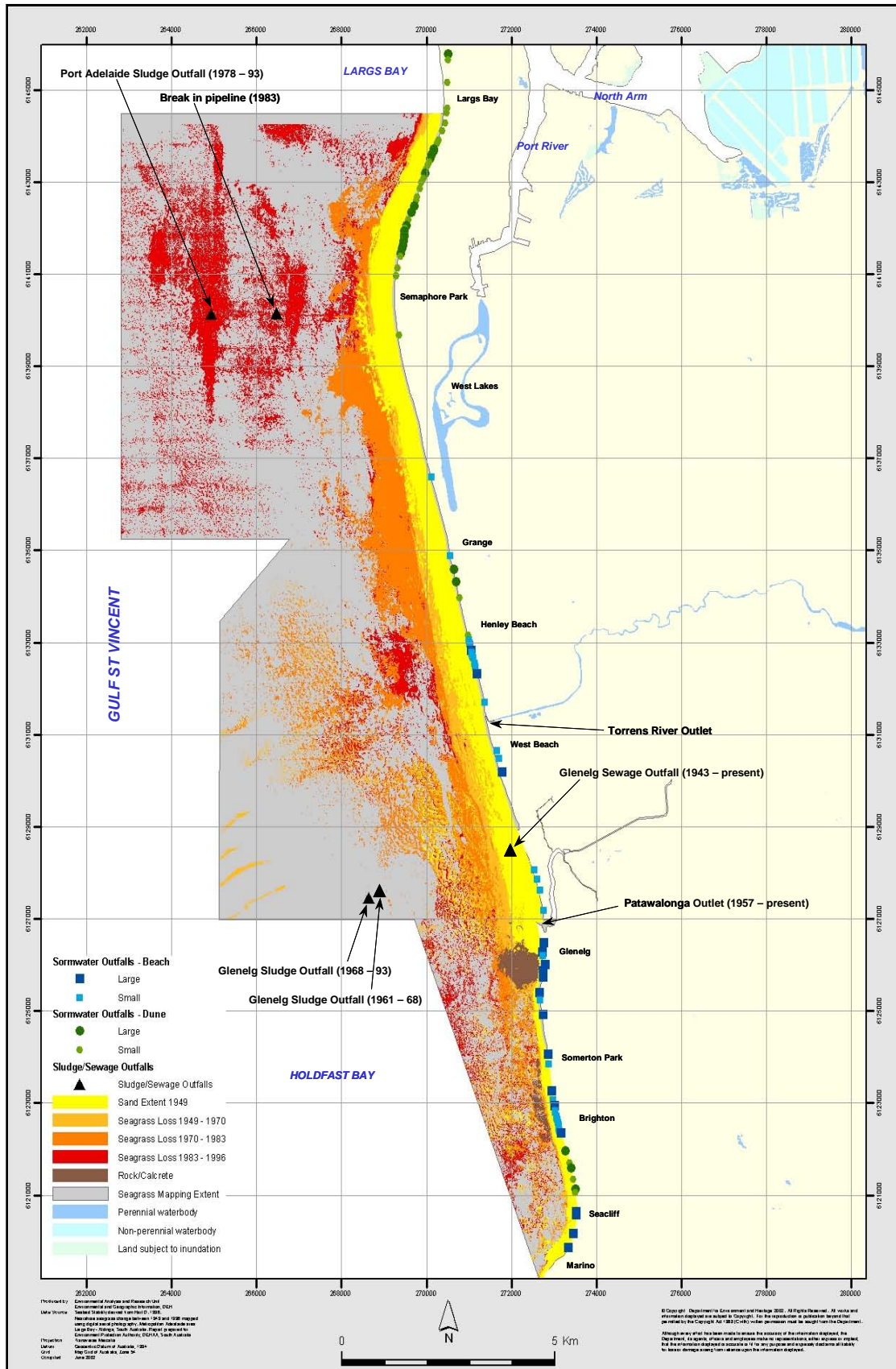


Figure 1. Map showing the locations of point-source discharges and seagrass degradation in the central Adelaide metropolitan coastal zone (from Seddon, 2002).

Recent developments at Glenelg include the decommissioning of Plant A in 2002 and upgrading of Plants B and C, to the “Integrated Fixed Activated Sludge” (IFAS) system to increase nitrification and denitrification, as part of the Environmental Improvement Programs (EIP). In addition, a new Plant D was constructed in 2002/03. The current capacity of the Glenelg WWTP is a design annual average flow of 60 ML/d (Dinesh, pers. comm.). The objective of the upgrading is to reduce the treated wastewater mean total nitrogen concentration to 10 mg/L (Sickerdick, pers. comm.) and is expected to be completed in June 2004 (Dinesh, pers. comm.).

2.2 Port Adelaide

The Port Adelaide WWTP (PAWWTP) currently discharges into the South Arm of the Port Adelaide River which flows into Barker Inlet. This plant will be decommissioned soon and by late 2004 will discharge, via a new high salinity treatment plant at Bolivar, directly into Gulf St Vincent (GSV) at the same location as the main Bolivar Outfall (Sickerdick *pers comm.*). The focus of the ACWS Input Studies is to characterise the inputs to GSV. The Barker Inlet is being treated as a “black-box” with a single input and the Centre for Water Research at UWA has an existing model of the system which will be used in their investigations. As the PAWWTP treated wastewater is not a direct discharge into GSV it has not been included in the WWTP audit document.

In addition, the Barker Inlet is the focus of a major new study by EPA. This work will involve composite sampling of the inlet mouth and feeder creeks with the emphasis on nutrients and additional metals fluxes and budgets. There are various complimentary aspects to the EPA and ACWS studies that will be discussed during the progress of the study.

The focus of the current text is on describing historical developments at PAWWTP in relation to the creation of the direct gulf sludge outfall that destroyed a large area of seagrass off Point Malcolm.

The first sewerage in the Port Adelaide area was available in 1916, and for the Semaphore Glanville area in 1919 (EWS, 1966). Sewage was pumped via a pumping station on Wellington St., Portland, 7 km to the “Sewage Farm” at Islington where it passed through sand filter beds to join the main Islington outfall which drained into the North Arm of the Port Adelaide River (EWS, 1966). Construction of the WWTP commenced in 1934 and the first flow into the new works, including additional sewage from Albert Park, Queensbury and Woodville, was received on the 28 March 1935, the estimated population at the time being 37,320 (EWS, 1966). In 1951, the Semaphore sewer collapsed and the treated wastewater was pumped into the sump of the Port Adelaide Pumping Station for about two years, awaiting completion of the Ethelton Pumping Station in 1953. Plant No. 2 at Port Adelaide came on-line in 1954, to cope with the additional inflow (EWS, 1966). The Port Adelaide WWTP does not discharge directly to the Adelaide coastal zone, but enters the South Arm of the Port Adelaide River and flows into the complex Barker Inlet system. It is important to note here that by 2005 the Port Adelaide wastewater will no longer discharge into the Port Adelaide River, it will be received at the Bolivar high salinity plant (see text on Bolivar).

In 1971 sewage sludge disposal at the Port Adelaide Works was reviewed. The development of the West Lakes complex meant that the practice of lagooning the sludge at the WWTP was no longer acceptable, and a sea-outfall was proposed as an alternative (EWS, 1971; SADEC, 1975). Surveys of the sludge outfall from Glenelg WWTP suggested only a minimal impact on local seagrass communities (see SADEC, 1975). A discharge point 4.5 km off Point Malcolm was selected for the PAWWTP sludge outfall (Figure 1).

Construction work commenced in April 1976 and was intended to take 17 weeks. Work was hampered by sea conditions, pipe breakages and unexpected difficulties in excavating the trench (Ochota, 1981) and the work took approximately two years to complete (Ochota, 1981) and involved the digging of a trench to bury the pipeline. This was unlike the Glenelg situation, where the pipes were laid above the seabed and disturbance of the seagrass mat was minimal (Steffensen, 1985). Off Point Malcolm the disturbance of the seagrass mat was considerable and

is clearly visible on aerial photographs (e.g. Hart, 1997). Where the depth of unconsolidated sediment was shallow, the trench was narrow, around 2.6 m, but further inshore where the depth to the solid calcrete layer was greater, the trench was up to 20 m wide, as a result of slumping of the sides and undercutting of the seagrass mat (Steffensen, 1985).

In late July 1978, the sludge outfall was commissioned (Thomas, 1981). Figure 2 shows a photograph of the outfall during operation, taken by marine survey divers (see Neverauskas, 1984). Operation of the pipeline was originally intended to take place, at night, between low and high tide when the current strengths would afford greatest dispersion and to minimise photosynthesis interruption by the turbid plume (Thomas, 1981). This was prevented by the nature of operations at the plant and it was necessary to split the discharge into two periods, one at night and the other during working normal hours (Thomas, 1981). The daytime discharges gave rise to complaints by the public about the visible sludge slick and it was proposed to discontinue daytime discharges (Thomas, 1981).

The first major problem with the Port Adelaide pipeline was encountered in 1979, when, on 23 March, Port Adelaide WWTP operators received a telephone call reporting a visible sludge field off-shore (Thomas, 1981). Subsequent investigations discovered a 2 m section of pipe missing 3 km off shore. The option of stock-piling the sludge on-site until the pipe was fixed, had, prior to the pipeline being commissioned, caused odour complaints, so night time only discharging via the broken pipe continued until its repair was complete on 8 April. This resulted in just over 2 weeks of discharge of sludge at that location. Other breakages occurred, one of which lasted for around 8 months (Neverauskas, pers comm.) and Seddon (2002) presents a map, reproduced as Figure 1 above, marked with a breakage in 1983.

The first seagrass survey of the area 1000 metres around where the Port Adelaide outfall would discharge was conducted in November 1974 (Lewis, 1975), showing *Posidonia* to be the dominant seagrass species, with moderate growth throughout the area. The seagrass blades were reported as being relatively young, long and healthy and having little epiphyte cover (Lewis, 1975). A later survey in 1978 prior to the commencement of discharge of sludge from the pipeline showed 80 to 100% cover of pure *Posidonia* in the seaward direction and nearer the outfall “very low cover” of a mixture of *Posidonia*, *Amphibolis* and *Halophila* (Steffensen, 1981). The first re-survey carried-out between November 1979 and April 1980 (Neverauskas, 1985) showed a “..rapid and extensive loss of *Amphibolis* and *Posidonia*...” (Steffensen, 1985) within a 25 to 30 m radius of the outfall (Neverauskas, 1985). Over the remainder of the 200 by 250 m survey grid a 50% reduction in density was observed (Neverauskas, 1985), and by mid 1981 the second re-survey indicated a loss of around 40 Ha of seagrasses. The full extent of seagrass loss around the Port Adelaide sludge outfall in 1983 was 365 Ha as reported by Steffensen (1985) in a north-south running strip extending over 6 km.



Figure 2. The Port Adelaide WWTP digested sewage sludge outfall, discharging 4.5 km off Point Malcolm (Photo by: V. Neverauskas).

In 1990, Gobbie presented a report suggesting alternative disposal options for Adelaide sewage sludges, and in March 1991 EWS presented a concept design report for the pumping of Glenelg and Port Adelaide sludges to Bolivar (as mentioned above, p.4). The sludge outfall was finally decommissioned in late 1993.

The main causes of the seagrass dieback in the Port Adelaide sludge outfall region were considered to be reduced light due to excessive epiphytism driven by eutrophication and light obstruction by the sludge plume (Neverauskas, 1987). A possible reason that the impact of thirty years of sludge discharges from Glenelg is believed to have been so minimal compared to that of the Port Adelaide outfall is the less energetic hydrodynamic environment surrounding the Port Adelaide outfall and hence less dispersal and greater settlement of solids and contaminants (SA Water, 1995). Previous studies of seagrass decline along the Adelaide coastline have included the PAWWTP sludge impact as part of the overall pattern of decline (e.g. EPA, 1998). It might be more appropriate to consider the different episodes of seagrass loss separately, i.e. when examining the historical trend in seagrass decline exclude discrete episodes of seagrass loss, such as that associated with the PAWWTP sludge discharge, from the calculations. The reason for this is that this was a very specific and major episode that was not related to the general upward trends in discharges of reclaimed wastewater and of stormwater. By removing this event from the trend a more realistic assessment of the near shore seagrass loss might be made.

Since the cessation of sludge discharging, seagrass seeds have germinated in the denuded zone recolonising the area from 0 to 33 % cover in a period of eight years (Bryars and Neverauskas, 2002). This is an important finding since it demonstrates that recolonisation can occur naturally once a disturbance is eliminated.

Port Adelaide WWTP will cease operating towards the end of 2004 (Sickerdick, pers comm.) A pumping station (under construction) will divert the works flow to Bolivar where a dedicated high salinity treatment facility will process the screened sewage. A high salinity treatment plant is necessitated by the ingress of saline groundwater into the Port Adelaide sewer network and

constituted up to 48 % of the works inflow (Bramley *et al.*, 2000). The high salinity renders the treated wastewater unsuitable for reuse and this flow will be discharged to sea with the Bolivar effluent.

A large component (40 %, 4000 ML/y) of the low salinity sewage originally contributing to the PAWWTP load came from the Queensbury Pumping Station. This wastewater was diverted to the Bolivar catchment in March 2002 to be treated by the main Bolivar plant. The separation of this low salinity component of the old PAWWTP inflow means that this water is potentially reusable, whereas it would have been too saline if it had remained mixed with the Port Adelaide wastewater.

2.3 Christies Beach

Since its inception this works has seen a ten-fold increase in connected population from around 13,000 in 1973, and approaching 150,000 by 1996 (both estimated figures from EWS data abstracts). The volume of coastal discharge has increased accordingly (see Section 3).

Christies Beach WWTP was first commissioned in 1971 (stage I) for a design population of 50,000 (Dinesh, pers. comm.). The plant outfall discharges at a point 300 m off-shore at a depth of 6 metres. In 1979, the plant was upgraded (stage 2) by duplicating the primary and secondary processes. In 1994, total hydraulic capacity of the plant was increased and in 1996 two additional clarifiers were commissioned.

A major development at Christies Beach has been the Willunga Pipeline Scheme, commissioned in August 1999. The 17 km pipeline carries secondary-treated wastewater to the Southern Vales grape-growing region to irrigate around 850 Ha of new vineyards (Sickerdick and Desmier, 2000) (see Section 3 for re-use values). Subsequent developments may include aquifer storage and recovery (ASR) which has the potential of using 9,500 ML/y (Sickerdick and Desmier, 2000).

In 2002/03, the plant was upgraded to Integrated Fixed-film Activated Sludge (IFAS) to enhance nitrification and denitrification. The IFAS system at Christies Beach is still being tested and optimised (Sickerdick, pers. comm.) and has not yet achieved the target nitrogen reduction (see Section 5 below). Testing and optimisation is ongoing (Dinesh, pers. comm.).

2.4 Bolivar

Bolivar WWTP has discharged into Gulf St Vincent since its commissioning in March 1967 (Lewis, 1975). It replaced the overloaded sewage farm at Islington which discharged into the North Arm of the Port Adelaide River causing serious deterioration in water quality (Steffensen *et al.*, 1989a).

The Bolivar discharge had a rapid impact on the adjacent marine environment, with some 130 Ha of intertidal seagrasses being lost from the mouth of Fork Creek within 18 months of commissioning (Steffensen *et al.*, 1989b). In 1989, that figure had risen to 500 Ha, with another 500 Ha of sub-tidal *Posidonia* lost, and a further 200 Ha where the density was reduced (Steffensen *et al.*, 1989b).

Improvements at Bolivar in recent years include the Virginia Pipeline Scheme which was commissioned in November 1999, and the conversion from trickling filter (secondary treatment) to the activated sludge system. Since commissioning, the Virginia Pipeline has reduced the discharge to sea by around 12,000 ML/y, and the upgrade to activated sludge has had noticeable effect on the treated wastewater nitrogen quality from March 2001 onwards (see Section 5). An increase in re-use to around 15,000 ML is anticipated for the 2003/4 season and further increases in re-use will be driven by demand.

3. WWTP Flow Volumes

Data on sewage flows for the WWTPs discharging into the Adelaide Coastal Zone are available for the majority of the period of operation. Until 1991 these data were only available in paper records and reports or plotted on charts (prior to 1962). These data have been manually entered and digitised from the charts and provide a valuable archive for the project and for future users of data derived from the Adelaide Coastal Waters Study. These data also offer the potential for a more detailed investigation of wastewater contaminant loadings to the Adelaide coastline than has been possible in this report. The available flow records for each WWTP are outlined below.

3.1 WWTP effluent

Glenelg has been discharging directly via an outfall since 1943, and the flow record commenced in 1946 (Figure 3). For Christies Beach and Bolivar recorded flows are available back to the time of commissioning, January 1973 and April 1967, respectively. The historical flow data are sewage flows, i.e. inflows to each plant. Actual values of outflow volumes are not available until later in the period of record. For Glenelg, treated wastewater flow data records commenced in August 1975, records of digested sewage sludge volumes for Glenelg commenced January 1962. For Christies Beach, treated wastewater flow records commenced July 1975 and for Bolivar outflow records were not logged in the archived monthly data summaries until as late as November 1989. For the Port Adelaide plant, only the sludge outfall has ever discharged directly to the coastal zone and data for that discharge commences in February 1979 up to mid-1992. Beyond that period, the mean of values for the appropriate month of the preceding two years has been used up to November 1993

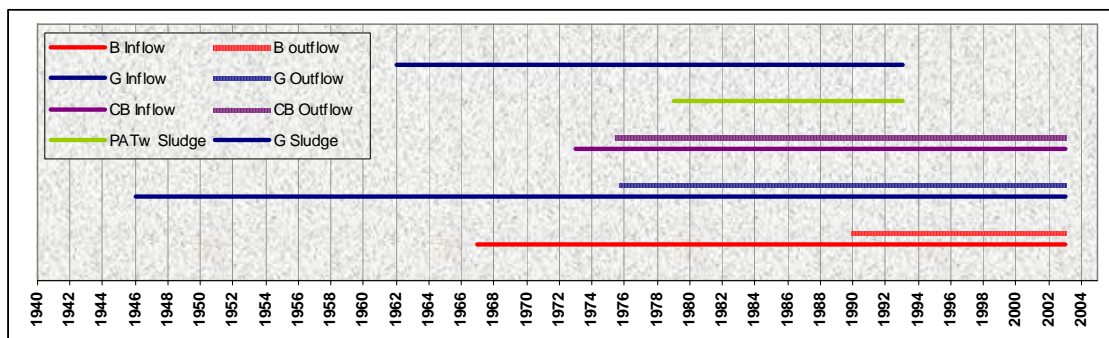


Figure 3. Chronology of availability of raw flow data for Bolivar (B), Glenelg (G), Christies Beach (CB) and Port Adelaide (PA) WWTP sewage (inflow), treated wastewater (outflow) and digested sludge.

At most sites the treated wastewater volume discharged to sea was less than that entering the works this is due to the removal of the sludge flow, re-use and evaporation. At Bolivar evaporative loss rainfall entering the 347 Ha area of lagoons can on occasion result in an outflow greater than the inflow, for example in July 1981, rainfall was in excess of evaporation contributing an additional 190 ML total to the works outflow for that month (Figure 4a). Evaporation at Bolivar annually, accounts for a reduction in flow to sea of around 5.2 GL. In June 1988, the Bolivar Inflow estimate was found to be in error, approximately 50 ML/d over-estimated. It was not clear when the error commenced or whether the instruments were always in error. For the purposes of the current study it was assumed that the error was the result of gradual drift, and the inflows have been corrected on this basis by applying a monthly incremental correction of 1.07 kL (Figure 4a).

Prior to 1991, inflow data only are available for Bolivar. For the other plants, this date is 1975. Figure 4b demonstrates the results of calculations made by ACWS to estimate the outflow from

Bolivar for the full period of operation. Annual summary data is presented for ease of interpretation. The first stage in these calculations was to estimate the net evaporative loss from the treatment lagoons. Evaporation and rainfall data are available from the beginning of operation to 1988. From 1988 it was necessary to use evaporation data for Adelaide Airport. A comparison of three overlapping years of data for the two locations gave a correlation coefficient $r^2 = 0.985$, indicating that use of the airport data gives a good approximation of the behaviour at Bolivar. The annual average net evaporative loss from the Bolivar lagoons is around 5.1 GL/yr. The estimated flow to sea was calculated by subtracting evaporative loss from the works inflow followed by a final correction to account for typical rates of on-site reuse within the WWTP based on monthly means for the period when records exist and prior to the Virginia Pipeline Scheme.

