# Technical Report

Adelaide Coastal Waters Study

***Stage 2** Research Program 2003 – 2005*

Technical Report No. 14 July 2006

# *Field surveys 2003-2005: Assessment of the quality of Adelaide's coastal waters, sediments and seagrasses*

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*Field surveys 2003-2005: Assessment of the quality of Adelaide's coastal waters, sediments and seagrasses*


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**"The coast-line from Outer Harbour to Marino consists of very gradually shelving sands finally succeeded by the blue line where the water deepens and the yellow of the sand is replaced by a dark blue due to the under-water meadows of *Cymodocea* (*Amphibolis* Edit.) and *Posidonia*."**

**Cleland (1935)**

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[1] While Rachel Wear is a co-author of the report, she contributed to Appendices A-C only, and was therefore considered by Dr Anthony Fowler to be an appropriate reviewer of the main body of the report.

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# Executive summary


Major losses of subtidal seagrasses (viz. *Amphibolis* and *Posidonia*) have occurred along the Adelaide metropolitan coastline (particularly in Holdfast Bay), with possible links to anthropogenic activities including land-based discharges. In 2003, Task EP 1 of the Adelaide Coastal Waters Study (ACWS) began an investigation of the current status of seagrass meadows and possible causes of past and ongoing decline in the defined ACWS region of Port Gawler in the north to Sellicks Beach in the south. The present report documents the results of a series of field surveys conducted during 2003-2005 that investigated the quality of Adelaide's coastal waters, sediments and seagrasses, and then put the results into the context of available historical data. Major outcomes of the report are:

*Water quality*

- Due to the various coastal inputs operating over the past 60 years, Adelaide's coastal waters are no longer pristine, with elevated levels of nutrients, toxicants, and turbidity regularly being reported over the last 30 years in both coastal inputs and coastal waters.
- Historical data show that nearshore (< 5 m depth) waters in Holdfast Bay have consistently had elevated levels of nutrients since at least the 1970's and must be considered as eutrophic in the context of an oligotrophic system.
- The area of most pronounced elevation of nutrients between Glenelg and Grange coincides with the area of major nearshore seagrass losses in Holdfast Bay.
- Water quality monitoring over existing offshore (5 and 10 m depths) seagrasses in the ACWS region between 2003-2004 showed no indication of elevated nutrients. However, results from a stable nitrogen isotope survey of offshore seagrasses indicates that the entire coast between Port Gawler and Port Noarlunga is being influenced by nitrogen from wastewater treatment plant (WWTP) and industrial outfalls. This outcome clearly demonstrates the inadequacies of solely monitoring dissolved nutrients in an oligotrophic system such as the offshore ACWS region.
- Water quality monitoring after three significant rainfall events during 2004-2005 indicated that major stormwater flows have a minor but localized influence on ambient nutrient levels in Holdfast Bay. In contrast, tests for other potential toxicants during these rainfall surveys detected only one compound (simazine) at a very low concentration out of a large suite of herbicides and pesticides. These results are in general agreement with results of previous studies.

*Sediment quality*

- Sediment quality analyses found very low or undetectable levels of toxicants in marine sediment samples collected adjacent to major stormwater outlets where they would most likely occur, and at offshore sites where terrestrially-derived sediments may be transported.
- No evidence was found to suggest that sediment quality is degraded due to toxicants.

*Seagrass quality*

- Historical records show that *Amphibolis* and *Posidonia* originally dominated Holdfast Bay and the nearshore region between Port Gawler and Outer Harbour. Since the 1940s, major losses of these meadows have occurred in several discrete locations.
- *Posidonia* is now the dominant seagrass across the ACWS area.
- *Amphibolis* is abundant only in the nearshore area from Semaphore to Henley Beach and Brighton to Marino. *Amphibolis* appears to have disappeared from much of the nearshore area between Port Gawler and Semaphore, and the area between Henley Beach and Brighton.

- The dominant species of *Posidonia* changes from *P.sinuosa* through *P. angustifolia* to *P. coriacea* going from north to south in depths of 5 and 10 m across the ACWS region. The lower depth limit (~15-18 m) of *Posidonia* in Holdfast Bay does not appear to have changed over the past 40 years.
- Aboveground biomass of representative offshore (5 & 10 m depths) *Posidonia* meadows generally appears healthy (although meadows directly adjacent to land-based discharges were not surveyed). However, cover of seagrass meadows is fragmented in the nearshore and offshore southern parts of Holdfast Bay, indicating a disturbed system.

*Possible links with seagrass loss*

- Toxicants from stormwater are unlikely to have caused large-scale historical seagrass losses and are unlikely to be contributing to any ongoing losses.
- Areas of major nearshore seagrass loss and apparent selective disappearance of *Amphibolis* coincide with the zones of greatest influence from Bolivar and Glenelg WWTP and Penrice outfalls (inferred from present day patterns of stable isotopes). There is a clear link between elevated nutrients and seagrass decline that warrants experimental investigation (see ACWS Technical Report No. 11 by Collings *et al.* 2006a).

*Recommendations for future monitoring/research*

- Experimentally test and model the effects of increased nutrients and turbidity on *Amphibolis* and *Posidonia*.
- Survey and commence long-term monitoring of seagrass quality at sites adjacent to land-based discharges and at suitable control sites.
- Survey and commence long-term monitoring of the outer depth margin of *Posidonia* meadows in Holdfast Bay.
- Survey and commence long-term monitoring of seagrass meadow fragmentation at a range of sites in Holdfast Bay.
- Conduct a detailed survey of the current distribution of *Amphibolis* between Port Gawler and Sellicks Beach.
- Conduct a spatially intensive $\delta^{15}N$ survey to determine the offshore and northern extents of nitrogen influence from WWTP and industrial outfalls in the ACWS region, and also characterise $\delta^{15}N$ signatures of potential nitrogen sources.

*Recommendations for management*

- Nutrient loads entering Adelaide's coastal waters need to be reduced in order for the system to have any chance of returning to its natural oligotrophic state.

# 1. Introduction

## 1.1. Background

### 1.1.1. Seagrass loss

Since the 1940s, over 5000 ha of seagrass meadows have been lost from the Adelaide metropolitan coast. In particular, major losses of nearshore seagrasses have occurred in the region between Outer Harbour and Seacliff (Westphalen *et al.* 2004). Degradation and loss of seagrass meadows is a major cause of concern for coastal managers due to the importance of these habitats to near-shore productivity, seabed stability, and biodiversity. Seagrass losses along the Adelaide coast have previously been linked to the construction of stormwater drains, sewage and sludge outfalls, coastal developments, and the re-channelling of the Torrens River to the sea (Westphalen *et al.* 2004). Nonetheless, the primary causes of seagrass decline are poorly understood for the Adelaide metropolitan coast where seagrass loss has mainly occurred from the shallow inshore margin advancing seaward. Potential causes of seagrass decline along the Adelaide coast include elevated nutrients, toxicants, increased turbidity, and decreased salinity (see Westphalen *et al.* 2004 for a review). Potential toxicants include heavy metals, pesticides, herbicides, and petrochemicals (Westphalen *et al.* 2004). Erosion processes have also played a role in ongoing seagrass losses (Clarke 1987, Seddon 2002). Furthermore, Tanner (2005) reported the disappearance of deep-water seagrasses (*Heterozostera*) from lower Gulf St Vincent between the 1970's and 2000/01, postulating that a possible cause of the losses was a long-term increase in turbidity due to coastal inputs from Adelaide.

### 1.1.2. Coastal inputs

Prior to European settlement there were very few coastal inputs to the Adelaide coast, particularly in the region between Outer Harbour and Seacliff where major nearshore losses have occurred. While the Patawalonga Creek and the Port River may have historically delivered some freshwater to the coast, engineering works and urbanisation during the 20th century have substantially increased coastal inputs (Wilkinson 2005, Wilkinson *et al.* 2003, 2005 a, b). Major changes included the diversion of the Torrens River away from inland wetlands directly to the ocean at West Beach, the commencement of industrial discharges into the Port River system (Wilkinson *et al.* 2005b), and the construction of numerous stormwater drains and several wastewater treatment plant (WWTP) outfalls (Wilkinson *et al.* 2005a, Figure 1).

Stormwater may contain elevated levels of nutrients, toxicants, colour dissolved organic matter, and sediments. Increased nutrients are also currently associated with Bolivar, Glenelg, and Christies Beach WWTP outfalls along the open Adelaide coast and the Penrice soda factory discharge within the Port River (Wilkinson *et al.* 2003, 2005, Figure 1). Sludge discharges also released nutrients, toxicants and sediments to the offshore coastal environment of Adelaide prior to the 1990s (Wilkinson *et al.* 2003, Figure 1). Due to the often poor water quality associated with the various coastal inputs, they have all been implicated at various times as the cause(s) of seagrass decline along the Adelaide coast (see Westphalen *et al.* 2004 for a review). In addition, stormwater and WWTP discharges represent a substantial input of freshwater to Adelaide's coastal waters that was not present prior to European settlement (Wilkinson *et al.* 2005a). It is also possible that industrial effluent discharges in the Port River system could be transported to the open coastal waters off Adelaide (Pattiaratchi *et al.* 2006).