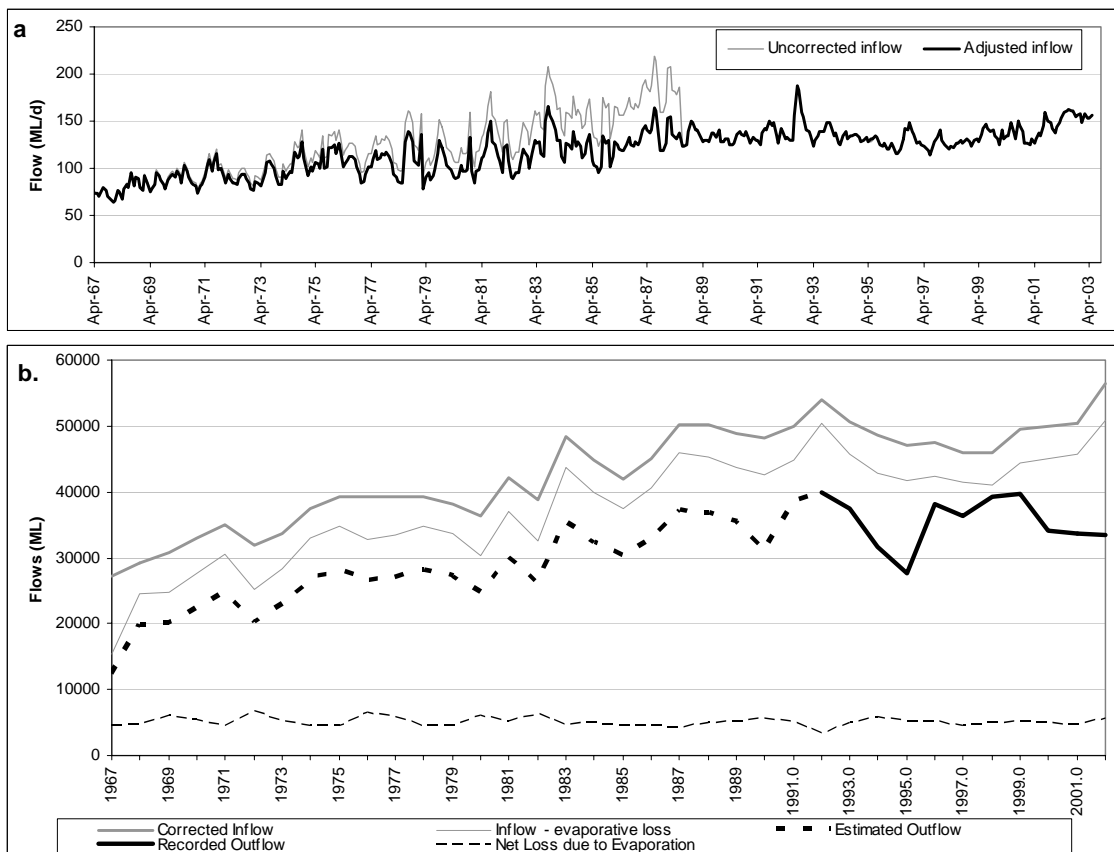


Figure 4. (a) Correction of Bolivar inflow instrumentation error. (b) Estimated annual flow to sea from Bolivar WWTP.

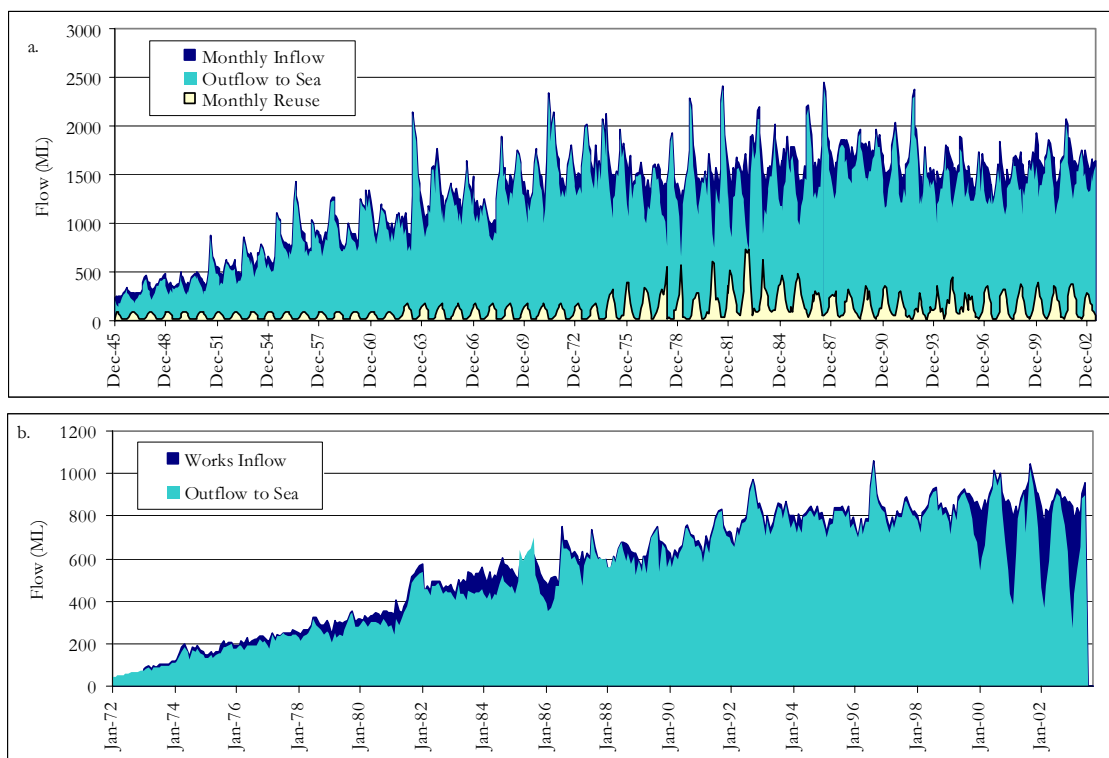


Figure 5. (a) Glenelg WWTP inflow (actual records) and outflow (assuming 20 ML per month works reuse and a halving of consumer usage from 1974 backwards and a further halving from 1962 backwards). (b) Christies Beach WWTP inflow and outflow (estimated prior to July 1975). Note: the data in these plots are overlaid to demonstrate the reduction of the outflow relative to the inflow.

For Glenelg, actual records of monthly flow to sea exist back to 1975, from this period back the flow to sea has been estimated (see Figure 5a). From August 1975 to December 1977 works reuse was recorded at around 20 ML per month. From December 1977 forward this figure rose during summer months. Usage of treated wastewater by consumers was also recorded from August 1975. In the early 1960s reuse by the West Beach Trust commenced and around the mid 1970s re-use increased again with the connection of two golf courses. The sum of works reuse and consumer reuse is presented in Figure 5a. The reuse and hence flow to sea prior to August 1975 was estimated by assuming that consumer reuse was halved in the period prior to August 1975 and that works reuse remained steady at 20 ML per month. Mean monthly consumer reuse values for the period 1975 to 1979 were the basis for these estimates. Similarly, the mean consumer usage was halved again for the period prior to 1962.

For Christies Beach the period for which flow to sea had to be estimated was relatively short. Values for flow to sea were available from July 1975 and it was only necessary to estimate flow to sea from 1972. The reuse was assumed to reduce relative to the inflow volume in similar proportion to the values observed in later years where records are available. Figure 5b clearly demonstrates the impact of the Willunga Basin Transfer Scheme is reducing summer flows to sea from Christies Beach WWTP.

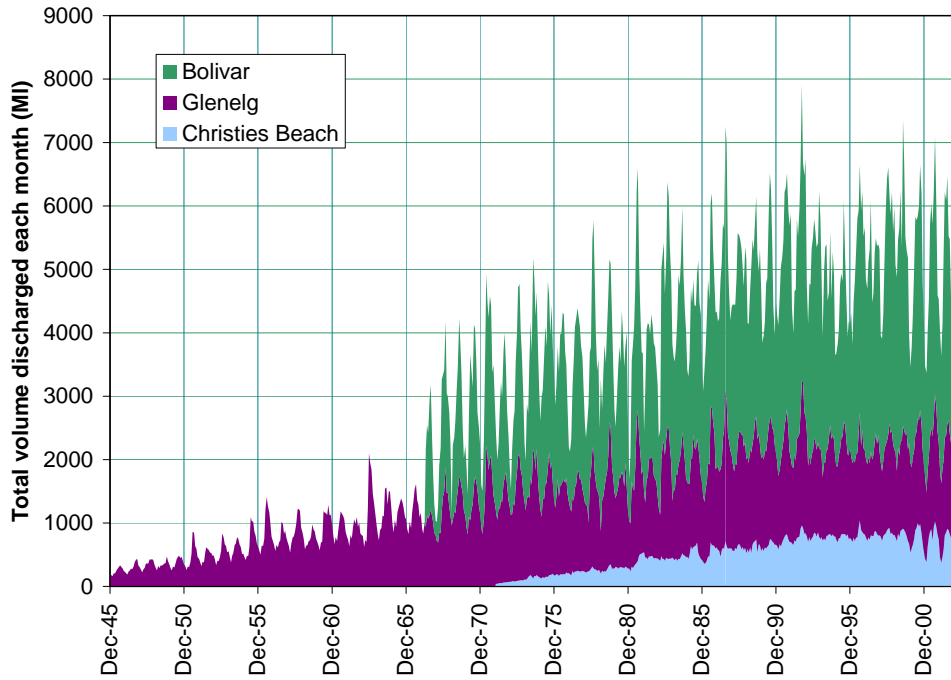


Figure 6. Monthly volumes of treated wastewater by the three WWTPs discharging directly to the Adelaide Coastline since 1945.

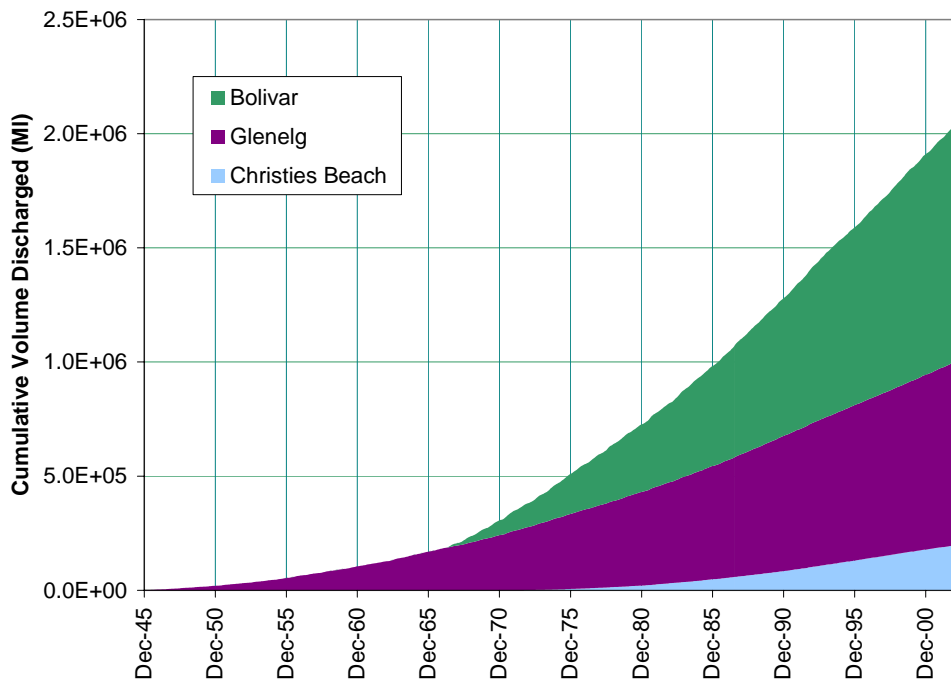


Figure 7. Cumulative monthly volumes of treated wastewater by the three WWTPs discharging directly to the Adelaide Coastline since 1945.

Figure 6 presents the reconstructed and adjusted monthly estimates of mean WWTP outflows, and Figure 7 demonstrates the cumulative totals. The two figures demonstrate the seasonality of the flows and also the extent to which wastewater flows to sea have increased in the last sixty years.

The time-series in Figure 6 are stacked, rather than overlaid, so that the highest value reached in any month represents the summed total discharge from the three direct discharging plants. For example, between December 1985 and December 1990, the highest value shown is approximately 7000 ML, which comprises around 4000 ML from Bolivar, 2300 ML from Glenelg and 700 ML from Christies Beach. The rate of increase in wastewater discharges reflects the increasing population of Adelaide since the mid 1940s. The Glenelg discharge increased rapidly over its first 25 years of operation. Growth of population and associated development in the Southern Metropolitan zone has driven the increase in discharge from the Christies Beach plant. Similarly there has been growth in the volume of treated wastewater from Bolivar (see Table 1).

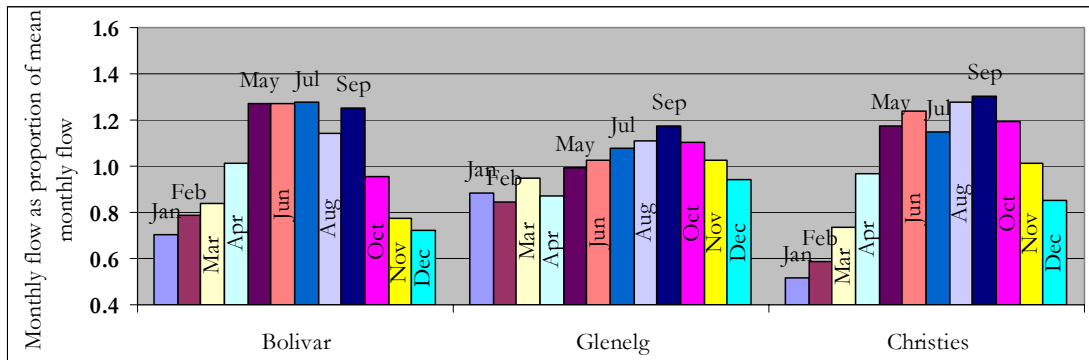


Figure 8. Seasonality of WWTP flows to Sea (mean monthly data for April 2001 to March 2003, net of re-use).

Since 1999, the Virginia and Willunga Pipeline Schemes have helped reduce summer discharge volumes from Bolivar and Christies Beach (Figure 8). The impact of the reuse scheme at Christies Beach is most apparent in Figure 5b. The Willunga Basin Transfer has clearly reduced the summer treated wastewater volume entering the sea. Annual re-use of treated wastewater from Christies Beach WWTP via the Willunga Pipeline has risen from 1364 ML in 1999/2000 to 2299 ML (22 % of plant inflow) in 2002/3. The volume of treated wastewater supplied from Bolivar to horticultural growers via the Virginia Pipeline Scheme has increased to 13,000 ML/y in 2002/3. This is around a 37 % reduction in the volume of treated wastewater that would otherwise be discharged to sea. By late 2004 the flow from Port Adelaide will have been diverted to the new high salinity facility at Bolivar, this will add around 10,000 ML/y to the total treated wastewater discharged to sea from Bolivar and correspondingly increase the loads of contaminants from that source. The increase in flow volume will be offset by the reduction in total nitrogen concentration in the treated wastewater.

The seasonality in treated wastewater flows is also affected by groundwater intrusion into the sewer network (Bramley *et al.* 2000). The sewers act in a similar manner to field drains, during the winter month's rainfall elevates the water table in the Adelaide Plains and the water seeps into the network. Bramley *et al.* (2000) estimated that on average 5 % of the inflow to Bolivar and 4 % of the inflow to Glenelg was from groundwater. Table 1 demonstrates the seasonal split between winter and summer flows, with between 57 and 53 % of treated wastewater volume being discharged in the wetter six months from May to November, and the relative contribution from each plant being approximately the same between summer and winter.

Table 1. Decadal totals, seasonal totals (ML) and percentages of treated wastewater discharges from the three WWTPs discharging to the ACWS coastline 1941 to 2001.

All flows in ML	Bolivar	<i>% 10 yr total</i>	Glenelg	<i>% 10 yr total</i>	Christies Beach	<i>% 10 yr total</i>	Total for Three WWTPs	<i>% 60 yr total</i>
1941-50 total	-	-	19,647	<i>100.0</i>	-	-	19,647	<i>1.0</i>
% 60 yr total	-	-	1.0	-	-	-	-	-
1951-60 total	-	-	85,162	<i>100.0</i>	-	-	85,162	<i>4.3</i>
% 60 yr total	-	-	4.3	-	-	-	-	-
1961-70 total	64,826	<i>32.2</i>	136,310	<i>67.8</i>	-	-	201,136	<i>10.2</i>
% 60 yr total	3.3	-	6.9	-	-	-	-	-
1971-80 total	231,153	<i>54.9</i>	169,135	<i>40.2</i>	20,452.1	<i>4.9</i>	420,740	<i>21.3</i>
% 60 yr total	11.7	-	8.6	-	1.0	-	-	-
1981-90 total	307,255	<i>55.6</i>	182,278	<i>33.0</i>	63,485	<i>11.5</i>	553,019	<i>28.0</i>
% 60 yr total	15.6	-	9.2	-	3.2	-	-	-
1991-2000 total	396,899	<i>57.3</i>	191,404	<i>27.7</i>	103,891	<i>15.0</i>	692,194	<i>35.1</i>
% 60 yr total	20.1	-	9.7	-	5.3	-	-	-
60 Year Totals	1,000,133		783,937		187,829		1,971,898	
% 60 yr total	50.7		39.8		9.5			
Summer total, %	421,891	<i>42.2</i>	334,118	<i>42.6</i>	87,558	<i>46.6</i>	843,567	<i>42.8</i>
% 60 yr, % summer total	21.4	<i>50.0</i>	16.9	<i>39.6</i>	4.4	<i>10.4</i>		
Winter total, %	578,242	<i>57.8</i>	449,819	<i>57.4</i>	100,271	<i>53.4</i>	1,128,331	<i>57.2</i>
% 60 yr, % winter total	29.3	<i>51.2</i>	22.8	<i>39.9</i>	5.1	<i>8.9</i>		

In Figure 7 the cumulative total treated wastewater discharge was demonstrated. Table 1 summarises these flow values for ten year periods from 1941 to the present time. In the period 1940 to 2000, the total volume of treated wastewater discharged to the coastline was in excess of 2000 GL (gigalitres), or 2 billion m³, and 63.1 % of this total was discharged in the latter two decades, i.e. 1981 to the end of 2000. In the last decade from 1991 to the end of 2000, a volume of around 692 GL was discharged, which represents 35 % of the total discharged, comprising 57.3 % from Bolivar, 27.7 % from Glenelg and 15 % from Christies Beach. The winter six month period was found to be May to the end of October for Glenelg and Christies Beach and April to the end of September for Bolivar. These dates gave the winter summer splits that most accentuated the seasonal differences in total flow for the 60 year period presented in Table 1. The summer percentage of the total discharge from each of the works over the full period of record were Bolivar 42 %, Glenelg 43 % and Christies Beach 47 %. Since the introduction of the re-use schemes at Bolivar and Christies Beach the summer percentages of flow from those plants had reduced to around 36 % for 2002, with further reductions anticipated.

In Figure 9, the WWTP flows are set into context against the flows from the larger rivers and creeks. Although the rivers are quite variable in their flow from year to year, the mean annual flows give an indication of their recent relative volumetric input to the Adelaide coastline. Of the major discharges presented the creeks and rivers account for 57% of the inflows, and the WWTPs 43%. Rainwater inputs have not yet been estimated. Note that from late 2004 when the Port Adelaide sewage flow has been fully diverted to the new plant at Bolivar the annual

discharge volume from Bolivar will increase by approximately 12 GL, comprising approximately 8 GL from the high salinity plant and 4 GL via the Queensbury pumping station.

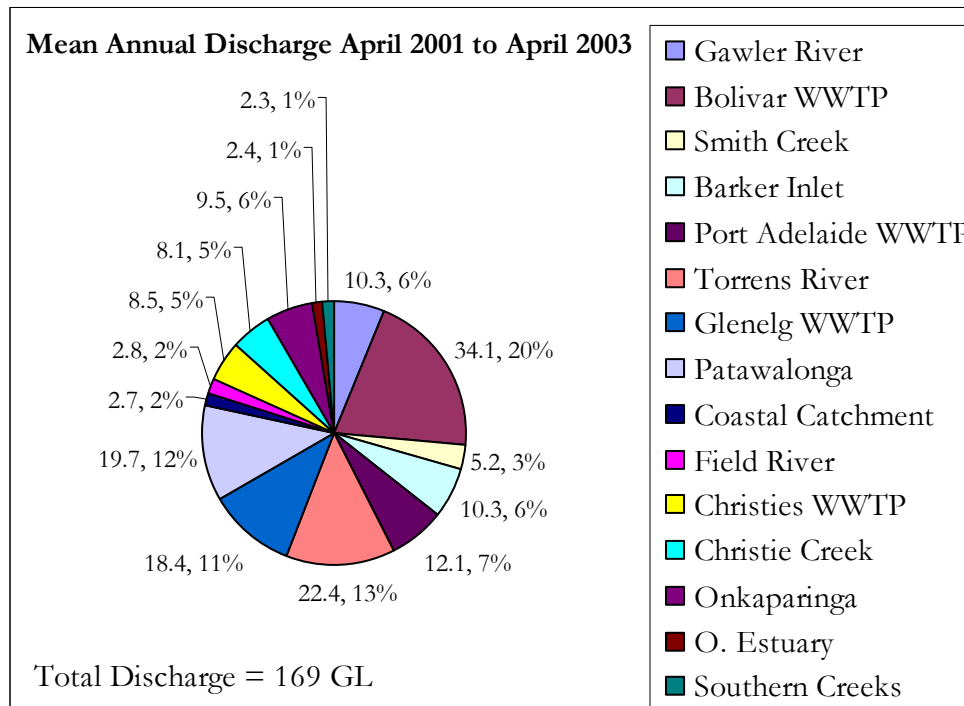


Figure 9. Mean annual flows in gigalitres (GL) from the WWTPs and storm-water sources for the period April 2001 to April 2003. Note that from late 2004 the diversion of flow from Port Adelaide and the Queensbury Transfer will add 12 GL/y to the Bolivar discharge.

3.2 Digested sludge flows to sea

Figures 10 and 11 present previously unpublished data for monthly volumes and the cumulative totals of digested sewage sludge discharged from the Glenelg and Port Adelaide outfalls. These volumes are far less than for treated wastewater and are not included in the preceding examination of flows to sea. The discharges of digested sewage sludge are significant due to the impact they had on the marine environment surrounding the point of discharge and because of the high concentrations of nutrients and metals they contained. As shown in Figure 10, the Glenelg sludge discharge increased in rate over the period of operation. The mean daily discharge in the first three years of record was 170,000 L/d (or to use a unit more easily visualised, 170 m³/d), by the mid 1970s, this figure had risen to more than 550,000 L/d. The total volume of sludge discharged by the Port Adelaide sludge outfall was less than that from Glenelg and at a slightly lower rate. The total volume from Glenelg was in excess of 5,115 ML during its 30 years of operation, whereas Port Adelaide discharged only 2,120 ML over 15 years. Port Adelaide discharged at mean daily rate of 317,000 L/d in it's first three years of operation. This rose to 428,000 in the last two years of operation. Figure 11 shows the cumulative discharge volume over the period of recorded operation of the two outfalls.

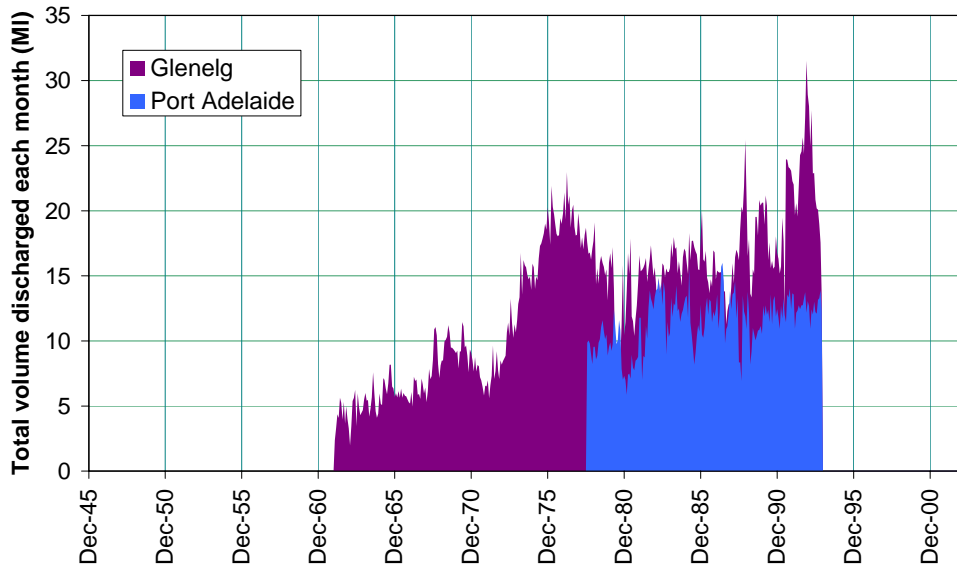


Figure 10. Monthly total volumes of digested sewage sludge discharged from Glenelg and Port Adelaide WWTPs (note data are plotted from zero for each site and overlaid).

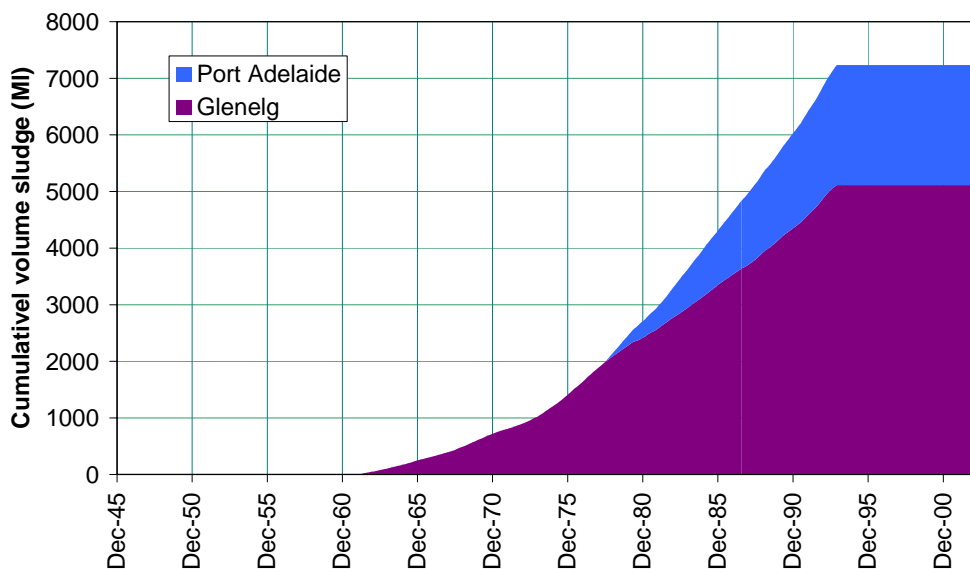


Figure 11. Cumulative total volume of digested sewage sludge discharged from Glenelg and Port Adelaide WWTP (stacked data).

The nature of the pumping regime for each outfall is also worth mentioning. Rather than discharging continuously at a low rate the pipelines were operated to minimise, as far as was achievable, the blocking of light, so night-time operation was favoured. This had the added benefit of minimising any visual impact. As mentioned above, the period midway between low and high tides afforded the greatest mixing and fastest currents, i.e. enabling the greatest dispersion. The other requirement for non-continuous discharge from the pipes was to provide sufficient flow velocity in the pipes to prevent sedimentation. This was more of an issue at Port Adelaide where the sludge had a significant sand component (Thomas, 1981). At Port Adelaide, the pumping rate was limited to 24 L/s to prevent draw-down in the sludge digesters (Thomas, 1981). At this rate, the pipeline would have had to have discharged for approximately 2.5 hours twice a day (on each tidal cycle) to draw-off the mean daily sludge volume of 428 kL/d towards

the end of its operation. It seems unlikely that this would have been possible given the problems encountered at Glenelg. The Glenelg pipeline suffered from the build-up of fibre, particles and struvite crystals (CCE, 1985). The pumping rate in 1968 had been 17 L/s, which would have required around 4.4 hours of pumping to discharge 270 kL/d. By 1974 this rate of flow had been reduced by 40 %, this would have required 13 hours of pumping to clear the 490 kL/d mean annual volume of sludge. The pipeline was acid or air scoured on various occasions. The pumping time in October 1977 had risen to 16 h/d and occasionally 20 or even 22 h/d to clear the necessary volume of sludge to maintain operation of the plant (CCE, 1985). A little over a year, later the pipe became completely blocked and had to be manually inspected and cleared by divers. A program of regular flushing was required to maintain pipeline operation from then on.

4. Suspended Matter

Suspended solids have both direct and indirect impacts on the aquatic environment. Indirectly, suspended material is a major vector for the transport of contaminants in the environment including heavy metals, radionuclides, persistent organic pollutants, microbial contaminants and phosphorus. Directly, suspended matter reduces light transmission and hence photosynthesis. Particulates, once delivered to the coastal zone, may settle-out during relatively quiescent conditions only to be resuspended by more energetic wind/wave action. Settled material coats plant surfaces and when resuspended blocks out light. Along the immediate Adelaide coastal strip, a brown mass of turbid water sits over the coastal seabed for extended periods during the winter months when incident light is already reduced (e.g. Thomas, 1995) (Figure 12). Extended turbid periods are a consequence of the hysteresis effect associated with fine sediment resuspension and settlement, i.e. the degree of sea-bed disturbance energy required to mobilise fine sediments is far greater than that required to maintain them in suspension, this is especially the case with clay particles which are plate-like, thus once redisturbed, sediments remain in suspension for extended period until such time as conditions are sufficiently quiescent to allow deposition. Clay particles typically settle at a rate of 1 m/d in still water.



Figure 12. Turbid waters containing resuspended fine particulates shroud the Adelaide coastal seabed (Photo J. Wilkinson, 14-9-03).

Once fine sediment has entered a system, the cycle of settlement and resuspension may occur continually (Koch, 1999). Townsend (2002) suggested that hydrodynamic disturbance of the seabed is likely to be wave-action dominated, and spring tide currents of 0.28 m/s have been measured around 5 m off-shore. Given that on-shore winds correspond with periods of high turbidity and that the flushing potential of tidal currents is low, the material resuspended off the Adelaide coast may represent the accumulation from many years of input, and as Ralph *et al.* (2003) suggests its impact may be difficult to reverse. Ralph *et al.* (2003) also summarised the literature regarding the impact of periods of extended light reduction on seagrasses. The minimum light requirement for seagrasses varies between 5 to 20 % of surface irradiance and that survival times under these conditions can vary between 20 days and 5 months depending on

the susceptibility of the species. Species with larger rhizomes and hence greater carbon reserves are able to survive the longest.

Figure 13a illustrates the typical annual loadings of suspended solids from the WWTP effluents and sludge outfalls at the end of the period during which the sludge outfalls were still in operation. At that time, the early 1990s, the sludge outfalls accounted for around 62% of the annual solids load from WWTP sources and Bolivar WWTP accounted for 31%. Glenelg and Christies Beach only contributed only 7% of the load. The cessation of sludge discharging in 1993 eliminated a major source of particulate matter to the Adelaide coastline. Upgrading at Bolivar WWTP from trickling filter treatment has also reduced the solids output from that plant (Table 2). Clearly the Bolivar treated wastewater has a far greater suspended load than Glenelg or Christies Beach (Table 2), this is largely a consequence of algal particles derived within the final polishing lagoons. The WWTPs currently discharge around 2000 tonnes a year (Figure 13b) compared to 8400 tonnes in the early 1990s (the latter figure includes sludge discharges). This is a four-fold reduction in the overall WWTP derived load, with the treated wastewater (i.e. non-sludge) load falling to 64 % of the load in the early 1990s. An additional WWTP-derived suspended solids load will be discharged from the Bolivar outfall following completion of the High Salinity WWTP. If the new plant achieves 10 mg/L suspended solids it would deliver a further 150 tonnes each year to the Bolivar discharge.

Table 2. Mean suspended solids concentrations in effluents from Bolivar, Glenelg and Christies Beach WWTPs.

	Bolivar mg/L	Glenelg mg/L	Christies Beach mg/L
1994-2000	90.2	15.6	13.1
2001-2003	36.0 (2003)	11.7	9.3

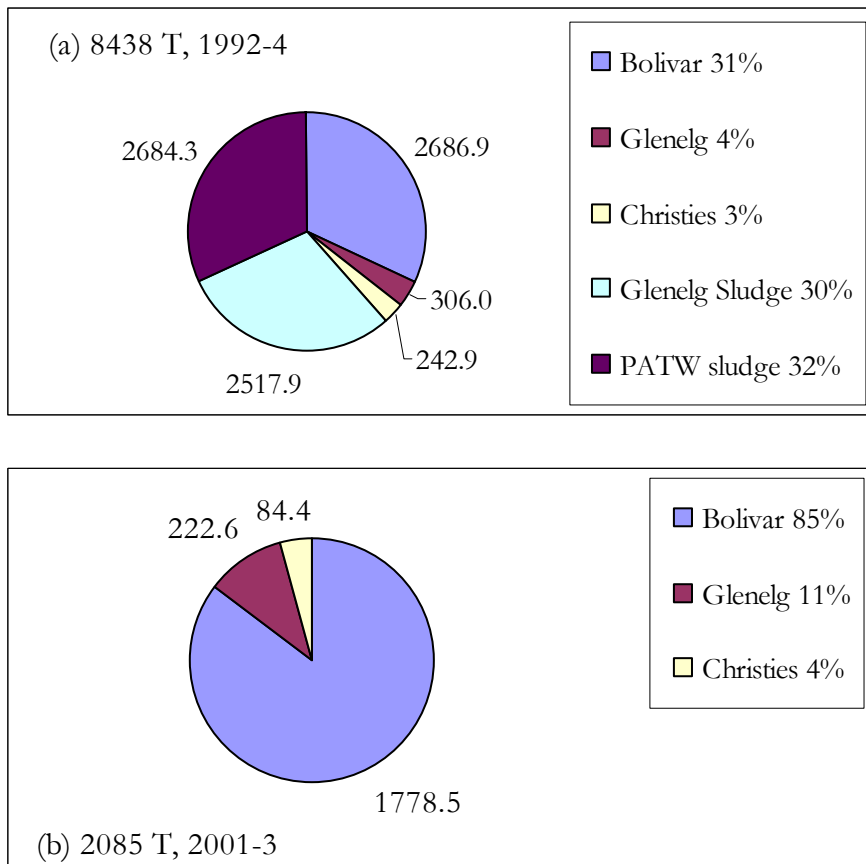


Figure 13. Suspended solids loads from the WWTPs in (a) the early 1990s and (b) after the cessation of digested sludge discharges and upgrading of Bolivar WWTP (2001/3).

5. Nutrients

The nutrient concentrations and loadings to the ACWS study area from Glenelg, Bolivar and Christies Beach WWTPs are presented below along with estimates for the sewage sludge discharges.

Due to the variety of sources of raw data for individual sites the combination of data sets was often problematic and requires further attention to generate reliable historical records of treated wastewater concentrations and loads. This was particularly the case for Glenelg.

Routine monitoring of WWTP effluents for nutrients was not carried-out until the mid 1970s, however, the Gulf St Vincent Water Pollution Studies (Lewis, 1975) did sample and analyse wastewaters for nitrate-N and phosphate-P. The earliest record for nitrogen dates from 1934 when Johnston (1934) reported a concentration of 10.2 grains per gallon as sodium nitrate for the Glenelg WWTP final treated wastewater. This equates to 23.9 mg/L NO₃-N in SI units and is around the 95 %ile total oxidised nitrogen concentration (23 mg/L) of recent data (1975 to 2003) where nitrite only forms a minor proportion of the total.

Data consistency has been an issue when attempting to compare studies and investigate historical loads of nitrogen in particular. Lewis (1975) only reported nitrate-N for monthly samples in 1973 and 1974 (mean concentration 15.8 mg/L). Steffensen (1985) reported total oxides of nitrogen (OxN) and total Kjeldahl nitrogen (which includes ammoniacal nitrogen and organic nitrogen). In his work, the OxN concentrations were twice those reported in the EWS monthly data abstracts (mean concentrations of OxN-N = 20.7 mg/L and 9.9 mg/L for the same period of data), and the latter part of the data reported from July 1980 used the lower EWS abstract values. Similarly, data from the EWS abstracts and monthly values provided by SA Water differed by a factor of nearly two, on this occasion the EWS abstract data has the higher values (July 1994 to July 1996, 19.5 mg/L nitrate-N and 11.9 mg/L, respectively). The Glenelg plant has three final treated wastewaters and a composite of these is generally presented as the “Glenelg Final Treated wastewater”. One possible source of error is that data for different waste streams were being reported, however, nutrient data for each of the three treated wastewater streams differ from each other from month to month in terms of nitrogen speciation, but no one plant discharges consistently more or less nitrogen than the others.

Table 3. Current and historical mean concentrations of nitrogen (mg N/L) and phosphorus (mgP/L) in WWTP effluents.

Nutrient concentration (mg/L)	Bolivar		Glenelg		Christies Beach	
	1976/97	2001/3	1976/97	2002/3	1976/97	2002/3
Total Nitrogen	38.8	11.9	31.0	20.8	27.6	28.0
TKN	34.9 [†] 15.6 n=175	9.25 2.9 n=24	15.6 10.8 n=165	8.99 7.1 n=12	8.94 n.d -	24.6 11.6 n=12
Ammoniacal N	21.7 11.9 n=145	3.14 2.7 n=24	10.7 6.4 n=151	6.87 5.8 n=12	6.10 n.d -	23.3 10.4 n=12
*Oxidised N	3.85 4.8 n=160	2.20 1.9 n=24	15.4 6.9 n=164	11.8 3.4 n=12	20.2 n.d -	3.31 1.71 n=12
Phosphorus	7.36 2.2 n=199	3.70 0.76 n=24	7.82 1.9 n=195	8.12 0.5 n=12	8.25 n.d -	7.88 1.27 n=12

Note: Data for 1976 to 1997 provided by the AWQC. Data for 2002/3 provided by SA Water. *Oxidised N is the sum of NO₃ and NO₂ species. [†]*Italicised values are the standard deviation of the means*

Prior to the Environment Improvement Programs (EIPs) total nitrogen concentrations of the three direct coastal discharging WWTPs were consistently around 30 mgN/L or greater (Table 3). It is important to note that Christies Beach WWTP is still being optimised with ultimate target of 10 mgN/L. The current situation is very much improved at Bolivar and Glenelg as the EIPs take effect, ultimately all plants will discharge around 15 mgN/L from Bolivar and 10 mgN/L from Glenelg. The digested sludge nutrient concentrations were very much higher than those in treated wastewater (Table 4), and despite the relatively low rates of sludge discharge, the actual annual loadings from the sludges amounted to around 15% of the annual load during the last five years of operation (1987-1993) (see Figure 14a). The estimated overall life-time nitrogen and phosphorus loadings from the sewage sludge discharges are greater than 5600 tonnes N, and 1300 tonnes P for Glenelg and 2000 tonnes N and approaching 500 tonnes P from the Port Adelaide WWTP (Table 4). Of the nitrogen discharged from Port Adelaide sludge outfall, less than 0.5% was in the oxidised form, 40% was ammoniacal nitrogen and the remaining approximately 60% was in the organic form.

Table 4. Concentrations (mg/L) and loads of nitrogen and phosphorus discharged to sea from the Port Adelaide and Glenelg digested sewage sludge outfall between 1986 and 1992.