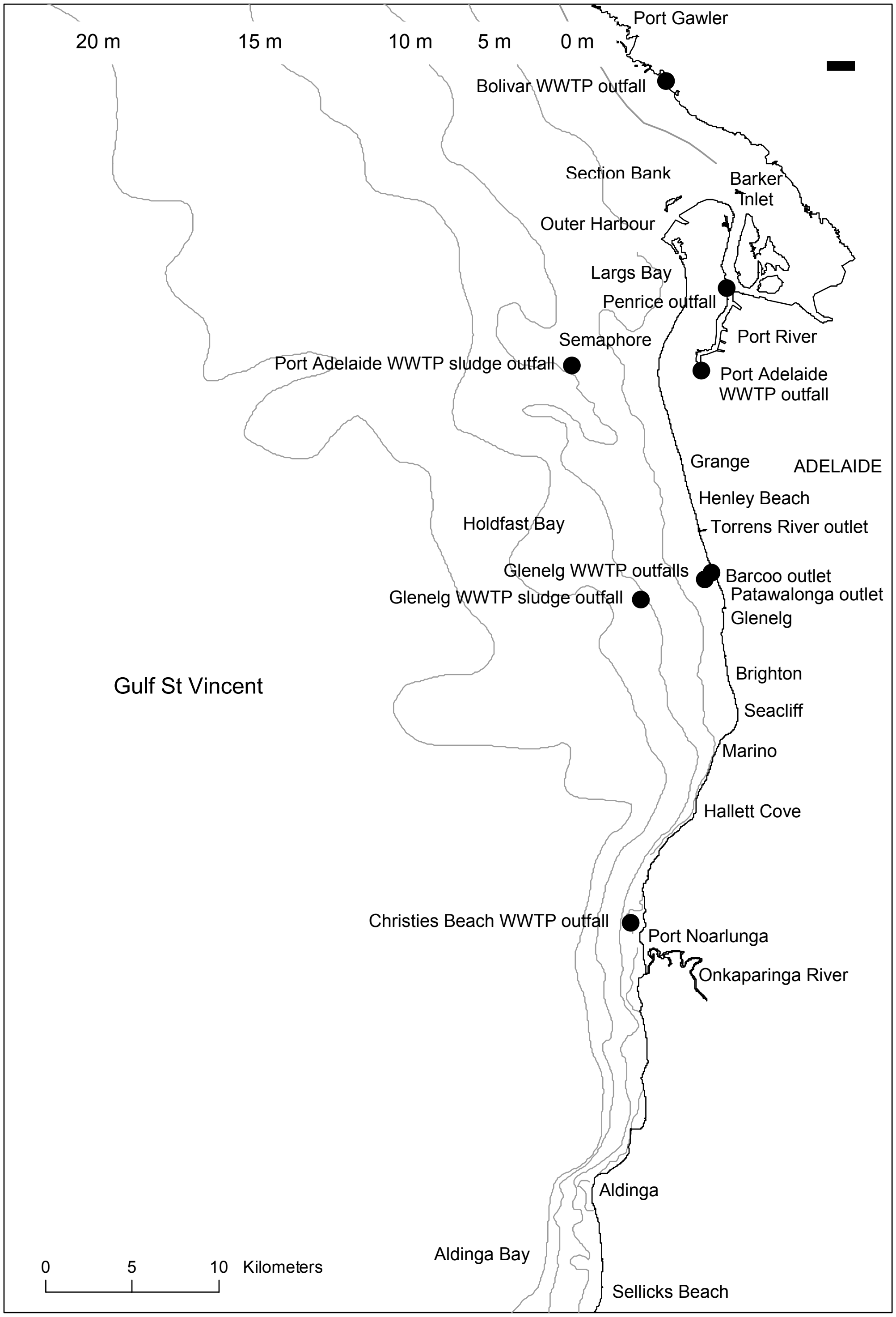


**Figure 1.** Map of the Adelaide Coastal Waters Study region in eastern Gulf St Vincent showing key locations and spatial extent of the study region. WWTP = wastewater treatment plant. Note that the sludge outfalls are no longer operational.

Whilst we have a reasonable understanding of historical and present coastal inputs from the WWTP and industrial outfalls (see Wilkinson *et al.* 2003, 2005b) and adjacent catchments (see Wilkinson 2005, Wilkinson *et al.* 2005a), we lack information on the fate of these inputs once they are released into Adelaide's coastal waters. Identifying possible links between these coastal inputs and the status and trends in the quality of Adelaide's coastal waters, sediments, and seagrasses is crucial to understanding historical and ongoing seagrass losses. Thus the following three sections deal with our understanding of the historical and present quality of Adelaide's coastal waters, sediments, and seagrasses.

### 1.1.3. Coastal water quality

Gulf St Vincent is a large, shallow (<40 m depth) marine embayment that is connected to the open ocean via Investigator Strait and Backstairs Passage. The waters of Gulf St Vincent are usually well mixed vertically by both tidal and wind-generated currents (Bye 1976). Due to the shallow bathymetry of the Gulf, water temperatures are strongly influenced by changes in seasonal air temperature. Gulf St Vincent receives relatively little freshwater inflow, with some inputs on the eastern coast (including the ACWS region between Port Gawler and Sellicks Beach; see earlier, and Wilkinson *et al.* 2005a) and virtually no inputs on the western coast. Prior to European settlement, the status of Adelaide's coastal waters was probably similar to the present status of most other parts of Gulf St Vincent, i.e. relatively clear water with low levels of both nutrients (oligotrophic) and toxicants. However, due to the various coastal inputs, Adelaide's coastal waters are no longer pristine, with elevated levels of nutrients, toxicants, and turbidity being detected and reported regularly over the last 30 years (See Appendix A).

Numerous potential indicators exist for detecting land-based and anthropogenic pollution in marine systems (ANZECC 2000). Indicators of nutrient enrichment of the water column include increased levels of organic and inorganic nitrogen (ammonia, ammonium, nitrate, nitrite), and phosphorus. Increased levels of chlorophyll-*a* associated with increased abundance of phytoplankton may also indicate nutrient enrichment of the water column. The presence of faecal coliforms and the bacterium, *Escherichia coli*, as well as increased levels of organic carbon are all indicators of organic pollution from land-based inputs. Increased levels of toxicants, such as heavy metals, may also indicate human pollution. Consequently, many of these parameters have been measured by various agencies in Adelaide's coastal waters over the last 30 years (See Appendix A). Recent advances in the use of stable isotopes for detecting and tracking the influence of anthropogenic nutrient sources may also be useful in determining Adelaide's coastal water quality (see below).

Investigations of Adelaide's coastal water quality began during the 1970's, when it became clear that seagrasses were disappearing from the Adelaide metropolitan coast and that there were water quality issues associated with Adelaide's land-based discharges. These studies were conducted by the Engineering and Water Supply Department (E&WS) who managed Adelaide's WWTP outfalls. Lewis (1975; from the E&WS) reported on the first major field study conducted off Adelaide examining spatial and temporal patterns of various water quality parameters between 1972-1975 (Appendix A). Lewis's (1975) report was followed by that of Steffensen (1985) documenting results of further water quality monitoring by the E&WS between 1976 and 1983. Steffensen (1981) and Neverauskas (1987a; also from the E&WS) reported values for various water quality parameters associated with the Port Adelaide WWTP sludge outfall during 1979 and 1984-85 (Appendix A). However, between 1985 and 1995, no systematic water quality monitoring was conducted off Adelaide. Subsequently, in February 1995, the Environment Protection Agency (EPA) commenced a water quality monitoring program that continues today. The EPA program has now produced a long-term dataset on various parameters measured at 2-4 week intervals over a 10-year period at seven nearshore jetty sites within the ACWS region (see EPA 1997a, Gaylard 2004; Appendix A). The authors of the present report examined each of the available E&WS

and EPA datasets to determine any spatial and temporal trends in water quality and any possible links they might have to seagrass loss (see Appendix A). The following four sections summarise the outcomes of these examinations as they relate to the four potential seagrass stressors of nutrients, toxicants, turbidity, and salinity.

*Nutrients*
Water quality monitoring since the 1970's has consistently shown elevated nutrient levels in Adelaide's coastal waters in locations associated with all of the major wastewater and stormwater inputs (see Appendix A). Lewis (1975) reported low nutrient concentrations offshore between Middle Beach and St Kilda, but elevated levels adjacent to Bolivar WWTP outfall (nitrate up to 0.6 mg $L^{-1}$, total phosphorus up to 0.37 mg $L^{-1}$). For the period 1972-1974, phosphorus and nitrate concentrations were highest in the region from Henley Beach to Seacliff; where the Glenelg WWTP outfall, the Patawalonga outlet, and the Torrens River outlet are all situated (Figure 1), and generally lower north and south of this area (Lewis 1975). During 1975-1983, Steffensen (1985) reported elevated nutrients adjacent to the Glenelg and Christies Beach WWTP outfalls but limited elevation of nutrients adjacent to stormwater outlets in Holdfast Bay. For the period 1995-2002, the EPA identified that ammonia, oxidized nitrogen, and total phosphorus were highest at the Glenelg, Henley Beach and Grange jetty monitoring sites, while total nitrogen was highest at the Grange and Henley Beach sites. Significantly, those three sites are the closest of the EPA's seven metropolitan jetty monitoring sites to the Glenelg WWTP outfall and the Patawalonga and Torrens River outlets (Gaylard 2004); they are also within the area that has suffered the largest amount of nearshore seagrass loss (see later). Average ammonia concentrations for the Glenelg, Henley Beach and Grange sites were around 0.04 mg N $L^{-1}$ with maxima around 0.19 mg N $L^{-1}$ (Gaylard 2004). Based on the Australian and New Zealand Environment and Conservation Council (ANZECC) water quality guidelines for ecosystem protection, Gaylard (2004) rated the three sites as moderate for both ammonia and oxidized nitrogen. Interestingly, Gaylard (2004) reported increased levels of chlorophyll-*a* (which may be a more sensitive and integrated indicator of water quality than nutrients) at sites north of Glenelg with highest concentrations at Henley Beach and Grange. Chlorophyll-*a* levels were classified as poor against the ANZECC guidelines for ecosystem protection at all sites between Largs Bay and Brighton (Gaylard 2004). Gaylard (2004) stated that there were no significant seasonal or annual trends in any of the nutrients over the period 1995-2002, but that ammonia levels had improved since the 1997 report that summarised data for the period 1995-1996 (see EPA 1997a). It is worth noting here that the ANZECC water quality guidelines of chlorophyll-*a* = 0.01 mg $L^{-1}$, total phosphorus = 0.1 mg P $L^{-1}$, filterable reactive phosphate = 0.01 mg P $L^{-1}$, total nitrogen = 1 mg N $L^{-1}$, oxidized nitrogen = 0.05 mg N $L^{-1}$, and total ammonia N = 0.05 mg N $L^{-1}$, for marine waters in south central Australia (low rainfall area), are generalised values and may not necessarily apply to the oligotrophic waters found in many parts of South Australia. Thus, previous values reported for Adelaide's coastal waters may be more of a concern than first thought.

In addition to ambient monitoring, various dispersion studies have clearly shown elevated nutrient levels immediately adjacent to the Glenelg and Christies Beach WWTP outfalls and the Port Adelaide WWTP sludge outfall (Steffensen 1985, Neverauskas 1987a). Minor increases in nutrient levels of coastal waters have also been associated with major stormwater events (Lewis 1975, Steffensen 1985).

*Toxicants*
Very few data exist on concentrations of potential seagrass toxicants in Adelaide's coastal waters (see Appendix A). Steffensen (1981) reported some heavy metals adjacent to the Port Adelaide WWTP sludge outfall during 1979. For the period 1995-2002, heavy metal concentrations were generally low off Adelaide's jetties, except for some elevated readings of nickel at Glenelg, Brighton and Port Noarlunga, and consistently high levels of zinc at all sites (Gaylard 2004). While Gaylard (2004) did not consider the elevated levels to be human-

induced, he did suggest that they could be harmful to ecosystem health. Nonetheless, heavy metals are generally not considered to be a major threat to seagrasses (Westphalen *et al.* 2004). No historical data were found on levels of herbicides or petrochemicals in Adelaide's coastal waters.

*Turbidity*
Turbidity and suspended solids readings have generally shown peak levels during winter, sometimes associated with stormwater events (see Appendix A). For the period 1995-2002, turbidity was elevated (relative to the other EPA jetty monitoring sites) at the Brighton, Glenelg and Henley Beach sites, which may be attributable to the Glenelg WWTP outfall, and the Patawalonga and Torrens River outlets (Gaylard 2004). Stormwater drains in the Brighton region may also be implicated (Gaylard 2004). Steffensen (1985) found higher turbidity values in bottom versus surface samples and suggested that re-suspension of sediments could be responsible. Personal observations by the authors indicate that re-suspension of mobilized sediments is currently a major problem during rough seas in Holdfast Bay.

*Salinity*
Historical data generally show that salinity is greater in summer than winter and spring, and that the effects of stormwater on salinity are relatively minor and localized, with lower values at the surface than the bottom (see Appendix A). These data are supported by more recent work within the ACWS (Kaempf 2005). Dispersion studies of Bolivar and Glenelg WWTP outfalls have shown very minor and localized reductions in salinity associated with effluent discharge; bottom waters show virtually no change in salinity (Lewis 1975, Steffensen 1985).

*Other parameters*
Various indicators of biological activity, including chlorophyll-*a* and enterococci, off Adelaide's coast clearly suggest an influence from land-based discharges, especially in the region between Glenelg and Henley Beach (Gaylard 2004, see Appendix A). Increased nutrients from Glenelg WWTP outfall are most likely responsible for the increased chlorophyll-*a* levels in the region (Gaylard 2004).

*Limitations in our historical and current understanding*
No water quality data are available pre-1970s when major seagrass losses occurred. Thus we can only speculate on the water quality at that time based upon subsequent data collections. Also, water quality sampling since 1970 is limited (see above). For example, while the EPA's dataset is current and is the most temporally and spatially comprehensive dataset available, it does not provide information on water quality for offshore areas of Holdfast Bay where seagrasses still occur and it does not deliberately monitor coastal water quality after stormwater events. Furthermore, the offshore water quality data collected by the E&WS in the 1970s and 1980s were specific to locations and operations at the time, such as the sludge outfalls, and do not necessarily relate to the current situation. Thus, our current understanding of coastal water quality off Adelaide is incomplete, particularly with respect to the impact of stormwater flows and the spatial extent of discharges from WWTPs.

### 1.1.4. Coastal sediment quality

Sediments along the Adelaide metropolitan coast (AMC) are derived from both marine and terrestrial sources. The quality of Adelaide's coastal sediments may be important for seagrass health as toxicants and nutrients can attach to sediment particles and/or occur in the porewater (Ralph *et al.* 2006), where they may be available for uptake at deleterious levels by seagrass roots. In addition, re-suspension of sediments may be important for seagrass health through increased turbidity and reduced light.

Relative to water quality, few data are available on coastal sediment quality for Adelaide's coastal waters. Apart from some monitoring by the E&WS of heavy metals around the Port Adelaide WWTP sludge outfall in the late-1970s (Steffensen 1981; See Appendix B), there do not appear to be any other coastal data. While sediment sampling within the Port River system has revealed the presence of some heavy metals in high concentrations (EPA 1997b, 2000, see Appendix B), the relevance of results from these studies to areas along the open metropolitan coast is unknown.

*Nutrients*
No data exist on nutrient levels in Adelaide's coastal sediments. Nutrient levels in sediments will not be investigated as part of the present report, but have been investigated as part of Task EP 1 and reported elsewhere (Collings *et al.* 2006a).

*Toxicants*
Steffensen (1981) reported very low values for heavy metals adjacent to the Port Adelaide WWTP sludge outfall where major seagrass losses have occurred (see below). A small-scale investigation of sediment toxicants along the AMC will be made as part of the present study.

*Sediment resuspension*
The composition of Adelaide's coastal sediments will influence whether resuspension of sediments and the subsequent increase in turbidity is a major issue for coastal water quality. Task PPM 1 of the ACWS investigated Adelaide's coastal sediments, while Task PPM 2 utilized hydrodynamic modeling to investigate stormwater plume dispersion and sediment resuspension. While turbidity is not addressed in the present report, light meters deployed as part of Task EP 1 (see Collings *et al.* 2006b) will provide an integrated approach to the issue of turbidity (and sediment resuspension) and its effect on the long-term survival of seagrasses in Holdfast Bay.

### 1.1.5. Seagrass quality

Seagrasses are known to be sensitive indicators of coastal water and sediment quality, with declines in nearshore water and sediment quality being associated with major seagrass declines worldwide (Ralph *et al.* 2006). Different seagrass species have varying physiological requirements and therefore varying responses to stressors. For example, *Posidonia australis* has a greater light requirement than *P. sinuosa* (Masini *et al.* 1995), which helps explain why *P. australis* is usually restricted to shallower waters than *P. sinuosa* (Clarke 1987). Fundamental structural and physiological differences exist between the two main seagrasses of concern to the ACWS; *Amphibolis* and *Posidonia*. While *Posidonia* has strap-like leaves and a large underground root/rhizome system, *Amphibolis* has a woody stem and branch system with terminal clusters of small leaves and a relatively small underground root/rhizome system (Figure 2, Lill 2005). *Posidonia* also grows basally while *Amphibolis* grows apically. Thus if we are to understand historical and ongoing seagrass losses along the AMC, we need to understand the historical status and trends of the different seagrass species.

*Assessing the quality of seagrass meadows*
The quality or 'health' of seagrass meadows has traditionally been assessed in a variety of ways for different spatial scales (e.g. see Wood and Lavery 2000; Pergent-Martini *et al.* 2005). Three spatial scales of data collection have historically been used for assessing the status of seagrass meadows along the AMC: airborne remotely sensed data, underwater remotely sensed data, and underwater *in situ* data that include field collection and later examination in the laboratory (see Appendix C for summary of published reports). Some key health indicators of meadow-forming seagrasses are:

- **Benthic coverage**. Meadows may vary from uniform to fragmented, with fragmentation being a potential indicator of degradation and losses (although sand patches and ‘blowouts’ do occur naturally in many areas and some species, such as *P. coriacea*, grow naturally in clumps over bare sediment rather than in expansive meadows). Benthic coverage can be assessed at all three spatial scales.

- **Temporal changes in seagrass coverage**. *Amphibolis* and *Posidonia* are both perennial, long-lived, slow-spreading plants for which detection of changes in broad-scale meadow coverage may take many years. Thus, substantial losses of meadow coverage, retreat of meadow edges (at both the shallow and deep extents of distribution), and occurrence of fragmentation within a short timeframe (years) are usually a cause for concern. Temporal changes in benthic coverage can be assessed at all three spatial scales.
- **Aboveground biomass**. There are several widely accepted indicators related to the amount of aboveground biomass in *Posidonia.* These include aboveground biomass, shoot density, leaf density and leaf length, with relatively high values in each of these parameters usually indicating healthy *Posidonia* meadows. For *Amphibolis*, however, equivalent indicators are lacking. For all species, quantitative estimates of aboveground biomass can be assessed either *in situ* or in the laboratory from field collections.
- **Species composition**. Some species are reportedly more sensitive than others to anthropogenic stressors (e.g. *Amphibolis* versus *Posidonia*; Shepherd *et al.* 1989) and some may take longer to recover from impacts (e.g. *Posidonia* versus *Halophila*, Bryars and Neverauskas 2004). Thus it is useful to monitor for changes in species composition. Species composition must be assessed *in situ* by divers. However, generic composition can be assessed underwater using remote video.
- **Epiphyte load and composition**. High epiphyte loads (particularly those comprised of fast-growing, opportunistic algal species) may be an indicator of unhealthy seagrass meadows due to increased levels of water column nutrients (Shepherd *et al.* 1989). Epiphyte load can be assessed underwater remotely or *in situ*, although remote video techniques allow only qualitative estimates (Gonzalez 2005). In contrast, epiphyte composition must be assessed from *in situ* observations or collections.