	NH ₃ -N	OxN-N*	TKN	PO ₄ -P	SRP*
Port Adelaide	n=71	n= 12	n=56	n=63	n=47
Mean concentration (mg/L)	397	2.4	981	233	20
Total output (ton)	841	5.1	2080	495	42.4
<i>Mean daily load (kg/d)</i>	<i>150</i>	<i>0.9</i>	<i>372</i>	<i>88.4</i>	<i>7.6</i>
Glenelg	n=66		n=54	n=59	
Mean concentration (mg/L)	540		1100	263	
Total output (ton)	2763		5624	1343	
<i>Mean daily load (kg/d)</i>	<i>238</i>		<i>485</i>	<i>116</i>	

*Data from Neverauskas (1987)

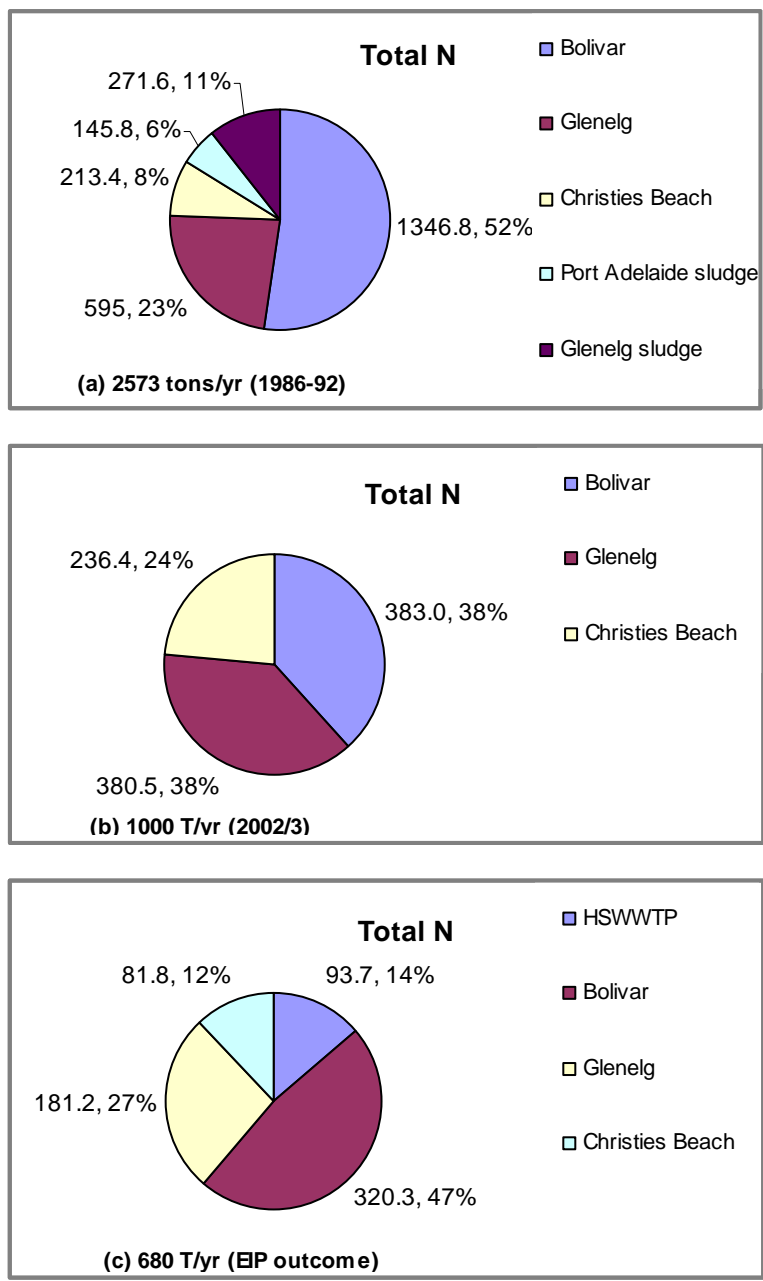


Figure 14. Total nitrogen discharges to sea from (a) the WWTP treated wastewater and sewage sludge discharges (based on flows and concentrations from 1986 to 1992), (b) WWTP treated wastewater only (following the cessation of sludge discharges, using actual data for 2002/3) and (c) indicating changes in load from Christies Beach and Glenelg when nitrogen output is optimised at a mean of 10 mgN/L at 2002/3 flow rates, and Bolivar discharging as for 2002/3 plus additional 9.4 GL at 10 mgN/L from the Bolivar High Salinity WWTP. All flow values are net of re-use.

Figure 14 demonstrates the changes in total annual nitrogen loadings from Christies Beach, Glenelg and Bolivar for three periods. Figure 14 a shows the typical loadings during the period when the sludge outfalls were operating (mean data for 1986 to 1992) resulting in an annual total of around 2700 T/y. In the year 1996/7 the nitrogen loading from Bolivar peaked at 1718 T/y. Nitrogen output from Glenelg and Christies Beach also peaked in the late 1990s. At Christies Beach the annual nitrogen load peaked at approximately 370 T/y in 1996/7, a period when total nitrogen concentrations were in excess of 35 mg/L.

The second period (Figure 14 b) indicates the total nitrogen load in the season 2002/3, ten years after the cessation of sludge disposal to sea and following the upgrade of Bolivar to activated sludge and including the Queensbury Transfer (see Section 2), a total to 1000 T/y. The load from Glenelg was already reduced by this time as a result of the EIP upgrade to IFAS which had reduced the mean nitrogen concentration from around 30 mg/L to 20 mg/L. The nitrogen load from Christies Beach in 2002/3 was greater than in the 1986 to 1992 period (Figure 14 a and b) partly as a consequence of the growth in connections and load into the plant. This load in net of the Willunga Basin Transfer which has reduced the annual discharge volume since 1999.

Figure 14c indicates the likely initial outcome of the EIPs. One of the aims of the EIPs is to achieve complete nitrification, thereby eliminating ammoniacal nitrogen species from treated wastewater. Further denitrification will reduce the nitrogen concentration by gassing-off nitrous oxide to the atmosphere. The ultimate goal is the reduction of total nitrogen concentrations to 10 mgN/L from Christies Beach and Glenelg and 15 mgN/L from Bolivar. Figure 15 demonstrates the changes in nitrogen speciation in the Bolivar treated wastewater since the mid-nineteen-nineties. The ammoniacal nitrogen concentration has been successfully reduced to 2.4 and 3.9 mg/L in 2001/2 and 2002/3. For Bolivar the mean of total nitrogen concentration for 2002/3, of 11 mg/L, this was used in Figure 14 c. The load estimate was based-on the 2002/3 figure minus the reduction in total nitrogen load that is anticipated from an increase in re-use by the Virginia growers (a rise of around 5 GL to 18,000 in 2004). The opening of the high salinity treatment plant at Bolivar (HSWWTP) will increase the total nitrogen load discharged by around 94 T/y (approx. 9.4 GL x 10 mgN/L). The EIP upgrade to IFAS is being optimised, this is apparent in Table 3, when optimised the nitrogen load will reduced by 64% to 82 T/y (assuming 2002/3 flow volumes).

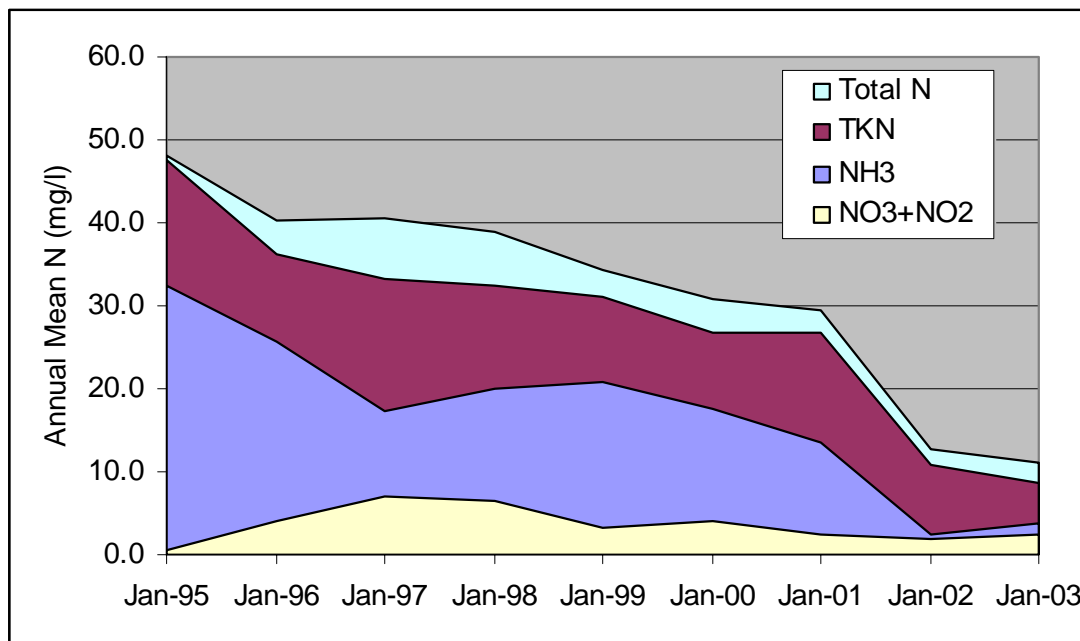


Figure 15. Mean annual concentrations of nitrogen species and total nitrogen in Bolivar final treated wastewater. The plot demonstrates the reductions in nitrogen output achieved since 1995 (each variable is plotted independently from zero.).

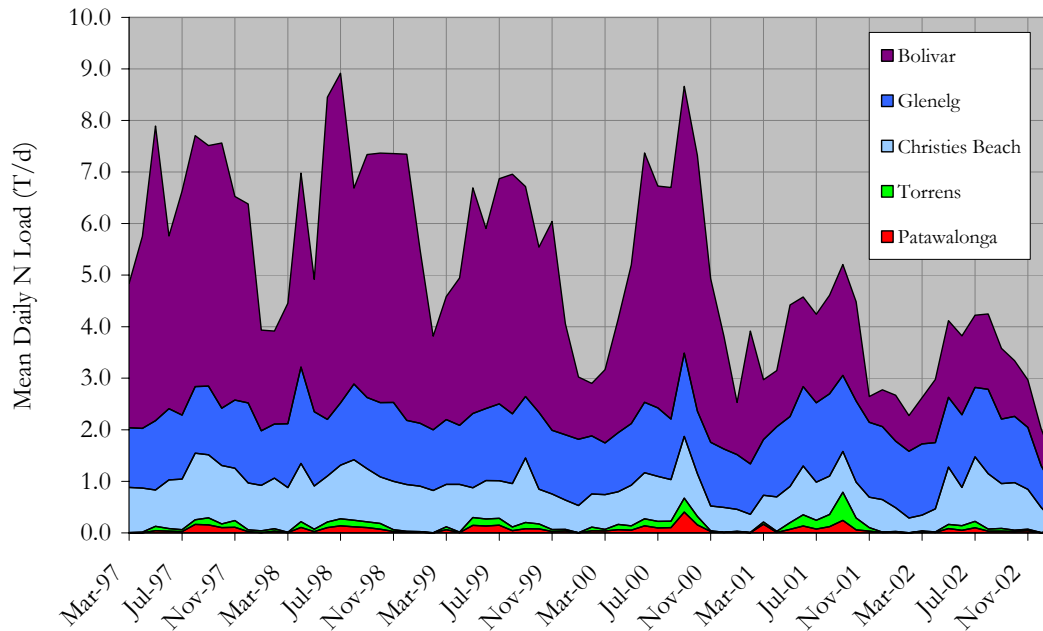


Figure 16. Monthly mean loads of nitrogen discharged to sea (expressed as T/d). The data are stacked to show the total daily load and nitrogen discharges from the Torrens River and the Patawalonga system are included to demonstrate the relative magnitudes of these inputs.

Figure 16 shows the seasonality in nitrogen loadings, highlighting the generally minor riverine component. The seasonality is influenced by higher winter concentrations in treated wastewater, greater winter sewage flows (fed by groundwater ingress), and summer re-use of reclaimed water. Coastal seagrasses receive the highest nutrient loads during the winter when suspended loads from stormwater inputs and turbidity, due to wind/wave resuspension, is greatest.

Figure 17 a to b indicate the changes in phosphorus discharges to the Adelaide coastline in a similar way to Figure 14 above. The cessation of sludge pumping from Port Adelaide and Glenelg removed 100 tonnes a year from coastal inputs. The sludges accounted for 20 % of phosphorus discharged in the early 1990s, the WWTPs delivered 80 % of phosphorus at that time (Figure 17a).

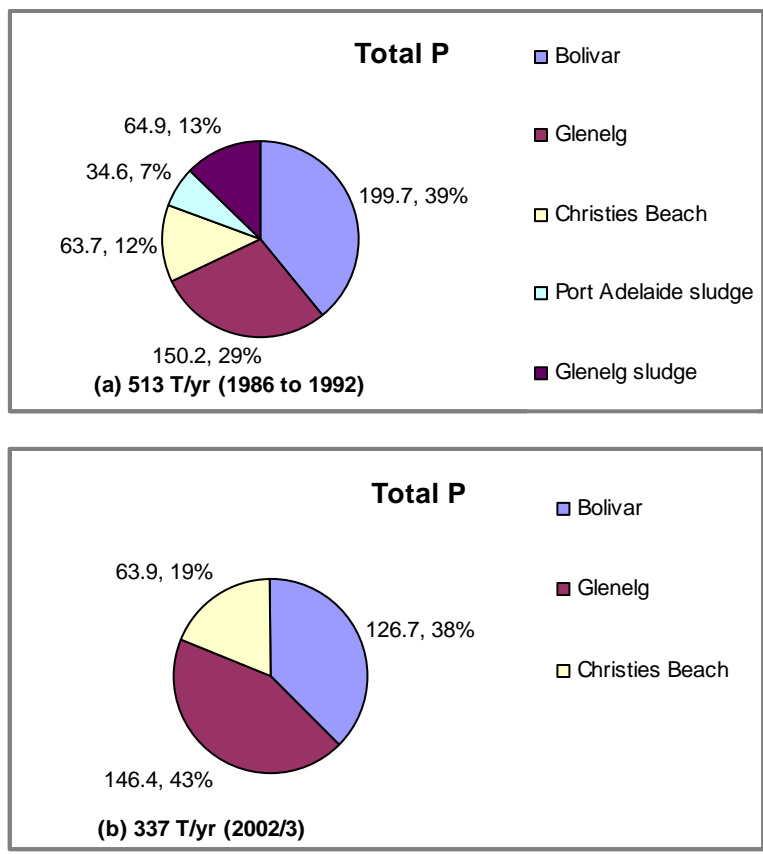


Figure 17. Phosphorus loads from (a) the WWTP effluents and sludges in the early 1990s and (b) the WWTP effluents 2002/3 data.

The reduction in phosphorus load from Bolivar from the early 1990s to 2002/3 is the result of the upgrade of Bolivar. This has reduced the mean phosphorous concentration, from 6.0 mg/L prior to July 2000 to 3.9 mg/L since then. At Christies Beach, P loads peaked in 1998/9 with a total annual load of 87.7 tonnes, the subsequent reduction (Figure 17b) is due to the transfer of treated wastewater into the Willunga Basin. There may be an increase in phosphorus loading to sea from the Bolivar outfall due to the HSWWTP. The phosphorus concentration from the HSWWTP is anticipated to be below that of the main Bolivar treated wastewater. For example if the HSWWTP produced an effluent with 2 mgP/L then the additional P load would be around 20 T/y (this estimate is conjectural since no actual data are available). Phosphorus out of the main Bolivar plant is likely to be reduced by anticipated growth in re-use via the Virginia Pipeline, an increase from approximately 12.75 GL in 2002/3 to 18 GL would reduce the P load by around 16 T/yr. Thus phosphorus loads from Bolivar, Glenelg and Christies Beach are not expected to undergo major changes.

6. Heavy Metals

Metals above calcium in the period table are often termed heavy metals. They are often toxic and are totally non-degradable and as such, tend to accumulate in the environment (Argue, 2003). Both stormwater and WWTP effluents are contributors of heavy metals to the Adelaide coastal zone, although the relative proportions and total annual loadings have changed over time. The data presented and discussed below are for total concentrations of metals, this means that the original samples have been digested in acid to release all solid phase and particle bound metals into solution. The concentrations presented, therefore, indicate a worst case because in the raw treated wastewaters the metals may not be in bio-available forms. The bio-available forms are generally the dissolved component of the metal. In well oxygenated waters many metals tend to bind to iron (rust) particles, to clay minerals and may otherwise be bound-up in organic complexes with biogenic substances.

Metals concentrations here are also considered relative to the ANZECC/ARMCANZ (2000) guide levels for the protection of marine species. Again the analysis presents a worst case. The concentrations are considered in their undiluted form, this is effectively assuming that organisms are exposed to the undiluted treated wastewater. This is an unrealistic condition since there will be dilution of the wastewater as it enters and mixes with the receiving water. From the point of entry into the sea there is a concentration gradient with the highest concentrations at the point of entry reducing to background seawater levels away from that point. This means that any effects would be most likely in the immediate vicinity of the discharge point. In addition, simple observation of metals concentrations in water may not reflect how these metals are taken-up by organisms. Metals concentrations from environmental samples of plant and animal tissues are a better indicator of load impacts because they effectively integrate-out the variations that occur in environmental waters, i.e. they are likely to give an indication of the mean impact of inputs to a given system over a long period. Evaluation of the actual metals concentrations in the inputs to the system is mainly of interest in examining key sources to that system.

In relation to sea-grasses, some background on the impact of metals on sea grasses helps to set the observed metals concentrations in treated wastewaters in context. The potential impact of heavy metals on seagrasses is two-fold, firstly via uptake in to the plants themselves, and secondly (and indirectly) where the inhibition of grazers impacted by the toxic effects of the metals results in denser epiphyte growth. There is little evidence that this potential indirect impact has ever been addressed in any Gulf St Vincent studies, yet Borum (1985) highlights the importance of assessments of herbivore activity when investigating epiphyte communities. The direct effects of heavy metals on seagrasses are also not well understood (Ralph *et al.*, 2003), however, seagrasses are known to take-up heavy metals via the root system from sediment interstitial waters and directly into the leaves and this is believed to be controlled by exposure concentration (see Ralph *et al.* 2003). As mentioned above bio-availability of heavy metals is another important factor and is a function of their solubility, which is affected by a wide variety of environmental factors. The impact of heavy metals on fish is well known with copper, cadmium, nickel, zinc, lead and chromium comprising the key metals in saltwater in decreasing order of toxicity (e.g. Peterson and Batley, 1992). Mance *et al.* (1984) found that fish tended to be less susceptible to many of the heavy metals than invertebrates.

This project has acquired a wide range of metals data for the WWTP effluents and digested sludge discharges in the form of raw data from the AWQC and SA Water, summaries from the EPA and SA Water, from data in tables and appendices in reports by EWS and in contract reports to SA Water. A thorough examination of all of the data is beyond the scope of the current meta-analysis, however, Tables 5 to 8 indicate the relative concentrations and loads of 18 metals and arsenic in the WWTP effluents and sludge. In Tables 5 to 8 all concentrations are of total metals, i.e. from acid digested samples so that all forms of each metal are included in the analysis. The WWTP treated wastewater concentrations were aggregated directly from SA Water summaries for 1994 to 2001 as reported to EPA. Sludge concentrations were derived

from raw data provided by AWQC via SA Water. A further comparison of metals loads relative to storm-water inputs will be presented in a later report.

The data in Tables 5 to 8 are presented in order of mass abundance, showing that iron, boron and aluminium are the most abundant of the metals listed in the Bolivar and Glenelg effluents in 2000/2. At Christies Beach, the iron concentration is much lower than at Bolivar and Glenelg; here there is more zinc than iron. The next group of metals, with concentrations between 10 and 50 µg/L are zinc, copper, nickel, manganese with minor differences between

Table 5. Bolivar WWTP: Past and present mean heavy metal concentrations of treated waste water and annual loads discharged to sea.

	Concentration (µg/L)				Load (kg/y)			
	2000/2	1995/7	1991/3	1978/80	2000/2	1995/7	1991/3	1978/80
Al	534	372	354		18365	14305	13176	
B	422	562	604		14508	21571	22481	
Fe	278	382	252		9550	14655	9380	
Mn	104	183	141	120	3568	7019	5248	2986
Cu	55.5	35.3	19.0	40.0	1909	1355	707	995
Zn	51.0	107	38.0	370	1751	4092	1414	9208
Ni	20.0	21.0	43.0	150	686	806	1600	3733
Mo	17.7	8.3	15.0		608	320	558	
Cr	10.7	24.8	18.0	30.0	368	951	670	747
Sn	7.3	<50.0			250			
Pb	4.2	1.1	3.5	220	145	42.7	130	5475
Se	3.7	<1.0	1.0		128			
Co	3.6	15.1	6.0		123	580	223	
As	3.6	1.9	4.0		123	71.3	149	
Th	1.3	<2.0			44.7			
Ag	1.3	0.8	1.3		44.1	31.2	48.4	
Sb	1.1	<1.0	3.0		37.9		112	
Cd	0.5	0.2	0.4	30.0	16.4	7.7	14.9	747
Hg	0.5	<0.1	0.4		16.3		14.9	
Flow, GL/y					34.37	38.42	37.22	24.89

2000-2 data provided by EPA, 1995-7 data from SA Water and 1991-3 data from AWQC, data for 1978-80 from SA Water (1995).

the three effluents. In general, the metals concentrations of the Christies Beach treated wastewater are less than the other two direct coastal discharging WWTPs which probably reflects the lower intensity of industrial activity in that catchment. Molybdenum is notably higher in the Bolivar treated wastewater than in the other effluents and the reason for this difference is currently not clear. In general the historic data suggest improvements in the metals concentrations of the three effluents with time, although certain metals have increased in concentration in the Bolivar effluent. These changes may be either input or process driven and reference to the sewage input metal concentrations may assist in determining the cause of increases in Al, Cu and Pb from 1995/7 to 2000/2 at Bolivar. Al and Cu have both fallen in concentration at the works inlet. Pb has increased, but by a smaller proportion to the increase in

the effluent. These changes in metal abundance may be due to changes in speciation and hence changes in solubility resulting from the impact of the switch from trickling filter treatment to the more activated sludge process.

The patterns in the loads of metals discharged from each works follow those of concentration (Table 5, 6 and 7). Bolivar discharges the greatest loads of metals because of its volume of flow (current data – 2000/2). Historically this has varied with Glenelg wastewater (sludge discharge excluded) in certain periods discharging more of certain metals than Bolivar (e.g. zinc, Figure 25).

Table 6. Glenelg WWTP: Past and present mean heavy metal concentrations of treated wastewater and annual loads discharged to sea.

	Concentration (µg/L)				Load (kg/y)			
	2000/2	1995/7	1991/3	1978/80	2000/2	1995/7	1991/3	1978/80
B	460	448	705		8305	7418	13269	
Al	191	280	472		3447	4647	8883	
Fe	84.3	155	193		1523	2565	3632	
Zn	80.8	141	256	270	1461	2337	4818	4425
Cu	45.6	48.3	75.0	60.0	824	800	1412	983
Mn	29.2	38.7	32.0	40.0	527	641	602	656
Ni	20.1	42.9	175	210	364	711	3294	3442
Cr	9.9	13.5	68.0	10.0	179	224	1280	164
Sn	7.7	<50.0			140			
Mo	6.4	7.0	10.0		116	116	188	
Co	3.8	13.1	7.0	130	68.8	217	132	
Se	3.8	<1.0	1.0		67.8			
Pb	3.0	3.0	3.0		53.9	50.3	57.1	2130
As	2.3	<1.0	2.0		41.9		37.6	
Ag	1.1	1.0	2.1		20.1	16.4	39.5	
Th	1.0	<2.0			18.1			
Sb	1.0	<1.0	1.0		17.4		18.8	
Cd	0.5	0.3	0.4	60.0	8.8	5.0	7.5	983
Hg	0.5	<0.1	0.5		8.6		9.4	
Flow, GL/y					18.07	16.57	18.82	16.39

2000-2 data provided by EPA, 1995-7 data from SA Water and 1991-3 data from AWQC, data for 1978-80 from SA Water (1995).

The metals concentrations in digested sludge are very much higher than in reclaimed water (Table 8). This is not surprising given that metals tend to attach to the particles settled out in the activated sludge process. Iron, zinc, aluminium, copper and chromium are the major components of the activated sludges, and in general are all higher in the Glenelg sludge. Iron has a greater concentration in the Port Adelaide sludge. Although the volumes of sludge discharged to sea were very much smaller than the treated wastewater flows, the high metals concentrations resulted in the sludges being a major contributor to the overall direct coastal WWTP loadings.

Table 7. Christies Beach WWTP: Past and present mean heavy metal concentrations of treated wastewater and annual loads discharged to sea.

	Concentration ($\mu\text{g/L}$)			Load (kg/y)		
	2000/2	1995/7	1991/3	2000/2	1995/7	1991/3
B	294	331	536	2701	3174	4791
Al	150	273	869	1372	2617	7768
Zn	51.6	90.6	120	474	870	1073
Fe	37.2	74.0	116	341	710	1037
Cu	27.0	35.2	52.0	248	338	465
Mn	23.9	33.5	37.0	219	322	331
Ni	8.2	12.6	31.0	75.0	121.0	277
Sn	8.0	<50.0		73.5		
Cr	6.9	9.0	15.0	63.4	86.5	134
Co	3.6	41.0	7.0	33.2	394	62.6
Se	3.3	<0.1	<0.1	30.3		
As	2.2	<0.1	1.0	20.1		8.9
Mo	2.2	4.0	7.0	19.9	38.4	62.6
Pb	1.6	1.9	1.9	14.8	18.6	17.3
Ag	1.0	0.5	2.2	9.2	4.8	19.7
Sb	0.9	<0.1	1.0	8.4		8.9
Th	0.7	<2.0		6.5		
Hg	0.5	<0.1	0.3	4.4		2.7
Cd	0.5	0.3	0.5	4.4	3.1	4.5
Flow, GL/y				9.18	9.60	8.94

2000-2 data provided by EPA, 1995-7 data from SA Water and 1991-3 data from AWQC, data for 1978-80 from SA Water (1995).

As mentioned in Section 2, the impact of 30 years of sludge discharges from Glenelg was far less than that of 15 years operation at Port Adelaide. This is believed to have been due to the less energetic hydrodynamic environment at the Port Adelaide outfall compared to Glenelg (SA Water, 1995). The movement of water in the region of the Port Adelaide outfall is dominated by tidal flow (Petruševics, 1986), the flood tide current is stronger than the ebb and any net transport northwards is weak. Dispersion in the area is minimal and Neverauskas (1987) suggested that sludge particles could in theory have persisted in the region for “some time”. In addition the regular sludge discharges meant that there was the potential for contaminant levels to build-up over a period of days (Neverauskas, 1988). Steffensen (1981) surveyed seawater metals concentrations at 4 depths (at the surface, and at 1, 5 and 10 m) at 9 locations within the Port Adelaide sludge plume in February 1979. Of the results it was stated that the plume had “little effect” on the adjacent seawater. The results of Steffensen may reflect the settleable nature of particle attached metals, combined with the lack of sensitivity of the analytical methods at that time. It is also important to note that the concentration values presented are for acid digested samples, i.e. they are total concentrations including dissolved, attached and solid phase components.

Table 8. Port Adelaide and Glenelg WTPP digested sewage sludge: Mean concentrations of heavy metals and annual loads discharged to sea at the time the Port Adelaide discharge commenced and towards the time the discharges ceased (note that these are total concentrations, there are no data for the dissolved component).

	Port Adelaide			Glenelg		
	Mean (mg/L)	Mean annual load (kg/y)		Mean (mg/L)	Mean annual load (kg/y)	
		1991/3	1978/80		1991/3	1978/80
Fe	216	32050	25116	113	32600	20813
Al	35	5237	4104	46.8	13499	8618
Zn	31	4608	3611	39.8	11483	7331
Cu	19	2841	2226	27.0	7785	4970
Cr	7.53	1117	875	17.2	4965	3170
Pb	3.03	449	352	5.67	1106	706
B	2.09	310	243	3.83	369	236
Ni	1.35	200	157	2.32	1637	1045
Mn	1.29	191	150	1.82	526	336
Ag	0.93	138	108	1.28	668	427
Cd	0.76	113	88.4	0.20	58.1	37.1
Mo	0.33	48.2	37.8	0.16	45.4	29.0
Sb	0.07	9.7	7.6	0.06	17.7	11.3
As	0.05	7.8	6.1	0.04	11.2	7.1
Hg	0.04	5.4	4.2	0.04	12.5	8.0
Sludge						
flow		148.1	116.1		288.7	184.3
ML/y						

The following text introduces certain metals of significant toxicity in the marine environment and compares the relative loadings from the treated wastewaters and digested sludge discharged to the Gulf over time.

6.1 Silver (Ag)

Of the heavy metals silver, in the free silver Ag⁺ form, is one of the most toxic metals to marine and freshwater fish (Hogstrand, 1998). Warrington (1993) reported phytoplankton toxicity of silver, which decreased with increasing salinity as free silver became increasingly complexed with chloride ion. Dinoflagellates were the most sensitive group to silver, Diatoms were least sensitive. The ANZECC/ARMCANZ (2000) trigger values for silver in marine waters is 0.8 µg/L for the protection of 99 % of species and 1.4 µg/L for the protection of 95 % of species. Current data for the Metro Adelaide wastewaters show that they have concentrations around these values (Tables 5 to 7). The digested sludges, however, were 1000 times more concentrated in silver than effluent. The 80 % species protection trigger value for silver in marine waters is 36 µg/L. The sludge plumes would have to have been diluted by a factor of around 30 to achieve the 80 % protection value and around 1000 to achieve the 99 % protection value. Figure 18 shows the contribution of the sewage sludges to the silver load to the Adelaide coastline, from Glenelg in particular, approaching 700 kg/y in 1991/3. It is uncertain to what degree this would be bio-available, however, direct ingestion of particles is possible. That the sludges contributed such high loads of silver indicates the efficiency of the activated sludge process in removing certain metals from wastewater, effectively eliminating those metals from the nearshore treated wastewater discharge and then pumping them out further offshore from the sludge outfall. The cessation of sludge discharges has dramatically reduced the WWTP loading of silver to the Adelaide coastline.

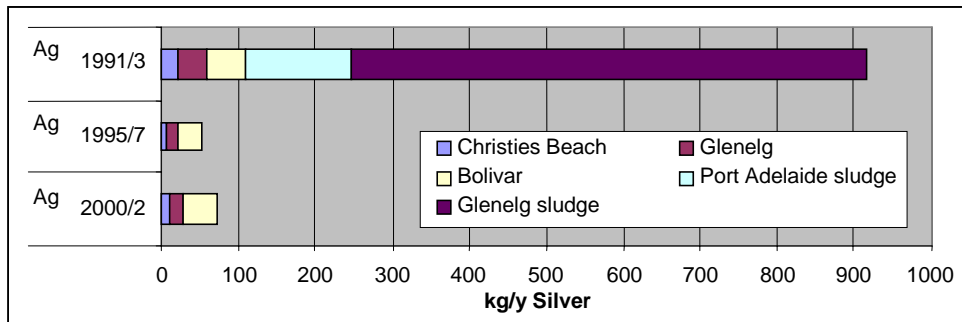


Figure 18. WWTP direct coastal discharged loads of silver.

6.2 Arsenic (As)

Arsenic is often listed with heavy metals, but is in fact a metalloid and its behaviour in the environment has similarities to those of the heavy metals. It is often associated with other metals such as copper, iron and nickel. Arsenic compounds, together with copper and chromium, are used in wood preservatives and arsenic (III) oxide is a by-product of lead and copper smelting (UKMSAC, 2003). The annual load of arsenic to the coastline is around 200 kg (Figure 19), however, it is only present in Adelaide wastewaters in the concentrations that might be expected in ambient seawater (Mance *et al.*, 1984). The ANZECC/ARMCANZ (2000) marine water quality guideline state that there is insufficient data to provide trigger values for arsenic. The ANZECC/ARMCANZ (1992) guide level for arsenic in marine waters was 50 µg/L. Arsenic from the wastewaters when diluted into seawater are unlikely to pose a threat to marine life. As for the sludge discharges, they only represented a small proportion of the overall load in 1991/3. The concentrations of arsenic in the sludges were 20 times those in wastewater and were around the ANZECC/ARMCANZ (1992) guide level concentration of 50 µg/L. Smith and Edwards (1992) highlighted the sensitivity of certain algal species to arsenic concentrations in seawater of around 50 µg/L. Arsenic is also known to bioaccumulate in bivalve molluscs, the flatworm *Planaria* and various algae, however a large proportion of the arsenic may be in the arsenobetaine form which Smith and Edwards (1992) suggest is of lower toxicity. Biomagnification of arsenic up the food chain does not occur (UKSMAC, 2003).

Given that dilution by seawater would generally have rapidly reduced the arsenic concentration around the outfall, and that the sludge discharges ceased 10 years ago, arsenic from wastewater does not appear to present a problem along the Adelaide coastline.

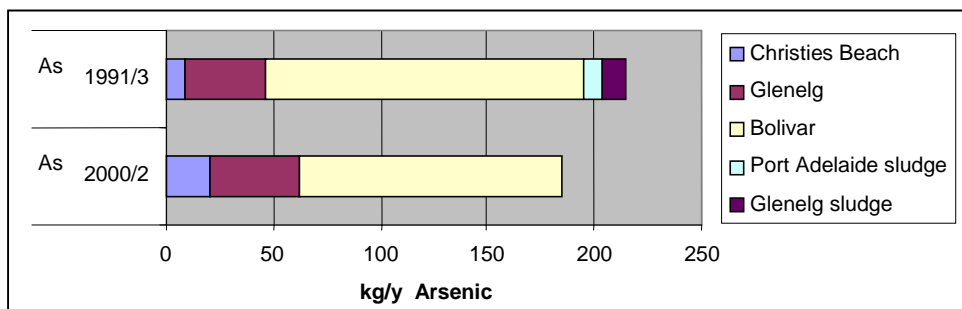


Figure 19. WWTP direct coastal discharged loads of arsenic.

6.3 Copper (Cu)

Copper in natural waters occurs as the free copper (ii) ion or in complexes with dissolved organic matter and adsorbed to organic matter, or suspended precipitate particles. Copper is removed from the water column by particulate scavenging and settlement (UKMSAC, 2003). Sampling of Adelaide coastal seawaters following wind-wave resuspension of sediments, might be expected to show far higher metal concentrations than would be measured during calm conditions when particulate matter has settled back out of the water column. Copper is readily accumulated in plants and animals, but is not biomagnified in the food chain (CCREM, 1987). Copper from the Metropolitan Adelaide direct coastal discharges has similar relative distribution in treated wastewater and sludge as silver (Figure 20), however the load discharged is more than ten times that of silver. The ANZECC/ARMCANZ (2000) trigger values for copper in marine waters are 0.3, 1.3, 3 and 8 µg/L (for 99, 95, 90 and 80 % species protection, respectively). The Metro Adelaide effluents have between 20 to 40 times the 95 % protection value and at least 3.5 times the 80 % protection value (Tables 5 to 7), yet put in context River Murray water at Murray Bridge had 4 times the 80% trigger value for freshwater (SA Water, 1995). The digested sludges had between 2000 (Glenelg) and 1500 (Port Adelaide) times the 95 % protection level and 336 and 240 times the 80 % protection value. Given that the majority of the copper would be sediment associated, settlement into bottom sediments would be the likely fate of this material – especially at the Port Adelaide discharge point.

The ANZECC/ARMCANZ (2000) Interim Sediment Quality Guideline (ISQG) is 65 mg/kg, the Canadian interim guideline level for copper in marine sediments is 18.7 mg/kg (CCREM, 1987). Port Adelaide dried sludge has copper concentration in excess of 1000 mg/kg, suggesting the potential to accumulate to toxic levels over a period of time once settled-out of the water column. At the Glenelg sludge outfall, the mean sediment copper concentration was >15 mg/kg, within a maximum limit of 100 m from the outfall, in March 1973 (SADEC, 1975).

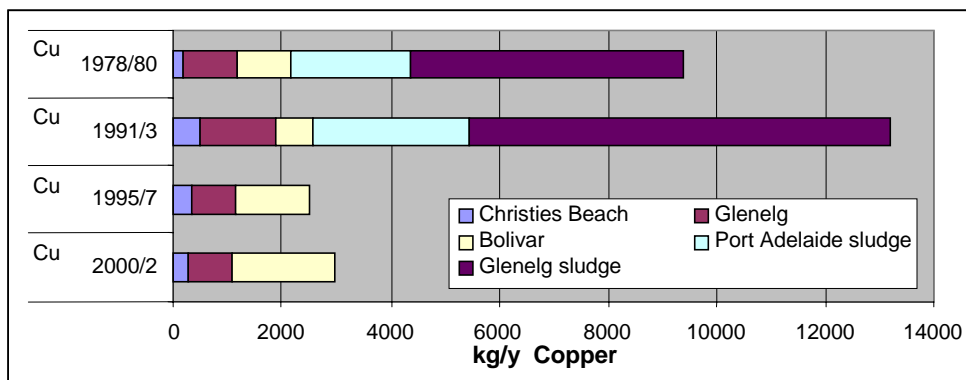


Figure 20. WWTP direct coastal discharged loads of copper.

6.4 Cadmium (Cd)

Cadmium availability is dependent on speciation, certain species being insoluble (sulphides, carbonate and oxides), whereas sulphates, nitrate and halides are soluble. Cadmium is also strongly adsorbed to sediments (UKMSAC, 2003). Uptake of cadmium is influenced by the presence of zinc and calcium, and lethal effects at 16µg/L to marine organisms have been noted (WHO, 1992). The toxicity of cadmium to invertebrates is reduced as salinity increases. Thus the dilution of treated wastewater with sea-water will reduce cadmium toxicity. Figure 21 shows that cadmium discharges in 1978/80 (data source SA Water, 1995) were far greater than in the last ten years. This reduction in concentrations is a likely consequence of efforts to improve industrial effluent quality rather than errors in analytical methods.

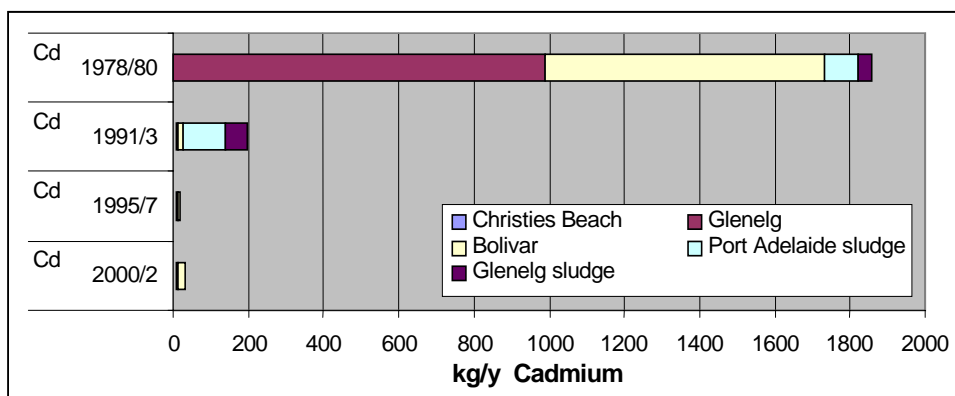


Figure 21. WWTP direct coastal discharged loads of cadmium.

The ANZECC/ARMCANZ (2000) trigger levels for the protection of 99, 95, 90 and 80 % of species in marine waters are 0.7, 5.5, 14 and 36 $\mu\text{gCd/L}$, the three direct discharging wastewater effluents have concentrations of 0.5 $\mu\text{g/L}$, suggesting that the treated wastewaters do not present a hazard to marine organisms. The digested sludges had mean cadmium concentrations of 762 $\mu\text{g/L}$ (PA) and 201 $\mu\text{g/L}$ (G), which are approximately 1000 and 300 times the 99 % trigger value and 21 and 5.5 times the 80 % trigger value for cadmium.

6.5 Chromium (Cr)

Chromium can exist in hexavalent, Cr (VI), and trivalent, Cr (III) forms. The hexavalent and most toxic form is predominant in aerated waters (Argue, 2003). In polluted waters lowered oxygen status favours reduction of to Cr (III) forms which is readily adsorbed to particles and plants, and is not rapidly desorbed (Florence *et al.*, 1980). In sewage, evidence is that most is Cr(III); in treated wastewater the same is likely to be true. Since the oxidation of Cr(III) is kinetically very slow it is likely that most chromium is preferentially removed by the activated sludge process and any remaining Cr(III) would be in the colloidal form (Batley, pers comm.). Cr (III) solubility is influenced by salinity and whether or not the element becomes bound in organic complexes, whereas the oxidised Cr (VI) form is very soluble and not strongly adsorbed to particles (Mance *et al.*, 1984). Fish larvae are particularly susceptible to chromium. Once again the sludges were major contributors of chromium to the Adelaide coastal zone, their elimination significantly reducing the WWTP burden (Figure 22). The current mean annual chromium concentrations of the Adelaide WWTP effluents are between 7 and 11 $\mu\text{g/L}$. This is around the 99 % species protection trigger value of 7.7 $\mu\text{g/L}$ for Cr (III) in the ANZECC/ARMCANZ (2000) water quality guidelines. The trigger values for the more toxic Cr (VI) are significantly lower than for Cr (III), reflecting the greater toxicity to marine life of Cr (VI). The 95 % species protection trigger value for Cr (VI) is only 4.4 $\mu\text{g/L}$.

The WWTPs treated wastewater total Cr concentrations are 50 times more concentrated than the 99 % trigger value for Cr (VI). The significance of this is uncertain since it is not known what the speciation of the dissolved components of that total chromium concentration are.