### ***Available data on seagrass quality***

<u>Airborne remote sensing</u>

A number of studies utilizing airborne remotely sensed data have demonstrated substantial losses of seagrasses along the AMC (see Appendix C). While previous studies using airborne remotely sensed data have provided useful large-scale assessments of seagrass meadows along the AMC across long time periods, the technologies utilised have difficulty dealing with unattached or ‘drift’ seagrass and algal material, and they have been unable to provide discrimination of seagrass species or genera (Appendix C).

<u>Underwater remote sensing</u>

There are relatively few underwater remotely sensed data for the AMC (see Appendix C). However, due to the high expenses and occupational safety issues associated with SCUBA diving, remote video techniques are being used more frequently for benthic assessments in South Australia (e.g. Cheshire *et al.* 2002, Fairhead *et al.* 2002, Marsh *et al.* 2002).

<u>*In situ* diver surveys</u>

Since the advent of SCUBA diving in South Australia during the 1960’s, *in situ* diver surveys have been conducted on numerous occasions at various sites along the AMC for fine-scale assessments of health (see Appendix C). Most of these surveys have been conducted for very specific reasons; most often associated with sewage and sludge outfalls in Holdfast Bay (see Appendix C).

*Historical status and trends of seagrass quality along the Adelaide metropolitan coastline*
Based upon information summarised in Appendix C and other sources, a reconstruction was made of the broad-scale distribution and composition of *Amphibolis* and *Posidonia* meadows in five depth classes across four zones during the early 1900s prior to any recorded losses (Figure 3). The reconstructed distribution utilizes records dating from 1935 to the present and assumes that seagrasses present at a given location anytime during that time period were also present at that location in the early 1900's; based upon the longevity of *Amphibolis* and *Posidonia* meadows (Larkum *et al.* 2006), this assumption appears valid.

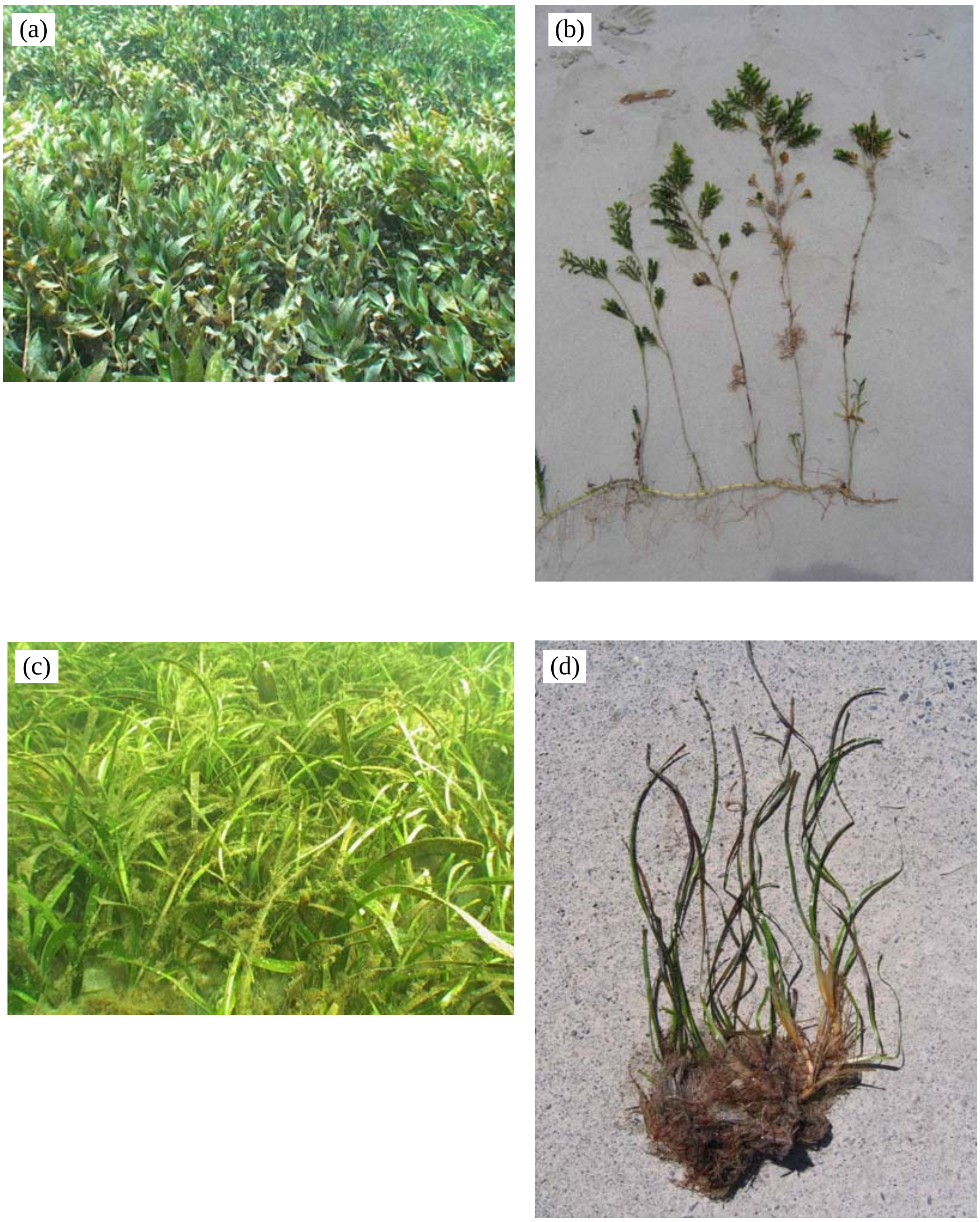


**Figure 2.** Photographs showing *Amphibolis antarctica* (a) meadow and (b) ratio of above and belowground biomass, and *Posidonia* (c) meadow and (d) ratio of above and belowground biomass. Note that the amount of belowground biomass shown for *Posidonia* is still only a fraction of the actual total amount.

The sections below summarise the historical status and trends of seagrass quality in each of the four geographical zones identified for use in Task EP 1 (see Westphalen *et al.* 2004).

Zone 1: Port Gawler to Outer Harbour (refer to Figure 1 for locations)
Declines in seagrass coverage have occurred throughout the nearshore region between Port Gawler and Section Bank since the 1960's, with substantial declines occurring adjacent to the Bolivar WWTP outfall at Fork Creek (Shepherd *et al.* 1989, Cameron 1999). Using estimates from Kinhill (1999), it is apparent that at least 300ha of subtidal seagrasses disappeared in the region between 1967 and 1997. These declines appear to be ongoing (Cameron 1999), but may have slowed during the 1990s (Kinhill 1999). Prior to the opening of the Bolivar WWTP outfall in 1967, it is thought that *A. antarctica*, *P. australis*, and *P. sinuosa* dominated the shallow subtidal area (Figure 3), while *Heterozostera tasmanica* dominated the intertidal area (Shepherd 1970, Shepherd *et al.* 1989; it must be noted that a recent taxonomic revision of the *Heterozostera* genus by Kuo (2005) now indicates that two species occur in South Australia, *H. nigricaulis* and *H. polychlamys*, but that *H. tasmanica* does not. Thus, any references in the current report to *H. tasmanica* may be referring to either *H. nigricaulis* or *H. polychlamys*). Indeed, Shepherd (1970) mapped a widespread coverage of *A. antarctica* just to the north of the Section Bank region in 1968. Yet in 2001, Cheshire *et al.* (2002) documented very small amounts of *A. antarctica* in the Section Bank/Barker Inlet area, with more extensive coverage of *Posidonia*, *H. tasmanica*, *Zostera muelleri*, and *Halophila australis*. During 1985/86, Connolly (1986) reported that *Amphibolis* was absent from the nearshore subtidal area adjacent to Bolivar WWTP outfall (where *Posidonia* still occurred), but that it was present further offshore out to 8m depth. Field surveys by consultants during the 1980s and 1990s also failed to find *A. antarctica* in the region adjacent to Bolivar WWTP (Kinhill 1999). Declines in coverage of *Amphibolis* and *Posidonia* in the nearshore of Zone 1 have been linked to increased epiphyte loads associated with elevated nutrient levels from the Bolivar WWTP outfall (Shepherd *et al.* 1989). There has been minimal surveying of offshore seagrasses in Zone 1. Neverauskas (1987a, 1988) found a mixed community of *P. angustifolia* and *P. sinuosa* at 11-12 m depth off Port Gawler and reported various measures of seagrass quality (Appendix C). From the spatial data of Tanner (2005) that fall within Zone 1, the offshore limit of *Posidonia* appears to lie between 10 and 15 m depth. In support of this, Shepherd (1970) recorded decreasing densities of *Posidonia* with increasing distance west of the Section Bank area but at locations well inside the 19 m isobath.