The Port Adelaide and Glenelg digested sludges had mean total chromium concentrations of 7.54 mg/L and 17.2 mg/L, around 1000 and 2000 times the 99 % Cr (III) trigger value and 80 and 190 times the 80 % Cr (III) trigger value, respectively. The sludges were around 54,000 and 120,000 times the 99 % trigger value for Cr (VI) and 200 and 90 times the 80 % protection trigger values. Mance *et al.* (1984) indicated that fish are less sensitive to chromium than invertebrates; however, fish larvae are particularly sensitive to chromium. It is worth highlighting that the data presented here are for acid digested samples and that the values are therefore taken to indicate the total mass of each metal in the volume of sample.

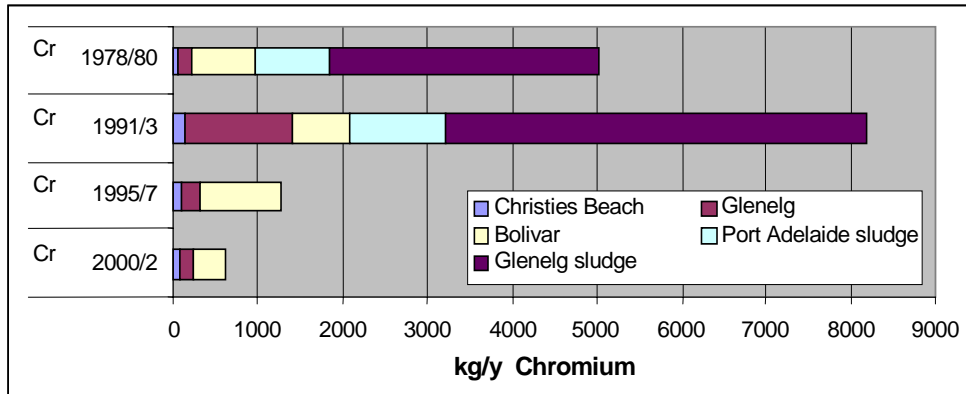


Figure 22. WWTP direct coastal discharged loads of chromium.

Despite the high Cr concentrations in the sludges, SADEC (1975) did not indicate significant accumulation of chromium at the Glenelg sludge outfall in 1973. Its concentration in and removal by the sludge process, but subsequent lack of appearance in sediments around the Glenelg outfall is puzzling.

6.6 Nickel (Ni)

Nickel is reported as being toxic to algae and molluscs (Hunt and Hedgcock, 1992). Nickel tends to occur as soluble salts adsorbed on mineral and organic particles. Treated wastewater nickel was the major WWTP input in the late 1970s and early 1990s, although the Glenelg sludge did discharge a significant proportion of the load. Current nickel discharges are still in excess of 1000 kg/y.

Nickel from Bolivar and Glenelg is in concentrations greater than the 99 % protection trigger value for marine waters of 7 µg/L, but well below the 95 % value of 70 µg/L ANZECC/ARMCANZ (2000). Of the digested sludges, Port Adelaide had a mean nickel concentration of around 200 times the 99 % protection value and 2.4 times the 80 % trigger value of 560 µg/L. Glenelg had greater than 300 times the 99 % trigger value and 4 times the 80 % trigger value. Figure 23 demonstrates the dramatic reduction in nickel loadings from wastewaters since the early 1990s.

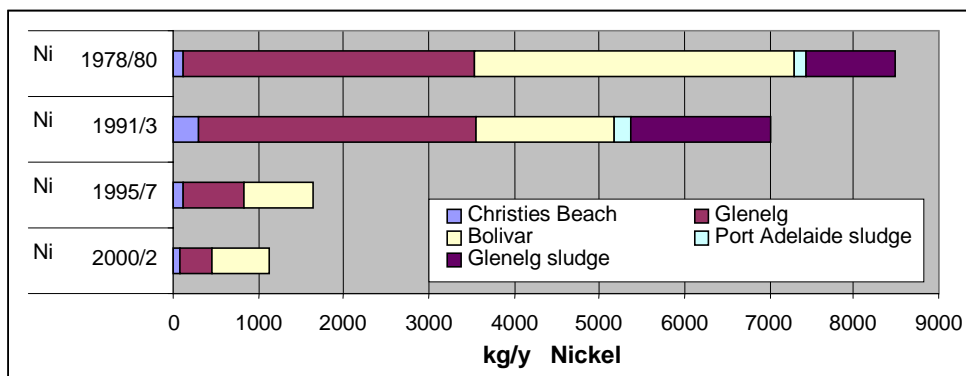


Figure 23. WWTP direct coastal discharged loads of nickel.

6.7 Lead (Pb)

Lead is generally poorly soluble in environmental waters, although nitrate and chlorides of lead are soluble. In marine environments most lead tends to bind to particulates thus the actual bioavailable component may be minimal. Current data demonstrate that WWTP lead concentrations and loadings to the Adelaide coastal zone have been dramatically reduced (Tables 5 to 7 and Figure 24), and that the mean concentrations of the effluents are below the ANZECC/ARMCANZ (2000) 95% species protection trigger level of 4.4 µg/L suggesting that WWTP discharges currently present a low hazard to marine life.

During the operation of the sludge outfalls (1963 to 1993) there may have been some impact from this source. The total lead concentrations of the Port Adelaide and Glenelg sludges were 3030 µg/L and 5670 µg/L, respectively (Table 8). These were 1400 and 2500 times the 99 % trigger level and 250 and 470 times the 80 % protection level. This indicates that these discharges would have remained harmful to marine organisms for some distance around the outfalls and would have required significant dispersion and dilution. SADEC (1975) demonstrated that the lead levels in sediments around the Glenelg sludge outfall of greater than 50 mg/kg within a 50 m distance from the outfall. The ANZECC/ARMCANZ (2000) interim sediment quality guideline (ISQG) trigger value for lead is 50 mg/kg. The Canadian interim level for lead in marine sediments is 30.2 mg/kg (CCREM, 1987). Thus any impact of lead in sediment would be expected to be localised.

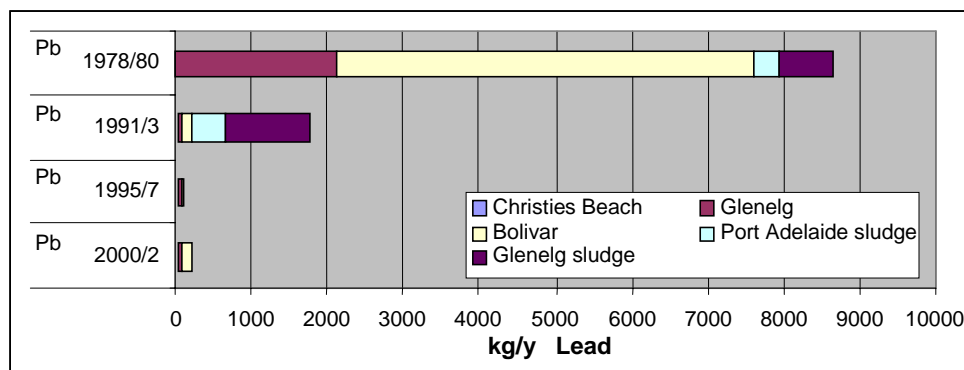


Figure 24. WWTP direct coastal discharged loads of lead.

6.8 Zinc (Zn)

Zinc is an essential trace element for marine life, but is toxic at higher levels. Zinc is highly mobile but tends to attach to particles in estuarine environments (Mance and Yates, 1984). In seawater, the majority of zinc tends to be in solution associated with organic and inorganic complexes. As one of the more mobile heavy metals, it is not surprising that it did not accumulate to a significant degree in sediments around the Glenelg sludge outfall (data from SADEC, 1975). The current mean annual levels of zinc in the WWTP effluents are around 50 µg/L from Bolivar and Christies Beach and 80 µg/L from Glenelg. These values are in excess of the 80 % species protection trigger value of 43 µg/L for zinc in marine waters (ANZECC/ARMCANZ, 2000).

The mean digested sludge concentrations of zinc were 31.1 mgZn/L at Port Adelaide and 39.8 mg/L Glenelg, greater than 4400 and 5700 times the 99 % protection trigger value of 7 µg/L and 700 and 1000 times the 80 % protection value of 43 µg/L for marine organisms. As for copper, significant dilution of the sludges would have been required to render them harmless. Figure 25 presents WWTP loadings of zinc to the Adelaide coastal zone. The sludges were a major source of zinc and their elimination has significantly reduced the zinc burden from WWTP sources.

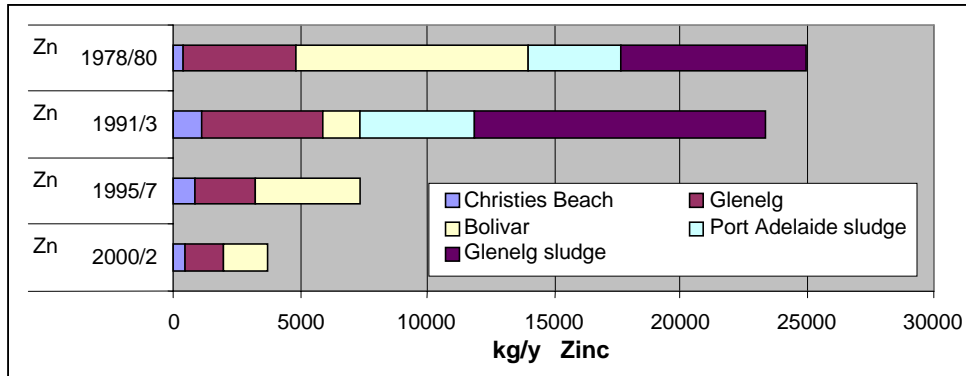


Figure 25. WWTP direct coastal discharged loads of zinc.

6.9 Boron (B)

Boron is a highly mobile metal. The predominant form of boron in seawater is boric acid (76%), while the borate ion accounts for 13% (CCREM, 1987). The high concentrations in reclaimed waters probably reflect the influence of clothes washing products which contain sodium perborate, as well as, cleaning agent borax. Boron toxicity of marine water is not well documented, although the European Union Quality Standard for boron in seawater is 7 µg/L. ANZECC/ARMCANZ (2000) considered that there are insufficient data to determine trigger levels in marine waters for boron.

6.10 Improvements and Targets

Table 9 summarises the changes in total WWTP direct coastal discharges of heavy metals since the cessation of sludge discharges. For the more particle associated metals, i.e. those best removed by the activated sludge process, the reduction in annual load varies between 77 and 92.5 %. As will be seen in a later report, the reduction in WWTP metal concentrations has dramatically increased the relative significance of stormwater inputs, especially for lead and zinc.

Figure 26 provides a visual summary of annual loads of heavy metals to the Adelaide coastline. The components from each source are stacked indicating the relative contribution from each source, clearly copper and zinc are the two main toxic heavy metal loadings.

Figure 27 takes these data a step further, by providing a comparison of the mean annual metal concentrations of each treated wastewater for 2000/2 with the ANZECC/ARMCANZ (2000) trigger value for the protection of species in marine waters. The data are represented without accounting for dilution and as such represent a worst case. The purpose, however, of this plot is to indicate where general metal concentrations fall with respect to the ANZECC/ARMCANZ trigger value guidelines. The metals concentrations of the Christies Beach WWTP treated wastewater are in general lower than for Bolivar and Glenelg. Copper, zinc, cobalt and chromium are present in each treated wastewater at concentrations greater than the ANZECC/ARMCANZ (2000) trigger value for the protection of 95 % of marine species. Cobalt, zinc and chromium are only between 1.5 and 3 times the trigger values, whereas copper is between 20 and 42 more concentrated in the effluents than the 95 % trigger value for marine waters. So, although the total zinc load is somewhat greater than that of copper, the copper concentrations are of more concern from an environmental point of view.

Table 9. Total loads of metals discharged to sea from the WWTP digested sludges and effluents and the percentage reduction in load.

	2000/2* (kg/y)	1991/3† (kg/y)	% reduction
B	25500	41200	38.1
Al	23200	48600	52.3
Fe	11400	78700	85.5
Mn	4320	6900	37.4
Zn	3690	23400	84.2
Cu	2980	13200	77.4
Ni	1130	7010	83.9
Mo	744	903	17.5
Cr	610	8170	92.5
Sn	463		
Se	226		
Co	225		
Pb	214	1759	87.8
As	185	214	14.0
Ag	73.4	914	92.0
Th	69.2		
Sb	63.7	167	61.8
Cd	29.5	198	85.1
Hg	29.3	44.8	34.7
Flow (GL/y)	61.6	65.4	

*Derived from data provided by Environment Protection Authority. †Derived from AWQC data.

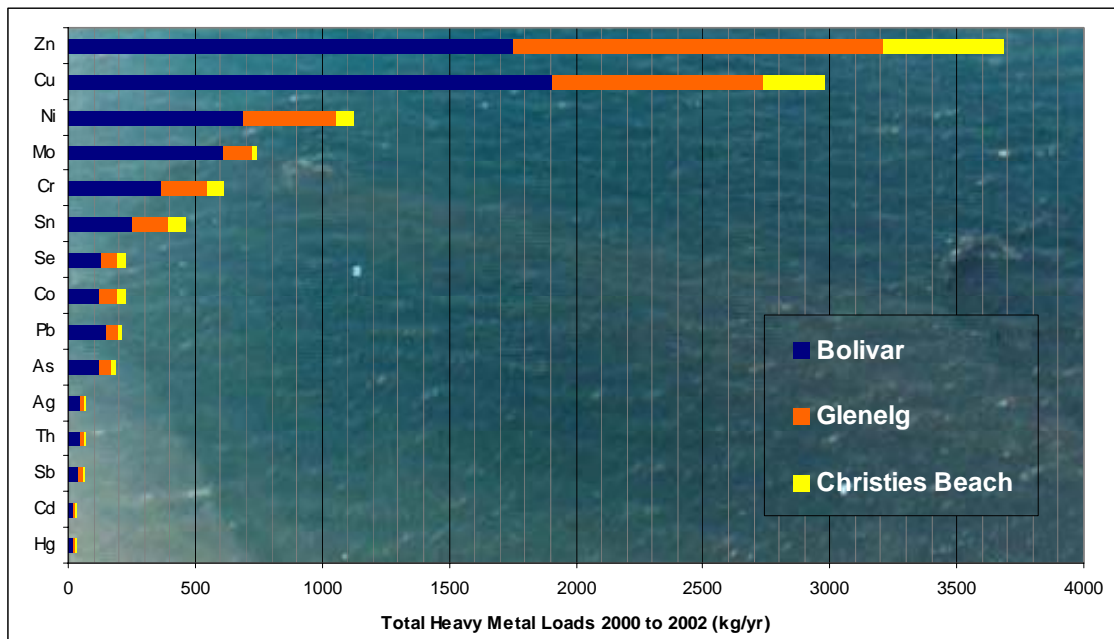


Figure 26. Total loads of heavy metals in 2000/2.

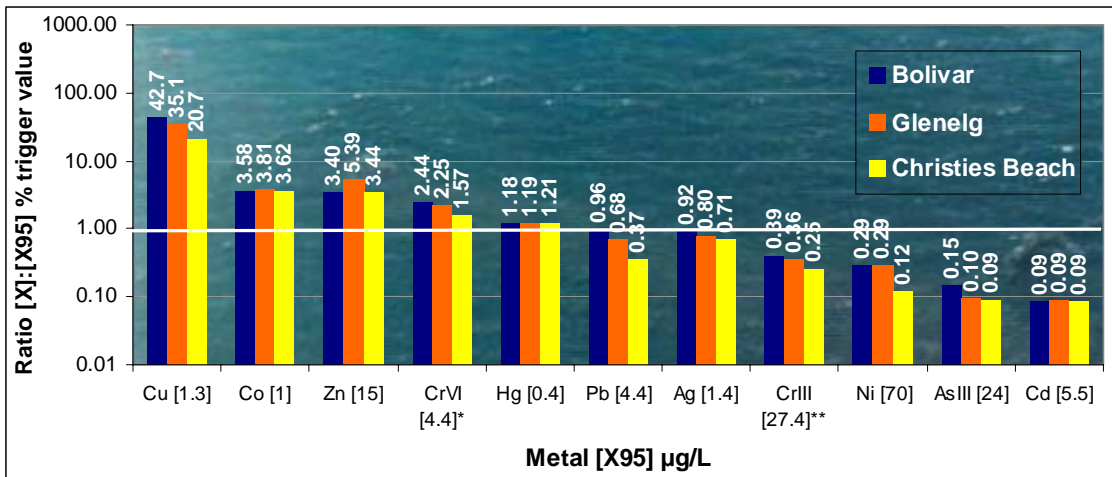


Figure 27. Comparison of treated wastewater metals concentrations and guide levels for the protection of marine species. (Expressed as ratios of the mean annual concentrations [X] in 2000/2 to the trigger level for the protection of 95 % of species in marine waters [X₉₅]. These figures present the worst case as the dilution and dispersion in the receiving water is not represented. Note that the predominant form of chromium is likely to be CrIII, thus the figures for CrVI would be a worst case assuming all chromium were in this form).

7. Organic Contaminants

This section summarises information on organic contaminants in WWTP effluents. The organic contaminants include pesticides, volatile chlorinated hydrocarbons (VCHs), polycyclic aromatic hydrocarbons (PAHs) and persistent organic pollutants (POPs). PAHs, PCBs, dioxins, phthalates, organochlorine pesticides (e.g. aldrin and dieldrin) and alkyl phenolic substances have estrogenic effects, i.e. they mimic female growth hormones and may impact on seagrass beds (SA Water, 1995). Potential direct impacts include disturbances of plant growth regulators (PGRs). Although little is known about PGRs in seagrasses (Muñoz, 1995), there may be an indirect effect from these chemicals via the death of epiphyte grazers.

A major summary of organics in the treated wastewater of Adelaide Metropolitan WWTPs includes data from 1992 to 1994 (Appendix III) and is reported in SA Water (1995). The study found that rarely more than one of the pesticides tested for was detected in each sampling and detections for any individual pesticide were random and infrequently detected. Similarly detections of VCHs were not frequent.

Later, analyses have been performed for a wider range of chemicals. The full listing of analytes is given in Appendix IV. Table 10 summarises the results of these analyses for 1998, 1999 and 2000. The results for each annual sample imply infrequent detection. The limits of detection for each analyte are also listed in Appendix IV. The limits of detection are generally within the range 0.5 to 2 µg/L.

Table 10. Summary of the frequency of detection of organic chemicals in Adelaide metropolitan WWTP treated wastewater in 1998, 1999 and 2000 (figures in brackets show the number of positive results and the percentage of tests positive).

	1998-1999			1999-2000			2000-2001		
	Glenelg	Christies Beach	Bolivar	Glenelg	Christies Beach	Bolivar	Glenelg	Christies Beach	Bolivar
Phenols	12	12	12	12	12	12	12	12	12
Polynuclear Aromatics	20	20	20	20	20	20	20	20	20
Phthalate Esters							6	6	6
Nitrosamines							9	9	9
Nitroaromatics & Cyclic Ketones							17	17	17
Haloethers							4	4	4
Chlorinated Hydrocarbons	10	10	10	10	10	10	10	10	10
Anilines & Benzidines							8	8	8
Monocyclic Aromatic Hydrocarbons							14	14	14
Oxygenated Hydrocarbons							4	4	4
Sulfonate Compounds							1	1	1
Fumigants							5	5	5
Halogenated Aliphatic Hydrocarbons	6	6	6				28	28	28
Halogenated Aromatic Hydrocarbons	9	9	9	9	9	9	9	9	9
Trihalomethanes	4	4	4				4	4	4
Organochlorine Pesticides	(1, 25%) 20	20	20	20	20	20	(3, 5%) 20	(3, 75%) 20	20
Organophosphorus Pesticides	20	20	20	20	20	20	20	20	20
Phenoxy Acid Herbicides	13 (1, 8%)	13	13	13	13	13	13	13	13
Total Compounds Tested	114 (2, 1.75%)	114	114	104	104	104	204 (3, 1%)	204 (3, 1%)	204

Given their infrequent recovery in Adelaide Metropolitan wastewaters, it is unlikely that there is a significant contributing effect from these chemicals, relative to known impacts of nutrients and suspended solids, on the overall patterns of seagrass degradation.

8. Summary, comments, conclusions and suggestions

The ACWS has collated a wide range of historical and contemporary data on treated wastewater and digested sludge composition and volumes discharged to sea. Paper records have been entered and charts digitised and these data processed to recreate the full record of flows direct to the coastal zone. Currently the WWTPs are the best represented component of the inputs to the Adelaide coastal zone and these data represent a valuable resource and will significantly contribute to the possibility of running model hindcasts of seagrass impact scenarios.

Archival literature has provided a history of the Metropolitan Adelaide WWTP developments and more recent information indicates the current direction of wastewater treatment and disposal with the EIP improvements in quality and water reuse and aquifer storage and recharge (ASR) proposals.

The WWTPs contribute around 43 % of the water discharged annually from land-based sources for the period April 2001 to April 2003. Winter flows account for 60 % of wastewater discharges while summer flows account for 40 %. More than a quarter of the total approximately 2000 gegalitres of wastewater discharged directly into the Adelaide coastal zone since 1945 has come from Bolivar in the last 20 years.

The cessation of digested sludge discharge to sea in 1993 dramatically reduced the impact of the WWTPs on the Adelaide coast. Further improvements in treatment have reduced suspended solids from around 8500 T/y in the 1980s to 2000 T/y (current data).

The differences seagrass decline between the regions impacted by the Glenelg and Port Adelaide sludge outfalls differed greatly. This was due mainly to differences in the hydrodynamic environments in both locations. The Glenelg site was subject to stronger currents and thus mixing and dispersion of the sludge plume compared to the PAWWTP site off Point Malcolm.

Total nitrogen loads have been reduced from approximately 2500 T/y to 1000 T/y currently (2003), further EIP improvement may reduce this to around 700 T/y. Ammoniacal nitrogen should have been eliminated from the wastewater stream from all WWTPs by late 2004.

Heavy metal concentrations are much lower than in the 1970s, and loads have been reduced dramatically since the early 1990s with reductions in the total wastewater load of between 75 and 95 % for chromium, lead, nickel, iron, zinc, silver and cadmium. Loads of boron, aluminium, manganese and mercury have fallen between 34 and 53 %, molybdenum and arsenic between 18 and 14 %. The heavy metals with the greatest loads are zinc and copper. The mean treated wastewater concentrations of copper, cobalt, zinc and chromium from Bolivar, Glenelg and Christies Beach are all in excess of the ANZECC/ARMCANZ (2000) trigger level for the protection of 95 % of marine life. Of these metals, copper is between 20 and 43 times the trigger value and cobalt, zinc and chromium are only between 2 and 4 times the trigger value.

Organic contaminants such as pesticides, PCBs, PAHs, POPs etc. have only been detected sporadically. This lack of detection suggests that these substances, if present at all, are present at very low levels and are of low significance when set against the other bulk inputs of substances known to have a major impact on the system.

Given that wastewater inputs have excellent data provision relative to the other sources, project effort in IS1 needs to be concentrated on quantifying the aspects of the other inputs that are less well represented.

8.1 Studies of seagrass loss

Previous studies have lumped all seagrass loss episodes together when investigating the relationship between land based discharges and seagrass loss. The authors suggest that due to the differing nature of the degradation mechanisms and sources of contamination that certain episodes of seagrass loss should be considered in isolation. Namely, that the sewage sludge associated seagrass loss is anomalous to the general pattern of decline along the beach face zone. This area of loss should be subtracted from the total when investigating the impact of WWTP treated wastewater and stormwater discharges.

Borum (1985) indicated the importance of grazers in the ecosystem interactions associated with epiphyte growth. If contaminants in waste and stormwaters impact on grazers, this will exacerbate any affect from epiphyte growth. Previous investigations of seagrass decline along the Adelaide coast have made no reference to grazer populations. Has there been a loss of grazer populations and if so what impact might this have had on epiphyte build-up?

8.2 Comment on sediment resuspension

Previous studies have demonstrated the impact of light loss due to turbidity on the self-maintenance capacity of seagrass rhizomes. Adelaide coastal waters become turbid in response to wind wave activity, in the absence of recent storm or waste water inputs of suspended matter. It has been suggested that the weak coastal circulation is unlikely to disperse the fine material that is resuspended. To what extent is this material (already stored in the near shore zone) responsible for the weakening and decline of seagrass?

References

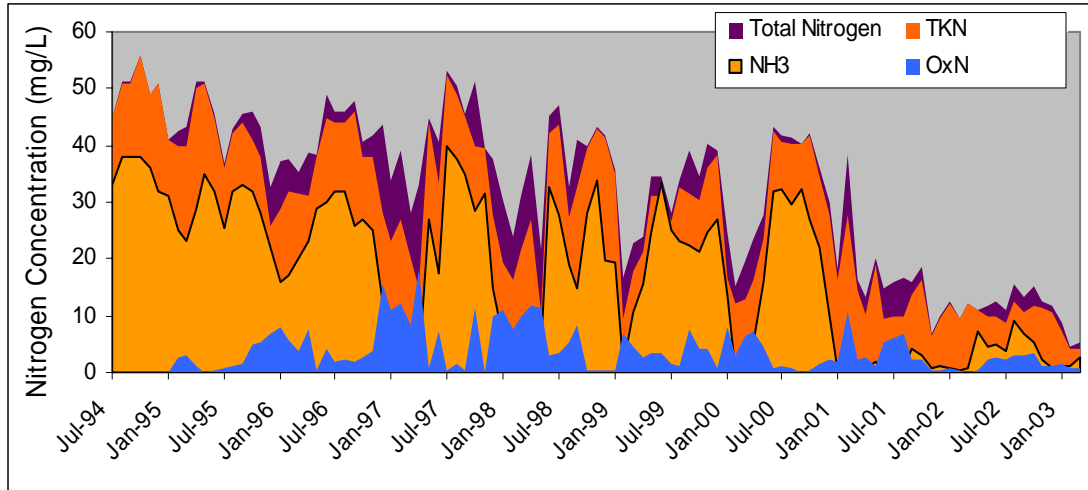
- ANZECC/ARMCANZ (1992) National Water Quality Management Strategy Guidelines for Fresh and Marine Waters. Australian and New Zealand Environment and Conservation Council.
- ANZECC/ARMCANZ (2000) National Water Quality Management Strategy Guidelines for Fresh and Marine Waters. Australian and New Zealand Environment and Conservation Council.
- Argue J R (2003) *WSUD: basic procedures for 'source control' of stormwater: A handbook for Australian practice*. SIA/UniSA, Adelaide.
- Borum J (1985) Development of epiphytic communities on eelgrass (*Zostera marina*) along a nutrient gradient in a Danish estuary. *Marine Biology*. 87,2:211-218.
- Bramley H, Keane R and Dillon P (2000) *The potential for ingress of saline groundwater to sewers in the Adelaide Metropolitan area: An assessment using a Geographical Information System. Report No. 95*. Centre for Groundwater Studies, Flinders University, SA.
- Bryars S and Neverauskas V (2002) Natural Recolonisation of seagrass at a disused sewage sludge outfall in Gulf St Vincent. In Seddon S and Murray-Jones S (Eds) *Proceedings of the Seagrass Restoration Workshop for Gulf St Vincent*.
- CCE (1985) *Disposal of digested sludge from Glenelg STW*. Engineering and Water Supply Dept. S.A.
- CCREM (1987) *Canadian Water Quality Guidelines*. Canadian Council of Resource and Environmental Ministers, Inland Waters Directorate, Environment Canada, Ottawa.
- EPA (1998) *Changes in seagrass coverage and links to water quality off the Adelaide Metropolitan Coastline*. Environmental Protection Agency, South Australia.
- EWS (1966) *Port Adelaide Treatment Works*. The Engineering and Water Supply Department, unpublished report.
- EWS (1971) *Review of sludge disposal practice for water and sewage treatment works. 15th biennial conference of engineers*. Perth W.A.
- EWS (1973) *Gulf St. Vincent Water Pollution Studies. Progress Report – May 1973*. EWS 3876/70
- EWS (1991a) *Glenelg / Port Adelaide sewage treatment works sludge disposal to Bolivar: Concept design report*. EWS 91/1.
- EWS (1991b) *Cessation of sewage discharge into the sea from Glenelg and Port Adelaide sewage treatment works*. EWS 4994/91.
- Florence T M and Batley G E (1980) Chemical speciation in natural waters. *CRC Crit. Rev. Anal. Chem.*, 9, 219-296.
- Ferguson E A and Hogstrand C (1998) Acute silver toxicity to seawater acclimated rainbow trout: Influence of salinity on toxicity and silver speciation. *Environ. Toxicol. Chem.* 14, 589-594.
- Gobbie M (1991) *Metropolitan Adelaide Sewage Sludge Management Plan*. EWS 89/34 (582/90).
- GHD (1993) *Strategies for future operation of the Christies Beach Wastewater Treatment Plant and Wastewater Treatment and Disposal for the Willunga Basin*. Gutteridge, Haskins and Davey, report to EWS.
- Hamilton C (2002) SA Water's Commitment to Gulf St Vincent. In Seddon S and Murray-Jones S (Eds) *Proceedings of the Seagrass Restoration Workshop for Gulf St Vincent*.

- Hart (1997) *Near-shore seagrass change between 1949 and 1996*. Department of Environment and Natural Resources, South Australia.
- Harris G, Batley G, Fox D, Hall D, Jernakoff P, Molloy R, Murray A, Newell B, Parslow J, Skyring G, and Walker S. (1996) *Port Phillip Bay Environmental Study Final Report*. CSIRO, Canberra, Australia.
- Hunt S and Hedgcock S (1992) *Revised Environmental Quality Standards for nickel in water*. WRc report to the Department of the Environment DoE 2685/1.
- Johnston T H (1934) *Report on proposals for the discharge of effluent from the Glenelg Sewage Treatment Works*. Harrison Weir, Government Printer, Adelaide.
- Koch E W (1999) Sediment resuspension in a shallow *Thalassia testudinum* bank ex König bed. *Aquatic Botany* **65**:269-280.
- Lewis S A (1975) *Gulf St. Vincent Water Pollution Studies 1972 – 1975*. Report of the committee on the effects of land-based discharges from metropolitan Adelaide upon the marine environment of Gulf St Vincent. EWS 75/14.
- Mance G, Musselwhite C and Brown V M (1984) *Proposed Environmental Quality Standards for list II substances in water - Arsenic*. Technical Report TR 212
- Muñoz J T (1995) Effects of some Plant Growth Regulators on the Seagrass *Cymodocea nodosa* (Urlica) Ascherson. *Aquatic Botany*. 51:311-318.
- Melbourne Water (2002) *Eastern Treatment Plant Update*. Melbourne Water Newsletter, March 2002.
- Melbourne Water (2003) *Environment Improvement Plan: Western Treatment Plant*. Melbourne Water, November 2003.
- Murray A G (1994) *Western Treatment Plant Outputs to Port Phillip Bay*. Technical Report No. 15, CSIRO Port Phillip Bay Environmental Study, Melbourne, Australia.
- Neverauskas V P (1985) *Port Adelaide STW sludge outfall - effect of discharge on the adjacent marine environment*. Progress Report July 1982 – May 1984. EWS 85/6.
- Neverauskas V P (1987) *Port Adelaide STW sludge outfall - effect of discharge on the adjacent marine environment*. Final Report. EWS 87/28.
- Neverauskas V P (1989) *Glenelg STW sludge outfall; relocation of outfall to a site 7 km offshore, possible biological effects*. State Water Laboratory Report No. 21.
- Newell B, Molloy R and Fox D. (1999) *Environmental Impact Assessment and Review of Effluent Disposal Options for Eastern Treatment Plant*. CSIRO, Wembley, WA, Australia.
- Ochota P (1981) Construction of the Port Adelaide Sludge Outfall. Appendix V in Steffensen D A (1981) *Port Adelaide STW sludge outfall - effect of discharge on the adjacent marine environment*. Phase I base line study. EWS 81/8.
- Peterson S M and Batley G E (1992) *Road runoff and its impact on the aquatic environment: A review*. CSIRO Investigation Report to the NSW Roads and Traffic Authority.
- Petrusevics P M. (1986) *Drogue studies around the Port Adelaide sewage treatment works sludge outfall*. Report to Department of Environment and Planning.
- Ralph P J, Tomasko D, Moore K, Seddon S and Macinnis-Ng C M O. (2003) Human Impacts on Seagrass: Contamination and Eutrophication. Chapter 25 in Larkum A W D, Orth R J and Duarte C M (Eds). *Seagrass Biology*.
- Seddon S (2002) Issues for Seagrass Rehabilitation Along the Adelaide Metropolitan Coast: An Overview. In Seddon S and Murray-Jones S (Eds) *Proceedings of the Seagrass Restoration Workshop for Gulf St Vincent*.
- SADEC (1975) *Report of the Port Adelaide Sewage Sludge Outfall Working Group*. SADEC 5. South Australian Dept of Environment and Conservation.

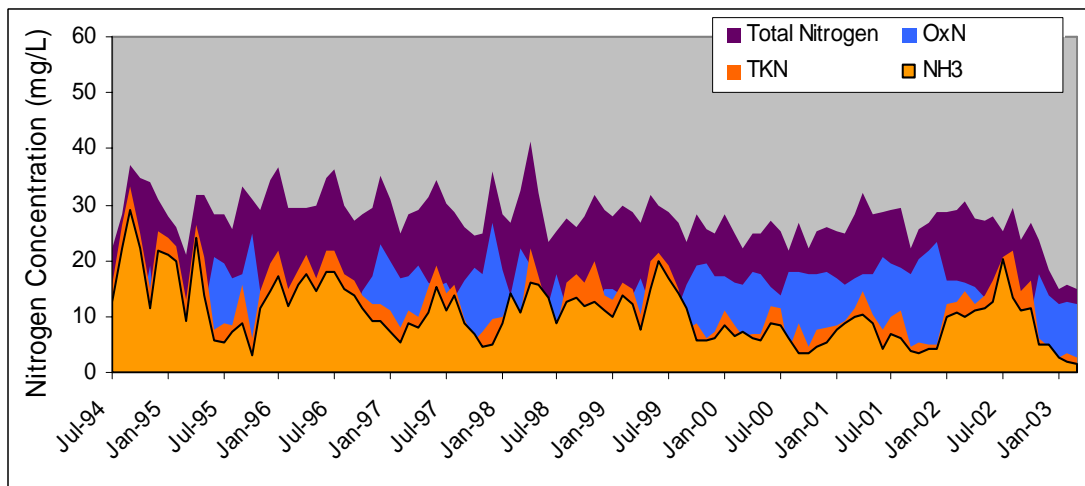
- SA Water (1995) *Adelaide Wastewater Treatment Plants – Environment Improvement Programs*. SA Water, South Australia.
- Sickerdick L (1995) *SA Water Reuse Schemes*. Proceedings of the AWWA seminar on Re-use of Sewage Effluent. 26 September 2003.
- Sickerdick L and Desmier R (2000) Use of Reclaimed Water from SA Water's Wastewater Treatment Plants. In Dillon P J (Ed) *Water Recycling Australia*. CSIRO and AWA.
- Smith I N H and Edwards V (1992) *Revised Environmental Quality Standards for Arsenic in water*. WRc report to the Department of the Environment DoE 2633/1.
- Steffensen D A (1981) *Port Adelaide STW sludge outfall - effect of discharge on the adjacent marine environment*. Phase I base line study. EWS 81/8.
- Steffensen D A (1985) *Gulf St. Vincent Water Pollution Studies Phase II 1976 - 1983. Part 1 Southern and Central Metropolitan Zones*. EWS 84/12 (142/84).
- Steffensen D A, I Kirkegaard and J Johnson (1989a) *Environmental impact of the effluent discharge from the Bolivar Sewage Treatment Works*. Unpublished EWS memorandum, enclosure with Steffensen *et al.* (1989b).
- Steffensen D A, I Kirkegaard and J Johnson (1989b) *Position and background papers on man-made changes to Gulf St Vincent*. Unpublished EWS report to the Department of Environment and Planning.
- Thomas R C (1981) Port Adelaide Sewage Treatment Works – Operation of Sludge Outfall to Sea. Appendix VI in Steffensen D A (1981) *Port Adelaide STW sludge outfall - effect of discharge on the adjacent marine environment*. Phase I base line study. EWS 81/8.
- Thomas R (1995) *Parliamentary Debates (Hansard)*, House of Assembly – Estimates Committee A, 20-23 June 1995, p. 96.
- Townsend M (2002) The Exposure of Adelaide's Seagrasses to Waves and Currents. In Seddon S and Murray-Jones S (Eds) *Proceedings of the Seagrass Restoration Workshop for Gulf St Vincent*.
- UKMSAC (2003) *Toxic substances profiles*. UK Marine Special Areas of Conservation Project. <http://ukmarinesac.org.uk/activities/water-quality/wq8.htm>
- Warrington P D (1993) *Ambient water quality criteria for silver*. Water Quality Branch, Environmental Protection Department, Ministry of Environment, Lands and Parks, British Columbia.
- WHO (1992) *Environmental Health Criteria No 135- Cadmium - Environmental Aspects*. World Health Organisation, Geneva.
- Young W (1992) *Revised Environmental Quality Standards for inorganic lead in water*. WRc report to the Department of the Environment DoE 2718/1.

Appendix I: Time series of monthly nutrient concentrations and loads from Bolivar, Glenelg and Christies Beach WWTPs