Zones 2 and 3: Outer Harbour to Hallett Cove (Holdfast Bay; refer to Figure 1 for locations)
Major declines in seagrass coverage in Holdfast Bay commenced sometime prior to 1949 (Freeman 1982). Declines in seagrass coverage have occurred predominantly (1) in the nearshore region with a seaward regression of the inshore meadow edge or 'blue-line' between Outer Harbour and Brighton, (2) at several clearly-defined offshore locations associated with the Port Adelaide and Glenelg WWTP sludge outfalls, and (3) as 'fragmentation' in the offshore parts of central and southern Holdfast Bay (Freeman 1982, Lewis 1975, Neverauskas 1985, Steffensen 1985, Clarke and Kirkman 1989, Hart 1997, Thomas and Clarke 2002, Cameron 1999, 2003). While nearshore declines in coverage appear to have stabilized and the sludge outfalls no longer operate, offshore fragmentation appears to be ongoing (Hart 1997, Cameron 2003). Selective decline in coverage of *Amphibolis* over *Posidonia* has been documented in several places (Lewis 1975, Neverauskas 1985). Prior to recorded declines, seagrass coverage in Zones 2 and 3 was apparently dominated by *A. antarctica*, *P. angustifolia*, and *P. sinuosa* (Lewis 1975, Shepherd and Sprigg 1976, Johnson 1981, Steffensen 1981, Clarke 1987, Sergeev *et al.* 1988, Clarke and Kirkman 1989, Thomas and Clarke 2002). In 1961, *Amphibolis* was the dominant seagrass species in Holdfast Bay between Henley Beach and Glenelg north (Lewis 1975). Diver surveys during the 1960s, 1970s and 1980s consistently reported both *Amphibolis* and *Posidonia* in Holdfast Bay from shallow waters out to ca. 12 m depth (e.g. Shepherd 1970, Shepherd and Sprigg 1976, Lewis 1975, Johnson 1981, Steffensen 1981,

Clarke 1987, Sergeev *et al.* 1988, Thomas and Clarke 2002, Cheshire and Miller 1996), although Clarke and Kirkman (1989) found very few localities with *Amphibolis* in Holdfast Bay during their surveys of blowouts during the early 1980's. In 2004, Tanner (2004) reported the presence of both *Amphibolis* and *Posidonia* in ca. 9-12 m depth adjacent to the Outer Harbour shipping channel in the northern part of Zone 2. *Heterozostera tasmanica*, *Halophila australis*, *P. australis*, and *A. griffithii* also occur in Zones 2 and 3 (Johnson 1981, Steffensen 1981, 1985, Thomas and Clarke 2002, Simon Bryars, personal observation).

Underwater excavations of old seagrass beds during 1973 (as well as numerous anecdotal accounts) revealed that seagrasses once grew only 200m from shore throughout Holdfast Bay (Lewis 1975). In support of this conclusion are the first aerial photographs of seagrass coverage in Zones 2 and 3, which show that in 1935, the blue-line was only 200-300m from shore at Glenelg and Henley Beach (Freeman 1982, Steffensen 1985). At the time, Cleland (1935) stated "The coast-line from Outer Harbour to Marino consists of very gradually shelving sands finally succeeded by the blue line where the water deepens and the yellow of the sand is replaced by a dark blue due to the under-water meadows of *Cymodocea* and *Posidonia*." As *Amphibolis* was formerly placed within the genus *Cymodocea* (Ducker *et al.* 1977), Cleland's (1935) statement confirms that *Amphibolis* occurred nearshore between Outer Harbour and Marino prior to the major declines post-1935. In 1968, when the blue-line had already regressed hundreds of metres offshore in the Glenelg to Henley Beach region, both *Amphibolis* and *Posidonia* constituted the nearshore community between Outer Harbour and Marino (Shepherd 1970). Even as late as 1977, Steffensen (1981) reported *Amphibolis* to be the dominant seagrass species in the nearshore area adjacent to Point Malcolm in northern Holdfast Bay. It is quite clear then that *Amphibolis* formed an integral part of the nearshore seagrass community in Holdfast Bay prior to any observed losses. Observations in other parts of Gulf St Vincent and southern Australia also show that *Amphibolis* often forms a fringing border on the shoreward side of *Posidonia* meadows (Cambridge 1975, Lewis 1975, Shepherd and Sprigg 1976, Simon Bryars, personal observation), possibly due to *Amphibolis* having a greater tolerance to rough conditions than *Posidonia* (Shepherd 1970). The historical dominance of *Amphibolis* in Adelaide's nearshore seagrass community may be critical to determining a proposed mechanism of initial seagrass decline off the Adelaide metropolitan coast.

In a benthic survey conducted in 1985, Thomas and Clarke (2002) reported *P. australis* from the inshore margin of seagrass meadows at Largs Bay in northern Holdfast Bay (Figure 1). Given that the blue-line had already regressed about 200 m in this region between 1935 and 1981 (Freeman 1982), combined with what is known about the high light requirements and shallow distribution of *P. australis*, it is highly likely that *P. australis* was originally common along the nearshore margin of Largs Bay. It is unknown whether *P. australis* once occurred in the higher energy shallows of central and southern Holdfast Bay, but based on the extremely shallow location of the blue-line in 1935 and the presence of *P. australis* in Zones 1 and 4 to the north and south of Holdfast Bay, it is likely that it did form at least part of the nearshore community throughout Holdfast Bay.

Some data exist on the offshore depth limits of *Posidonia*/*Amphibolis* meadows in Zones 2 and 3. Shepherd and Sprigg (1976) reported that during the 1960s, the depth limit of *A. antarctica* in Holdfast Bay was ca. 12m, but that *Posidonia* extended to ca. 15 m. Neverauskas (1987a) found that during the early 1980s, *Posidonia* was present at 16.5 m but absent at 18 m depth in Zone 3, while Thomas and Clarke (2002) also recorded an outer depth limit of 18 m for *Posidonia* in Holdfast Bay during 1985. Limited spatial data from Tanner (2005) indicates a lower depth limit around 15 m for *Posidonia* in Zones 2 and 3. Thus it appears the lower depth limit for *Posidonia* is around 15-18 m in Holdfast Bay. However, there is insufficient historical data to determine if the offshore margin has been regressing shoreward in conjunction with the observed seaward regression of the nearshore margin in Holdfast Bay over the past 60 or so years.

While various *in situ* measures of seagrass quality have been made in Holdfast Bay since the 1960's (see Appendix C), it is difficult to determine what constitutes a healthy or good quality seagrass meadow due to variation associated with alongshore locality, season, and depth. There are also very few localities where measures of seagrass quality have been made over time (see Appendix C), such that temporal trends can be ascertained. Shepherd (1970) reported what he thought were degraded *Posidonia* meadows adjacent to north Glenelg in 1968, with aboveground biomass of 100 g wet wt. $m^{-2}$, increased epiphyte loads (not quantified), and leaf lengths rarely reaching 30 cm. Patterns of *Posidonia* aboveground biomass related to depth appear to vary (Shepherd 1970, Lewis 1975, Neverauskas 1987a). While Shepherd (1970) reported a decline in aboveground biomass with depth from 5 to 15 m, Lewis (1975) reported lowest mean values at sites 500 m from shore between Outer Harbour and Marino during 1972-1975, intermediate values at 1000 m offshore, and highest values at sites > 1000 m offshore (Appendix C). Neverauskas (1987a) reported highest aboveground biomass at 10 m depth along a depth gradient from 0 to 18 m (Appendix C). In an attempt to determine the current status and any possible trends in seagrass quality in Holdfast Bay, further analyses of previous *in situ* measures of seagrass quality (Appendix C) will be made in the results and discussion sections of this report. In all previous *in situ* studies of seagrass quality along the AMC, very little information has been documented for *Amphibolis* (Appendix C).

Zone 4: Hallett Cove to Sellicks Beach (refer to Figure 1 for locations)

While no declines in seagrass coverage have been documented for Zone 4, very little information is available to inform this debate. The nearshore region of Zone 4 is dominated by reef and sandy bottom habitats rather than seagrass meadows (Edyvane 1999, Bryars 2003), however, remote sensing work indicates that seagrasses are prevalent in the offshore waters of Zone 4, particularly off Aldinga Bay (Cameron 1999, Edyvane 1999). Furthermore, *Posidonia* meadows are known to occur off Christies Beach in 15-18 m depth (Cheshire and Miller 1999), and Lewis (1975) reported both *Amphibolis* and *Posidonia* from ca. 7 m depth within a 1 km radius of Christies Beach WWTP outfall during 1972-1975. A subsequent survey in 1978 failed to find any seagrasses around the Christies Beach WWTP outfall (Steffensen 1985), which may suggest some seagrass decline. *Posidonia coriacea*, which grows in discrete patches rather than extensive meadows, occurs adjacent to Sellicks Beach in Aldinga Bay (Figure 4, Edyvane 1999) and is an uncommon habitat type within the ACWS region. *Posidonia australis* has been recorded from the southern end of Aldinga Reef in 0-1 m water (Robertson 1984). Kuo and Cambridge (1984) reported herbarium specimens of *P. coriacea* from Port Stanvac (7 m) and Marino; the latter record is probably near the northern limit of this species in eastern Gulf St Vincent, as it prefers high wave energy environments that are lacking north of Marino. It is possible that species other than *P. coriacea* within the *P. ostenfeldii* species complex also occur in Zone 4.