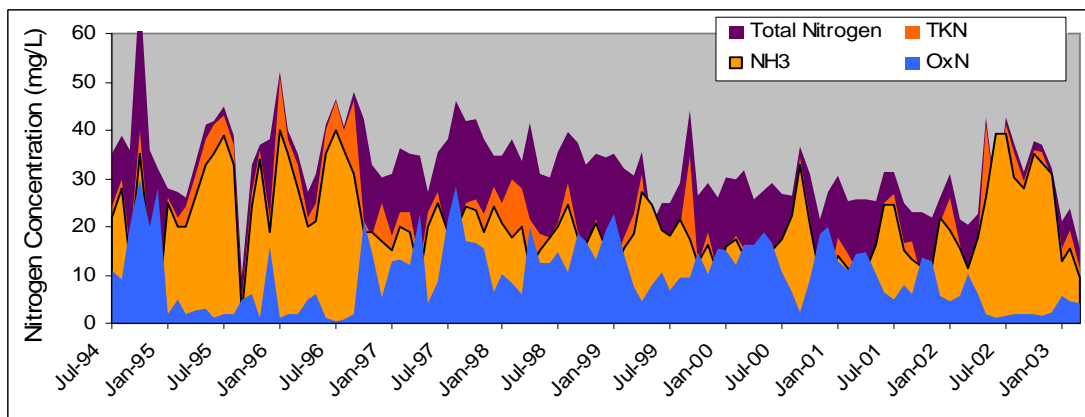
Bolivar concentrations



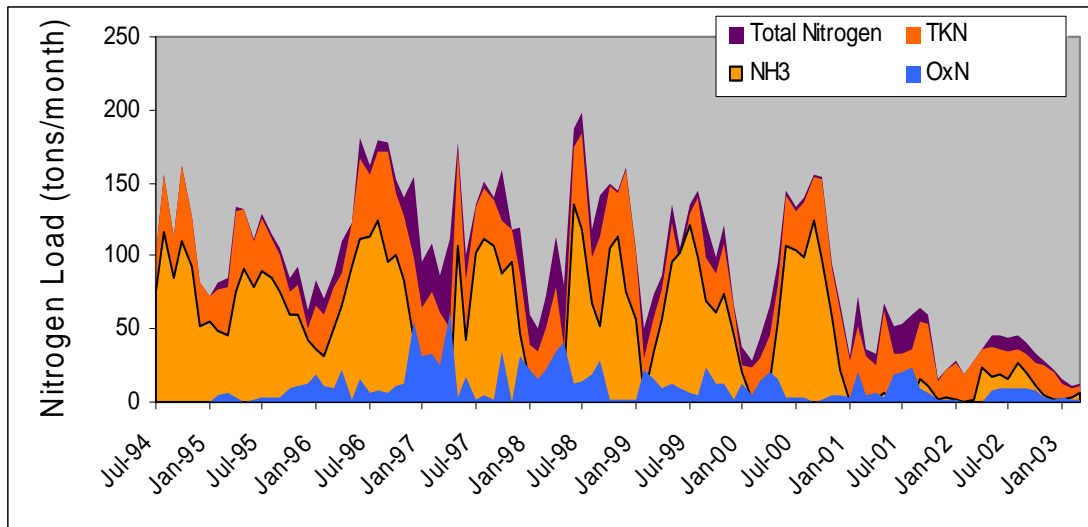
Glenelg concentrations



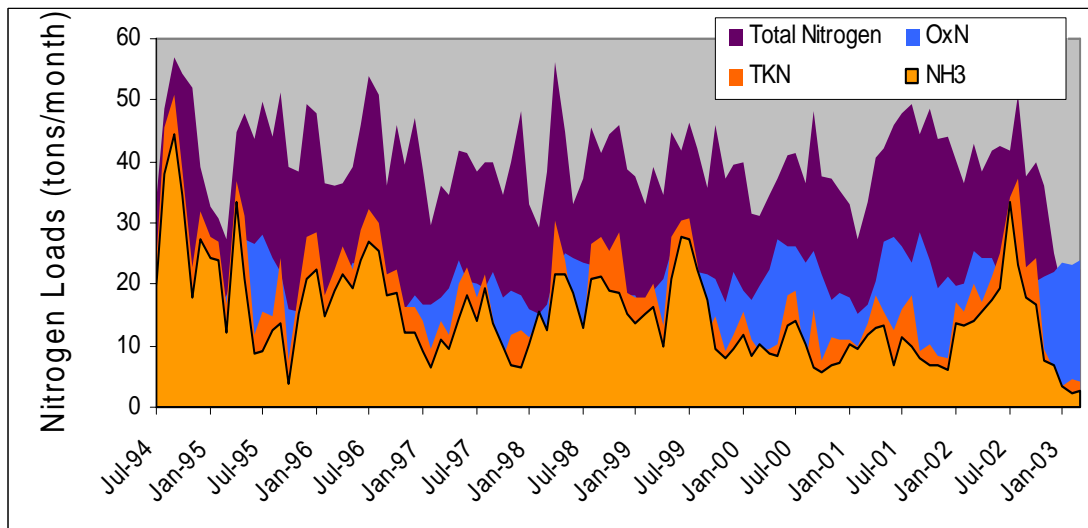
Christies Beach concentrations



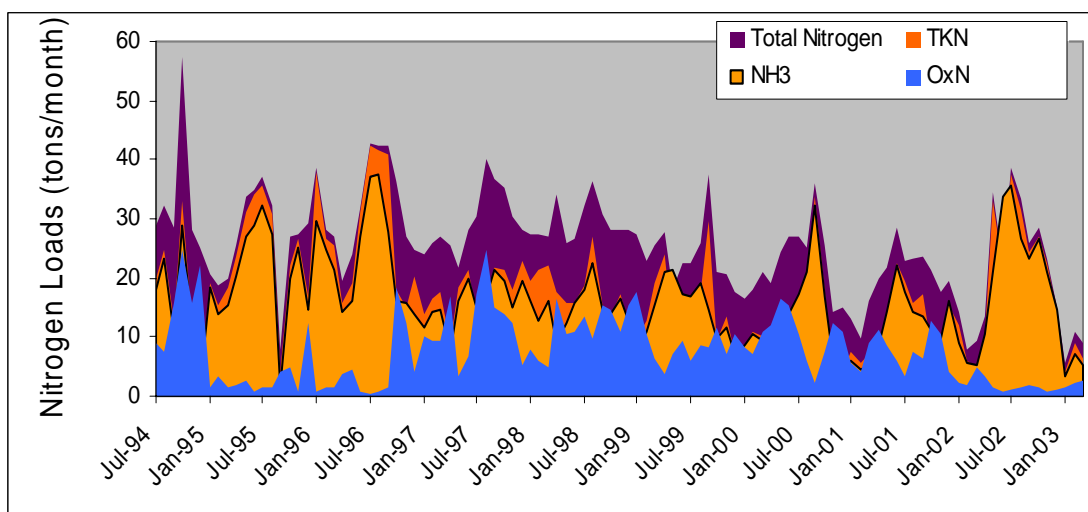
Bolivar loads



Glenelg loads



Christies Beach loads



Appendix II: Estimated total loads of metals discharged from the Metropolitan Adelaide sewage sludge outfalls

Port Adelaide WWTP – Sludge outfall data

	Mean (mg/L)	Percentile 05	Percentile 95	Std Deviation	Count	Total load (tonnes)	Daily load kg/d
Fe	216	79.3	308	<i>67.4</i>	64	459	82
Al	35.4	16.1	62	<i>13.1</i>	63	75	13.4
Zn	31.1	13.0	63.4	<i>15.8</i>	71	66	11.8
Cu	19.2	8.18	33	<i>6.96</i>	72	40.7	7.27
Cr	7.54	2.94	13.1	<i>2.95</i>	72	16	2.86
Pb	3.03	1.35	5.10	<i>1.22</i>	64	6.43	1.15
B	2.09	0.110	3.74	<i>0.799</i>	34	4.43	0.79
Ni	1.35	0.407	4.08	<i>1.12</i>	63	2.87	0.51
Mn	1.29	0.507	2.01	<i>0.408</i>	70	2.74	0.49
Ag	0.934	0.213	2.27	<i>0.641</i>	33	1.98	0.35
Cd	0.762	0.058	1.47	<i>0.391</i>	63	1.61	0.29
Mo	0.325	0.107	0.673	<i>0.226</i>	57	0.69	0.12
Sb	0.065	0.016	0.286	<i>0.102</i>	32	0.14	0.025
As	0.053	0.013	0.105	<i>0.026</i>	31	0.11	0.020
Hg	0.036	0.011	0.074	<i>0.019</i>	42	0.08	0.014

Glenelg WWTP – Sludge outfall data

	Mean (mg/L)	Percentile 05	Percentile 95	Std Deviation	Count	Total load (tonnes)	Daily load kg/d
Fe	113	62.4	175	<i>104</i>	64	578	49.7
Al	46.8	30	78.5	<i>14.9</i>	63	239	20.6
Zn	39.8	25.8	58.7	<i>10.2</i>	74	203	17.5
Cu	27	16.5	40	<i>6.82</i>	75	138	11.9
Cr	17.2	9.40	27.3	<i>5.54</i>	75	88.0	7.57
Ni	5.67	2.97	9.33	<i>2.03</i>	65	29.0	2.50
Pb	3.83	1.65	6.63	<i>1.65</i>	64	19.6	1.69
Ag	2.32	0.688	4.71	<i>1.30</i>	28	11.8	1.02
Mn	1.82	1.37	2.43	<i>0.619</i>	64	9.31	0.80
B	1.28	0.6	2.84	<i>0.811</i>	35	6.54	0.56
Cd	0.201	0.038	0.28	<i>0.428</i>	61	1.03	0.09
Mo	0.157	0.053	0.349	<i>0.089</i>	63	0.80	0.07
Sb	0.061	0.026	0.105	<i>0.019</i>	28	0.31	0.027
Hg	0.043	0.011	0.095	<i>0.021</i>	63	0.22	0.019
As	0.039	0.006	0.092	<i>0.025</i>	28	0.20	0.017

Appendix III: Summary of sampling and detection of organic chemicals in Adelaide Metropolitan WWTP effluents between May 1992 and October 1994.

	Bolivar	Glenelg	Christies Beach
Pesticides			
n samples	24	19	24
n _d detections	16 (*288, 5.5%)	17 (228, 7%)	20 (288, 6.9%)
median concentration of detections µg/L	0.05	0.04	0.05
VCHs			
n samples	16	13	16
n _d detections	2 (128, 1.6%)	3 (104, 2.9%)	6 (128, 4.7%)
median concentration of detections µg/L	7.0	8.0	17

Pesticides tested: aldrin, chlordane, chlorpyrifos, dde, dieldrin, endosulphan, heptachlor, heptachlor epoxide, lindane, malathion, diazinon. (*The first figure in brackets shows the potential number of detections if all samples were positive for every substance, the percentage figure gives the percentage detection rate)

VCHs tested: chloroform, bromodichloromethane, bromoform, carbon tetrachloride, trichloroethylene, tetrachloroethylene, PCB, PAH.

Listing of organic chemicals analysed in Adelaide Metropolitan WWTP effluents, May 1992 to October 1994.

PaHs	VCHs (Solvents)	Pesticide	POPs
acenaphthene	bromodichloromethane	alachlor (lasso)	aroclor 1016
anthracene	bromoform	aldrin	
benzo (a) pyene	carbon tetrachloride	dieldrin	
benzo (a) anthracene	chloroform	endosulfan	
benzo (ghi) perylene	tetrachloroethane	endosulphan sulphate	
benzo (k) fluoranthene	trichloroethane	heptachlor	
chrysene	trichloroethylene	heptachlor epoxide	
dibenzo (ah) anthracene	total VCH	lindane	
dibromochloromethane		phenols	
fluoranthene		propachlor (ramrod)	
fluorine		trifluralin	
indeno (123-cd) pyene		total herbicides	
naphthalene		total insecticides	
phenanthrene			
pyene			

Appendix IV: Listing of analyses for organic chemicals in Adelaide Metropolitan WWTP effluents 1998 to 2001.

Phenols

Phenol
 2-Chlorophenol
 4-Methylphenol
 3- & 4-Methylphenol
 2-Nitrophenol
 2,4-Dimethylphenol
 2,4-Dichlorophenol

 2,6-Dichlorophenol
 4-Chloro-3-methylphenol
 2,4,6-Trichlorophenol
 2,4,5-Trichlorophenol
 Pentachlorophenol

Polynuclear Aromatics

Naphthalene
 2-Methylnaphthalene
 2-Chloronaphthalene
 Acenaphthylene
 Acenaphthene
 Fluorene
 Phenanthrene
 Anthracene
 Fluoranthene
 Pyrene
 N-2Fluorenylacetamide
 Benz(a)anthracene
 Chrysene
 Benzo(b) & (k)fluoranthene
 7,12-Dimethylbenz(a)anthracene
 Benz(a)pyrene
 3-Methylcholathrene
 Indeno(1,2,3-cd)pyrene
 Dibenz(a,h)anthracene
 Benzo(g,h,i)perylene

Phthalate Esters

Dimethyl phthalate (00-01 only)
 Diethyl phthalate (00-01 only)
 Di-n-butyl phthalate (00-01 only)
 Butyl benzyl phthalate (00-01 only)
 Bis (2-ethylhexyl) phthalate (00-01 only)
 Di-n-octyl phthalate (00-01 only)

Nitrosamines

N-Nitrosomethylethylamine (00-01 only)
 N-Nitrosodiethylamine (00-01 only)
 N-Nitrosopyrrolidine (00-01 only)
 N-Nitrosomorpholine (00-01 only)
 N-Nitrosodi-n-propylamine (00-01 only)
 M-Nitrosopiperidine (00-01 only)
 N-Nitrosodibutylamine (00-01 only)
 N-Nitrosodiphenyl & Diphenylamine (00-01 only)
 Methapylene (00-01 only)

Nitroaromatics and Cyclic Ketones

2-Picoline (00-01 only)
 Acetophenone (00-01 only)
 Nitrobenzene (00-01 only)
 Isophorone (00-01 only)
 2,6-Dinitrotoluene (00-01 only)
 2,4-Dinitrotoluene (00-01 only)
 1-Naphthylamine (00-01 only)
 4-Nitroquinoline-N-oxide (00-01 only)
 5-Nitro-o-toluidine (00-01 only)
 Azobenzene (00-01 only)
 1,3,5-Trinitrobenzene (00-01 only)
 Phenacetin (00-01 only)
 4-Aminobiphenyl (00-01 only)
 Pentachloronitrobenzene (00-01 only)
 Pronamide (00-01 only)
 Dimethylaminoazobenzene (00-01 only)
 Chlorobenzilate (00-01 only)

Haloethers

Bis (2-chloroethyl) ether (00-01 only)
 Bis (2-chloroethoxy) methane (00-01 only)
 4-Chlorophenyl phenyl ether (00-01 only)
 4-Bromophenyl phenyl ether (00-01 only)

Chlorinated Hydrocarbons

1,3-Dichlorobenzene
 1,4-Dichlorobenzene
 1,2-Dichlorobenzene
 Hexachloroethane
 1,2,4-Trichlorobenzene
 Hexachloropropylene
 Hexachlorobutadiene
 Hexachlorocyclopentadiene
 Pentachlorobenzene
 Hexachlorobenzene

Anilines and Benzidines

Aniline	(00-01 only)
4-Chloroaniline	(00-01 only)
2-Nitroaniline	(00-01 only)
3-Nitroaniline	(00-01 only)
Dibenzfuran	(00-01 only)
4-Nitroaniline	(00-01 only)
Carbazole	(00-01 only)
3,3'-Dichlorobenzidine	(00-01 only)

Monocyclic Hydrocarbons

Benzene	(00-01 only)
Toluene	(00-01 only)
Ethylbenzene	(00-01 only)
meta- & para-Xylene	(00-01 only)
Styrene	(00-01 only)
ortho-Xylene	(00-01 only)
Isopropylbenzene	(00-01 only)
n-Propylbenzene	(00-01 only)
1,3,5-Trimethylbenzene	(00-01 only)
sec-Butylbenzene	(00-01 only)
1,2,4-Trimethylbenzene	(00-01 only)
tert-Butylbenzene	(00-01 only)
p-Isopropyltoluene	(00-01 only)
n-Butylbenzene	(00-01 only)

Oxygenated Hydrocarbons

Vinyl acetate	(00-01 only)
2-Butanone (MEK)	(00-01 only)
4-Methyl-2-pentanone (MIBK)	(00-01 only)
2-Hexanone (MBK)	(00-01 only)

Sulfonated Compounds

Carbon disulfide	(00-01 only)
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Fumigants

2,2-Dichloropropane	(00-01 only)
1,2-Dichloropropane	(00-01 only)
cis-1,3-Dichloropropylene	(00-01 only)
trans-1,3-Dichloropropylene	(00-01 only)
1,2-Dibromoethane (EDB)	(00-01 only)

Halogenated Aliphatic Hydrocarbons (VOL)

Dichlorodifluoromethane	(00-01 only)
Chloromethane	(00-01 only)
Vinyl chloride	(00-01 only)
Bromomethane	(00-01 only)
Chloroethane	(00-01 only)
Trichlorofluoromethane	(00-01 only)
1,1-Dichloroethane	(00-01 only)
Idomethane	(00-01 only)
trans-1,2-Dichloroethane	(00-01 only)
1,1-Dichloroethane	(00-01 only)
cis-1,2-Dichloroethane	(00-01 only)
1,1,1-Trichloroethane	(98-99 & 00-01 only)
1,1-Dichloropropylene	(00-01 only)
Carbon tetrachloride	(98-99 & 00-01 only)
1,2-Dichloroethane	(98-99 & 00-01 only)
Trichloroethane	(98-99 & 00-01 only)
Dibromoethane	(00-01 only)
1,1,2-Trichloroethane	(00-01 only)
1,3-Dichloropropane	(00-01 only)
Tetrachloroethane	(98-99 & 00-01 only)
1,1,1,2-Tetrachloroethane	(00-01 only)
trans-1,2-Dichloroethane	(00-01 only)
cis-1,2-Dichloroethane	(00-01 only)
1,1,2,2-Tetrachloroethane	(98-99 & 00-01 only)
1,2,3-Trichloropropane	(00-01 only)
Pentachloroethane	(00-01 only)
1,2-Dibromo-3-chloropropane	(00-01 only)
Hexachlorobutadiene	(00-01 only)

Halogenated Aromatic Hydrocarbons (VOL)

Chlorobenzene	
Bromobenzene	
2-Chlorotoluene	
4-Chlorotoluene	
1,3-Dichlorobenzene	
1,4-Dichlorobenzene	(98-99 & 00-01 only)
1,2-Dichlorobenzene	(98-99 & 00-01 only)
1,2,4-Trichlorobenzene	(98-99 & 00-01 only)
1,2,3-Trichlorobenzene	(98-99 & 00-01 only)

Trihalomethanes

Chloroform
Bromodichloromethane
Dibromochloromethane
Bromform

Organochlorine Pesticides

HCB
Chlordane-trans
Chlordane-cis
Endrin sulfate
Endrin Ketone
Methoxychlor
alpha-BHC
beta-BHC & gamma-BHC
delta-BHC
Heptachlor
Aldrin
Heptachlor epoxide
Endosulfan 1
4,4'-DDE
Dieldrin
Endrin
Endosulfan 2
4,4'-DDD
Endosulfan sulfate
4,4'-DDT

Phenoxy acid herbicides

4-Chlorophenoxy acetic acid
Dicamba
Mecoprop
MCPA
2,6-D
2,4-DP
2,4-D
Triclopy
2,4,6-T
2,4,5-TP (Silvex)
2,4,5-T
MCPB
2,4-DP

Organophosphorous Pesticides

Demeton-S-methyl
Monocrotophos
Parathion-methyl
Parathion
Bromophos-ethyl
Fenamiphos
Carbophenothion
Azinphos-methyl
Dichlorvos
Dimethoate
Diazinon
Chlorpyrifos methyl
Malathion
Fenthion
Chlorpyrifos
Pirimiphos ethyl
Chorfenvinphos-E
Chorfenvinphos-Z
Prothiofos
Ethion

Appendix V: Comparison of Metro Adelaide WWTPs with Melbourne's WTP and ETP

The two major WWTPs of Melbourne the Western and Eastern Treatment Plants (WTP and ETP) discharge a total of around 802 ML/d (mean daily flow for 1990-95 430 ML/d from WTP (Harris *et al.*, 1996) and 1993-97 373 ML/d from ETP (Newell *et al.*, 1999)). The ETP receives around 42% of Melbourne's waste water the WTP serves approximately 50% of Melbourne's population of 3 million and 90% of the city's trade waste. The sum of the mean daily flows from Bolivar, Glenelg and Christies Beach WWTPs for 1990 to 1997 was 165.4 ML/d. This suggests that with a third of the population of Melbourne, Adelaide uses a fifth of the annual volume consumed by Melbourne.

The suspended load discharged from Metro Adelaide WWTPs is less than from the WTP and ETP (Table A6.1), although the suspended particulate matter concentrations from Bolivar have been far greater than the other sites due to the production of algal particles in the final treatment lagoons.

Table A6.1 Comparison of concentrations and loads of suspended particulate matter in Melbourne and Adelaide Metropolitan recycled water.

	Concentration mg/L	Load T/y
Bolivar 1994 to 2000	90.2	3366
Glenelg 1994 to 2000	15.6	265
Christies Beach to 2000	13.1	123
ETP 1995/7 (Newell <i>et al.</i> , 1999)	17.5	2383
WTP (Harris <i>et al.</i> , 1996)	38.3	6000

The nutrient concentrations have historically been comparable between the different treatment plants (Table A6.2). The total loads discharged by Melbourne plants was significantly greater as a consequence of the greater flow volume. The EIP upgrades of the

Table A6.2 Comparison of concentrations and loads of total nitrogen and total phosphorus in Melbourne and Adelaide Metropolitan recycled water, prior to EIP upgrades.

	Median N Concentration mg/L	Load T/y	Median P Concentration mg/L	Load T/y
Bolivar 1994 to 2000	39.3	1380	6.0	210.4
Glenelg 1994 to 2000	28.1	484	7.8	133.3
Christies Beach 1994 to 2000	33.8	318.5	8.6	79.8
Total		2183		424
ETP 1995/6 (Newell <i>et al.</i> , 1999)	33.7	4588	6.0	2383
WTP 1993/5 (Harris <i>et al.</i> , 1996)	21.7	3373	7.6	1173
Total		7961		3555

Adelaide plants is currently reducing the nitrogen load to Gulf St Vincent, similar upgrades have reduced the WTP nitrogen load to 3500 T/y (Melbourne Water, 2003), similarly proposed upgrades to ETP are aimed at a 75% reduction in ammoniacal nitrogen from the plant (Melbourne Water, 2002).

Table A6.3 Metals concentrations in Metropolitan Adelaide effluents and Melbourne effluents in mg/L.

	Bolivar 2000/2	Glenelg 2000/2	Christies Beach ETP 2000/2	ETP 1995/96	WTP Murray (1994)
Cd	0.0005	0.0005	0.0005	<0.001	0.0001
Cr	0.011	0.010	0.007	<0.01	0.027
Cu	0.056	0.046	0.027	0.02	0.019
Fe	0.278	0.084	0.037	nd	1.235
Hg	0.0005	0.0005	0.0005	<0.0002	0.0001
Ni	0.020	0.020	0.008	nd	0.034
Pb	0.004	0.003	0.002	<0.01	0.008
Zn	0.051	0.081	0.052	nd	0.040

Table A6.3 provides concentrations of heavy metals in Adelaide and Melbourne treated effluent. The values do not indicate any major differences between the different plants, although the Bolivar and Glenelg effluents are elevated in copper compared to the other plants.

Information on organics in treated wastewater from the two Melbourne plants are consistent with the results for the Adelaide plants (Harris *et al.*, 1996), indicating that these chemicals do not readily pass through the treatment process, they are degraded and settle-out within the works.

This brief comparison of the Adelaide and Melbourne effluents indicates that there are no major differences in quality and that the differences in load are merely a reflection of the larger volume of water processed for the greater population of Melbourne. The volume of treated wastewater from the Melbourne plants is greater per capita than the volume from the Adelaide plants. There may be a simple reason for this and this may warrant a further brief investigation.