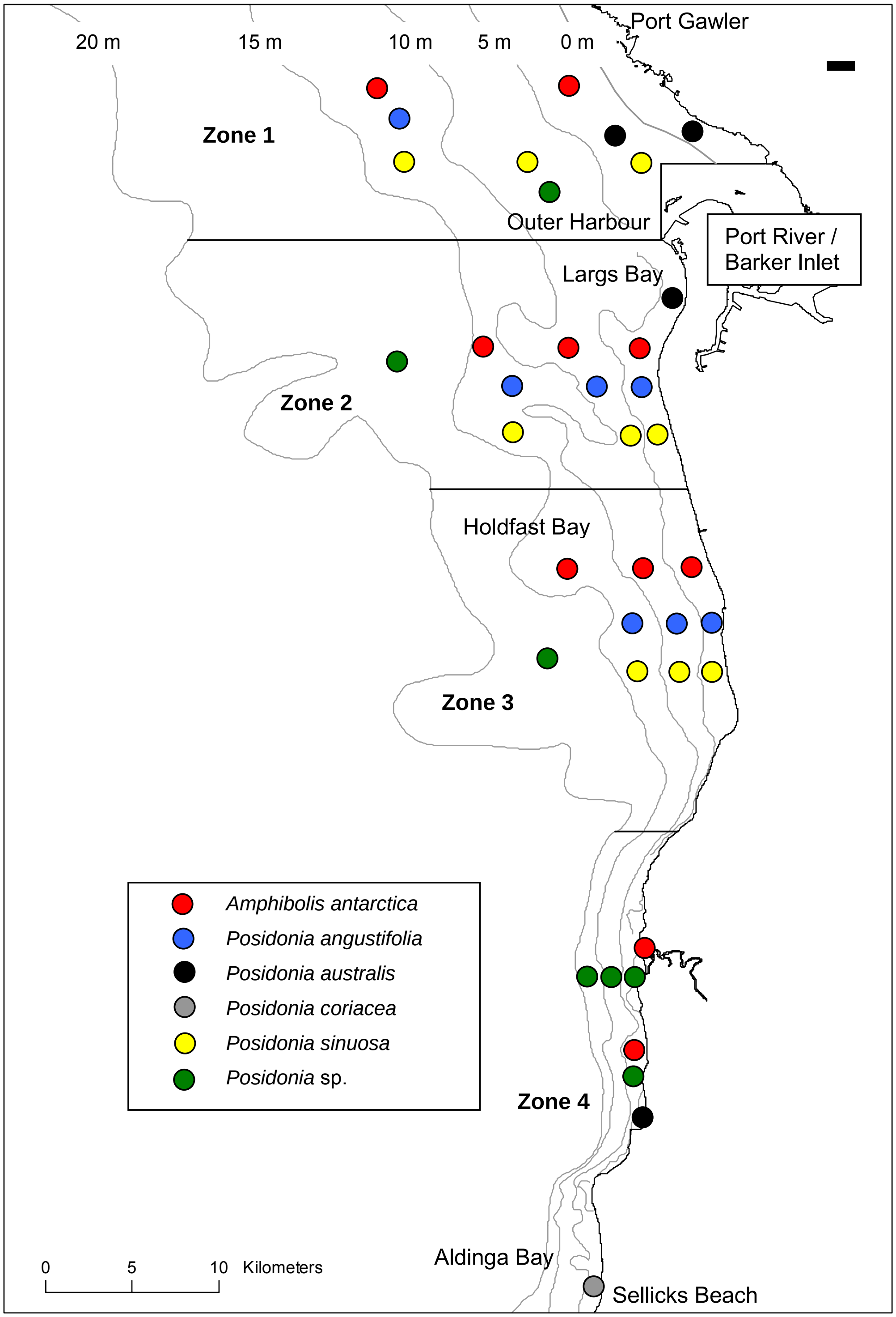


**Figure 3.** Reconstructed distribution of the main meadow-forming species of *Amphibolis* and *Posidonia*, along the Adelaide metropolitan coastline between Port Gawler and Sellicks Beach during the early 1900's prior to any losses being recorded. Distributions are based on published reports available to the authors at the time of writing. See text for further details. Presence of coloured circles indicates the occurrence of a species within Zones 1-4 across five depth intervals of <0 (intertidal), 0-5, 5-10, 10-15, and 15-20 m. Zones are based on Westphalen *et al.* (2004). Note that species of *Halophila*, *Heterozostera*, and *Zostera* are not shown.

**Figure 4.** Aerial photograph of Sellicks Beach showing discrete patches of *Posidonia coriacea*. Photo taken in 1997 at 1:10000 and supplied courtesy of SA Department of Environment and Heritage. Ground-truthing information supplied by Hugh Kirkman.

## 1.2. Needs and objectives

### 1.2.1. Coastal water quality

While some current data are available on nearshore levels of some potential stressors, there are currently no data being collected from offshore waters where seagrasses occur and no data collection in the marine environment that is aligned to times when we would expect peak levels of stressors from stormwater. Historically, there is also a lack of information for the offshore waters along the AMC. Thus there is a real need to collect data on current levels of potential stressors where seagrasses occur and at times of peak stormwater discharge. This information will allow a comparison with historical offshore levels, with current and historical nearshore levels, and with accepted levels for ecosystem health, determination of peak levels, and an assessment of the possible need to experimentally investigate the various stressors within Task EP 1. In addition, recent advances in stable isotope research now enable the detection in seagrasses of nitrogen derived from wastewater, such that the spatial influence of wastewater discharges can be determined (see below). Thus the specific objectives of this part of the study were to determine:

- Ambient offshore water quality within Zones 1-4.
- Extremes of water quality within Zones 2 and 3 associated with major rainfall events.
- The spatial influence of nutrients derived from wastewater treatment plants and industrial discharges in Gulf St Vincent.

Water quality parameters considered useful for investigation included, nutrients, chlorophyll-*a*, faecal coliforms, *E. coli*, dissolved organic carbon, and several potential toxicants: organochlorine and organophosphate pesticides, triazine herbicides, and the herbicide Glyphosate (see Westphalen *et al.* 2004, for review of potential effects of these stressors on seagrasses). A large-scale survey was also conducted using the isotopic signature of seagrasses to determine the spatial extent of nitrogen derived from WWTPs and other sources in Gulf St Vincent. Neither turbidity nor light attenuation were measured during the surveys, as a series of light loggers were deployed to continuously measure underwater light intensity over a 12-month period as part of Task EP 1 (see Collings *et al.* 2006b). Salinity was also not measured because historical and ACWS data clearly indicate that decreases in salinity due to stormwater and wastewater are minimal (see Kaempf 2005, and Appendix A). Subsequent experimentation has also shown minimal effects of reduced salinity on adult *Amphibolis* and *Posidonia* (Westphalen *et al.* 2005).

*Stable isotopes*

Employing traditional water quality measurements to determine the significance of nitrogen inputs in an ecosystem provides limited temporal information and can be hampered by rapid dilution of point sources in surrounding waters and through its prompt incorporation into plant material (Lepoint *et al.* 2004; EPA 2005). Seagrasses are continuously utilising the surrounding dissolved inorganic nitrogen (DIN) pool during photosynthesis, which is incorporated into biomass. When combined with their sessile nature and long lifetime, this makes seagrass an ideal bioindicator (Jones *et al.* 2001; Lepoint *et al.* 2004). As a result, these plants have been used extensively in nitrogen stable isotope studies around the globe to source and profile the movement of anthropogenic nitrogen in coastal waters (McClelland *et al.* 1997, Constanzo *et al.* 2001, Lepoint *et al.* 2004). These studies utilise the variation in $\delta^{15}N$ in naturally occurring seagrass communities as a bioindicator of $\delta^{15}N$ in the surrounding available environmental DIN, with particular development having been made in utilising seagrass to detect and map the geographical extent of biologically available nitrogen derived from sewage (Costanzo *et al.* 2001, Jones *et al.* 2001). The main goal of these studies is to detect an environmental impact from a particular source before widespread changes in water quality and community structure arise (Jones *et al.* 2001).

McClelland *et al.* (1997) and McClelland and Valiela (1998) were the first to show the correlation between $\delta^{15}N$ values in seagrass and wastewater nitrogen sources. They showed an increase in the $\delta^{15}N$ of the eelgrass *Zostera marina* from -1‰ to +6‰ with increasing wastewater input in the Waquoit Bay estuary, Massachusetts, USA. Udy and Dennison (1997) also found a correlation between source inputs and $\delta^{15}N$ values of seagrasses in Moreton Bay, Queensland, Australia. These authors found highest $\delta^{15}N$ values (9.3‰) for *Zostera capricorni* meadows receiving effluents directly from a sewage treatment plant, and lowest levels at sites remote from anthropogenic nutrient sources (2.4‰). Since this time, numerous studies have shown similar results, with seagrasses in the vicinity of sewage sources displaying elevated $\delta^{15}N$, generally in the range of 5-10‰, in contrast to background levels of approximately -2‰ to +3‰ (Jones *et al.* 2001, Costanzo *et al.* 2001,Yamamuro *et al.* 2003). One exception to this general trend can be seen in the work of Fourqurean *et al.* (1997), in which enriched (12‰) $\delta^{15}N$ values were found during summer in seagrasses located in the pristine Tomales Bay, California, USA. At this location anthropogenic inputs were considered to be minor, with oceanic nitrogen forming the dominant dissolved inorganic nitrogen source to the estuary and the high $\delta^{15}N$ was attributed to natural denitrification in the upper estuary, said to result from limited nitrogen influx from adjacent oceanic waters during summer months.

Source assignment using $\delta^{15}N$ has been strengthened from the results of a number of studies illustrating that $\delta^{15}N$ in seagrass appears to be dictated by source nitrogen over and above the influence of inter and intra specific plant variation and environmental conditions such as light intensity (Grice *et al.* 1996, Yamamuro *et al.* 2003, Guest *et al.* 2004). However, intra-annual variability can be significant, with a study by Vizzini *et al.* (2003) finding seasonal variation in $\delta^{15}N$ as high as 2.5‰ in *Posidonia oceanica* collected from Mediterranean meadows in Italy. Levels were lowest in spring and highest in winter, and were said to reflect the increased accumulation of nitrogen into the biomass afforded through increased DIN during winter.

Costanzo *et al.* (2001) were the first to categorise the variation in seagrass $\delta^{15}N$ within a given ecosystem in order to map the distribution of sewage being received by the environment. In their study, conducted in Moreton Bay, they developed a framework using marine plants as bioindicators to map sewage impacts using $\delta^{15}N$. Spatial distributions of $\delta^{15}N$ were developed throughout the bay using both seagrass and macroalgae, classifying $\delta^{15}N \leq 3‰$ as being representative of 'natural' oceanic influenced levels (i.e. background) and categories of 3-4, 4-5, 5-7, 7-9, 9+ ‰ to represent increasing impact from sewage. Values were highest adjacent to (8.7‰) and south of (9.3‰) the Brisbane River, while plants collected from the oceanic influenced sites always had levels considered background (~2.5‰). The maps developed in this study guided nutrient reduction strategies undertaken in the region and allowed for continuous monitoring during strategy implementation. The success of this approach can be seen in a review undertaken by Costanzo *et al.* (2005), in which the red macroalgae *Catenella nipae* was incubated *in situ* at various locations within Moreton Bay as part of the annual environmental monitoring developed in their 2001 study. Large investments in nitrogen removal from sewage effluents had resulted in a dramatic reduction in $\delta^{15}N$ levels throughout the bay, areas adjacent to the Brisbane River dropped from 7-9+ ‰ to 3-4‰, and the majority of the bay has become defined by levels considered background. This program demonstrates the value of this technique in providing a continuous feedback mechanism to drive management.

### 1.2.2. Coastal sediment quality

While it is unlikely that toxicants would have been responsible for broad-scale seagrass losses off Adelaide (Westphalen *et al.* 2004), it is still worthwhile to do some limited investigations of current levels of toxicants in Adelaide's coastal sediments. Information on current levels of potential stressors within sediments will enable a comparison with historical levels, a comparison with accepted levels for ecosystem health, and allow an assessment of the possible need to experimentally investigate the various stressors within Task EP 1. Thus the specific objectives of this part of the study were to determine:

- Ambient levels of sediment quality across Zones 1-4.
- Extreme levels of sediment quality associated with major stormwater discharges in Zone 3.

Potential toxicants considered useful for investigation included organochlorine and organophosphate pesticides, triazine herbicides, the herbicide Glyphosate, polyaromatic hydrocarbons, total petroleum hydrocarbons, and heavy metals (see Westphalen *et al.* 2004 for review of potential effects of these stressors on seagrasses). Ambient nutrient levels in Adelaide's coastal sediments will be investigated with other work within Task EP 1 of the ACWS (see Nayar *et al.* 2006).

### 1.2.3. Seagrass quality

Considerable previous work has been conducted along the AMC using airborne remotely sensed data to assess seagrass quality (Appendix C). However, the most recent study by Cameron (2003) utilized 2002 aerial photography and an updated 'snapshot' of the ACWS region is therefore required. Task RS 1 of the ACWS, using advanced techniques, aims to provide an updated seagrass map of the AMC including species and generic discrimination, and to revisit the historical aerial photographs. Thus airborne remotely sensed techniques do not form part of the present study.

Considerable previous work has also been conducted along the AMC using *in situ* techniques to assess seagrass quality. Nonetheless, as much of this work was conducted during the 1980s, it is timely to collect *in situ* information on broad-scale spatial patterns of seagrass coverage and composition, as well as *in situ* information on small-scale spatial patterns of seagrass habitat structure, aboveground biomass and associated epiphyte communities. This survey information will allow comparison with historical information, provide a picture of the current situation, allow comparison with the published literature, contribute to nutrient flux estimates for the AMC (see Nayar *et al.* 2006), and enable refinement of sampling strategies for field experiments within Task EP 1 (e.g. Collings *et al.* 2006a) and any ongoing monitoring programs within Task EMP 1 (see Henderson *et al.* 2006). Thus the specific objectives of this part of the study were to determine:

- The typical composition of existing seagrass meadows across Zones 1-4.
- The quality of existing meadows in Zones 1-4.
- The outer depth limit of *Posidonia* meadows in Zones 2 and 3.
- The cover and taxonomic composition of epiphytes on existing meadows.

It must be noted that the emphasis of our work was on the dominant subtidal genera of *Amphibolis* and *Posidonia*. However, some work was done on *Heterozostera* in the intertidal region of Zone 1. Assessments were also deliberately conducted in areas away from obvious land-based discharges in order to provide an indication of seagrass quality representative of the whole AMC.

# 2. Methods

## 2.1. Water quality

### 2.1.1. Ambient surveys

Nine sites spread across the 4 Zones were chosen as being representative of 'ambient' conditions within the ACWS study region (Figure 5). In each zone there was a deep (10 m depth) and shallow (5 m depth) site located over seagrass meadows, plus an additional inshore site over seagrasses in the lower intertidal part of Zone 1. None of the nine sites was directly adjacent to a major land-based discharge. At each site, two stations approximately 200-300 m apart were designated for sampling.

Four ambient surveys were conducted over a number of days during each of the four seasons: summer (3, 15, 16 December 2003), autumn (10, 12, 16 March 2004), winter (27, 28 May 2004, 2 June 2004), and spring (21, 24 September 2004). Survey dates did not coincide with major rainfall events (Figures 6 and 7). Water samples were collected on the four surveys to measure levels of a number of parameters (Table 1). The design of each survey varied according to emerging priorities and direction within the study, with some parameters being excluded after the summer survey and the number of replicate samples and stations varying between surveys (Table 1). Sampling of faecal coliforms, *E. coli*, and dissolved organic carbon was not undertaken after the summer survey as it was felt that they were not contributing towards an understanding of possible links between water and seagrass quality. The number of replicates and stations varied between surveys based on advice from Task EMP 1 within the ACWS and because little variation was found between stations within sites on the summer survey.

**Table 1.** Water sampling regime for ambient surveys off the Adelaide metropolitan coast. S = summer; A = autumn; W = winter; Sp = spring; Site codes are as shown on Figure 5 with A and B signifying the 2 stations at each site. Numbers within cells indicate the number of replicate samples collected.

| | Nutrients (TN, TAN, $NO_x$, TP)* | | | | Chlorophyll-*a* | | | | Faecal coliforms & *E. coli* | | | | Dissolved organic carbon | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Site | S | A | W | Sp | S | A | W | Sp | S | A | W | Sp | S | A | W | Sp |
| 1IA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 1IB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |
| 1SA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 1SB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |
| 1DA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 1DB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |
| 2SA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 2SB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |
| 2DA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 2DB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |
| 3SA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 3SB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |
| 3DA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 3DB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |
| 4SA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 4SB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |
| 4DA | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | | | | 3 | | | |
| 4DB | 3 | | 1 | 1 | 3 | | | | 3 | | | | 3 | | | |

* TN = total nitrogen, TAN = Total ammonia N, $NO_x$ = oxidized nitrogen, TP = total phosphorus

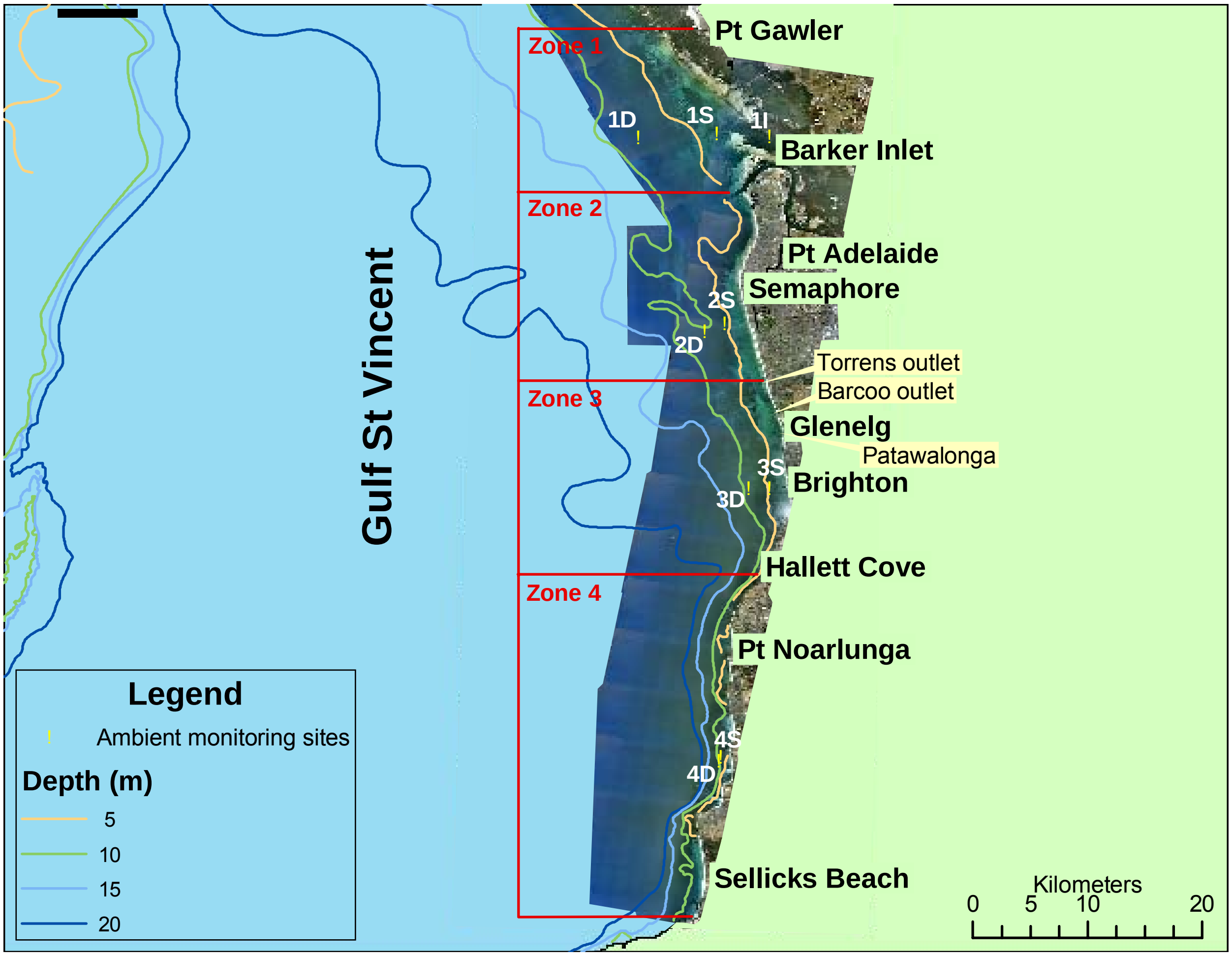


**Figure 5.** Map of the Adelaide Coastal Waters Study region showing the nine ambient sampling sites used for measuring water, sediment and seagrass quality within the four zones. Site codes signify the zone (1-4) and depth (deep, D; shallow, S; inshore, I).

*Faecal coliforms and* Escherichia coli

100 ml water samples were collected at the surface in sterile plastic containers, kept on ice in darkness, and delivered the same day to the Department of Environmental Health, Flinders University. Standard analyses were performed using Colilert®-18 for faecal coliforms and *E. coli.*

*Nutrients*

Water samples were collected from approximately 1 m above seagrass meadows using a Niskin bottle. Each replicate sample at a station was taken from a separate Niskin sample. Samples were filtered (0.45 µm) into 50 ml plastic containers and placed on ice in darkness in the field. Samples were then transferred to a -30° C freezer once ashore. Samples were analysed within one month of freezing for dissolved concentrations of total phosphorus (TP), total nitrogen (TN), total ammonia N (TAN) which includes ammonia and ammonium, nitrate plus nitrite or oxidized nitrogen ($NO_x$), and dissolved organic carbon (DOC) at the Water Studies Centre, Monash University, Victoria using flow injection analysis (FIA) on a QuickChem 8000 Automated Ion Analyser. Limits of detection for each nutrient were: TP = 0.005 mg P $L^{-1}$; TN = 0.02mg N $L^{-1}$; TAN = 0.001mg N $L^{-1}$; $NO_x$, = 0.001mg N $L^{-1}$, and DOC = 1 mg $L^{-1}$. Tests for TP, TAN and $NO_x$ are NATA accredited.

*Chlorophyll-*a

Water samples were collected within 1 m of the water surface. Water samples were collected with a Niskin bottle and transferred to 1.25 L plastic bottles. The samples were then placed on ice in darkness. Upon returning to the laboratory, 500 ml of the water was vacuum filtered using muffled (pre-combusted at 450°C overnight), Whatman GF/F glass fibre filters (0.7 µm pore size). Filters were then wrapped in aluminium foil and stored in a -80°C freezer until extraction. Filters were transferred to labelled test tubes containing 5 ml of methanol ($CH_3OH$) and refrigerated for 24 hours. The filters were removed from the test tube and the resultant extract was then centrifuged for 10 minutes at 3000 rpm. Spectrophotometer readings were then taken on a Hitachi U2000 spectrophotometer at 750 and 665 nm. Absorbance was converted to chlorophyll-*a* concentration (µg $L^{-1}$) as: [(Absorbance at 665 – Absorbance at 750) x 139]. This technique had a detection limit of 0.1 µg $L^{-1}$.

*Data treatment*

Values below detectable limits were treated as zero for data plotting and analyses. Mean values were calculated by pooling all values from the two stations for each site. In some cases, univariate statistical analyses were performed on the data to test for differences between sites and seasons. Assumptions implicit in these tests were generally met. Nonetheless, the objective of this part of the study was not to identify differences, but rather to characterize ambient water quality conditions.

### 2.1.2. Rainfall event surveys

Four rainfall event surveys were conducted: March, May, and June 2004, and June 2005. The timing of sampling coincided with significant downpours or extended periods of heavy rainfall, often after extended periods without rain (Figures 6 and 7). It was anticipated that these would be the most likely times to detect peak levels of contaminants and nutrients that had accumulated on land, as heavy downpours and the first rains after dry periods result in rapid surface run-off in urban catchments (Wilkinson *et al.* 2005a). The March 2004 survey was conducted after the first significant post-summer rains of 15.8 mm on 29 March (Figure 6). The May 2004 survey coincided with a major rainfall event of 25.8 mm on 18 May, while the June 2004 survey was conducted following several days of steady rainfall (total of 20.2 mm from 2-9 June; Figure 6). The June 2005 survey coincided with 36.4 mm of rain on 10-11 June (Figure 6). At the time of sampling on all four surveys, significant flows of water from the Torrens River were observed entering the marine environment and for the time period when flow data were available, significant flows were recorded in the Torrens River at those same times (Figure 7).

A number of water samples were collected at various sites in and around the Torrens River on each of the rainfall event surveys. Sample sites were chosen to detect 'peak' levels of various contaminants and nutrients within Zones 2 and 3 that might be associated with major rainfall events and thus discharges to the coast. Sample sites included several shore-based locations; in the Torrens River just above the weir on Military Road (TR), in the Torrens River estuary below the weir (TRE), and at five locations north and south of the Torrens River estuary (HB, NT, ST, BO, BW), as well as several ocean-based locations offshore from the Torrens River estuary (Figure 8). The nine ocean sites included four of the ambient sites (2S, 2D, 3S, 3D; Figure 5) to make possible direct comparisons between ambient and potentially peak conditions. The three inshore sites (2I, TI, 3I) were located over areas of bare sand from where seagrasses have been lost (Figure 8). The remaining two sites were located offshore from the Torrens River (TS, TD) over seagrass meadows in ~5 and ~10 m depths (Figure 8).

*Nutrients*

Water samples were collected for nutrient analyses on the March, May, and June 2004 surveys (Table 2). Nutrient samples were collected and analysed using the same methodologies as for the ambient surveys. During the March and June 2004 surveys a first set of samples was collected on a single day from the shore-based sites, followed by a second set of samples from the ocean sites on the following day when weather permitted boating (Table 2). In May 2004, water samples were not collected from shore-based sites, but two sets of samples were collected two days apart from the ocean sites (Table 2).

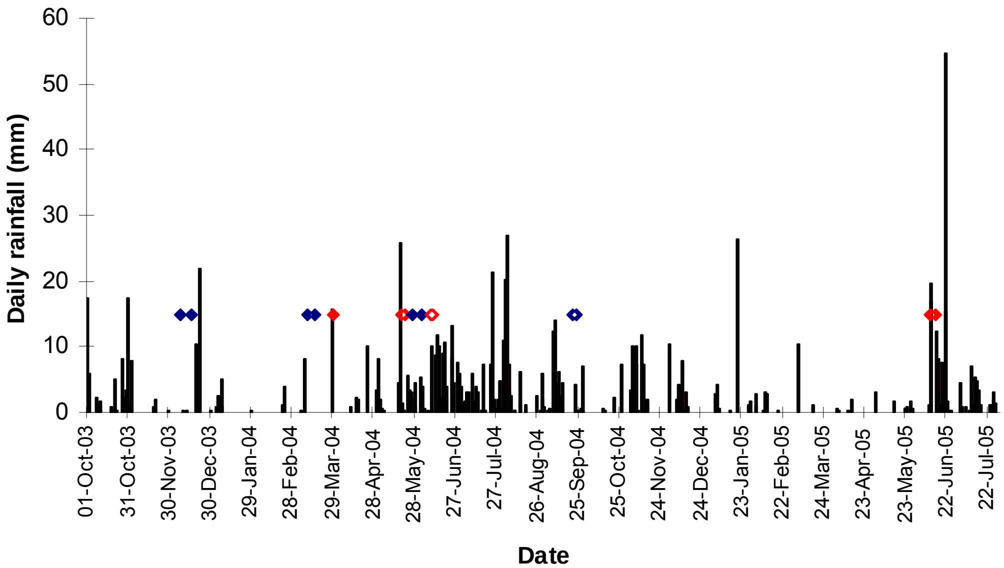


**Figure 6.** Total daily rainfall at Kent Town (Adelaide) for the period October 2003 to July 2005. Blue diamonds represent ambient survey dates; Red diamonds represent rainfall event survey dates. Data collected from the Kent Town weather station, approximately 2 km east of Adelaide city centre. Data courtesy of the Australian Government Bureau of Meteorology.

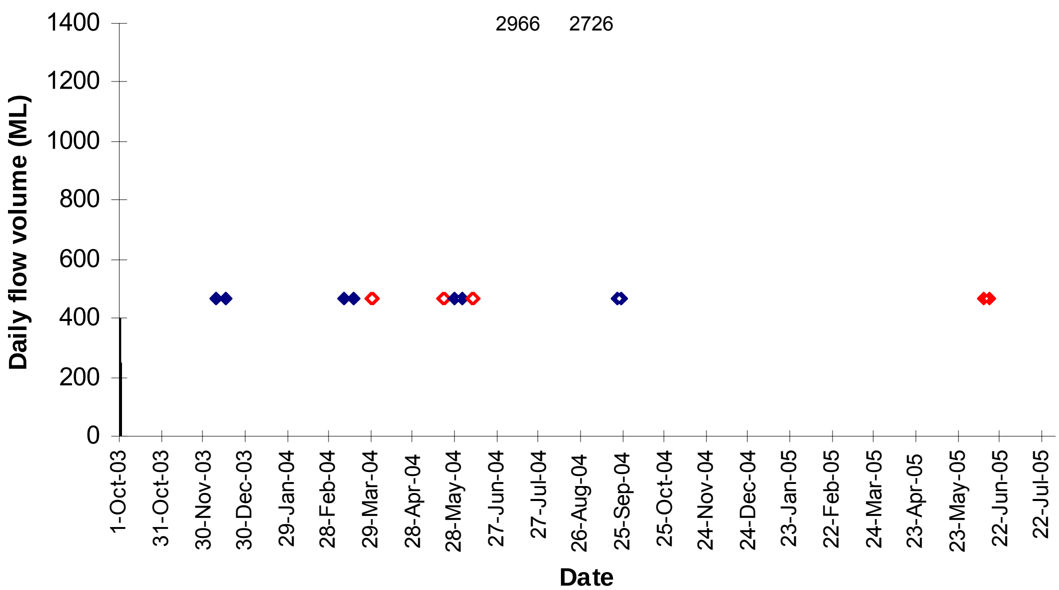


**Figure 7.** Total daily flow volume (ML) through the Torrens River gauging station at Holbrooks Road for the period 1 October 2003 to 31 July 2005. Blue diamonds represent ambient survey dates; Red diamonds represent rainfall event survey dates. Holbrooks Road is located downstream of the Torrens Lake and gives a direct indication of flows reaching the coast. Unpublished data (2006) courtesy of the Department of Water, Land and Biodiversity Conservation.

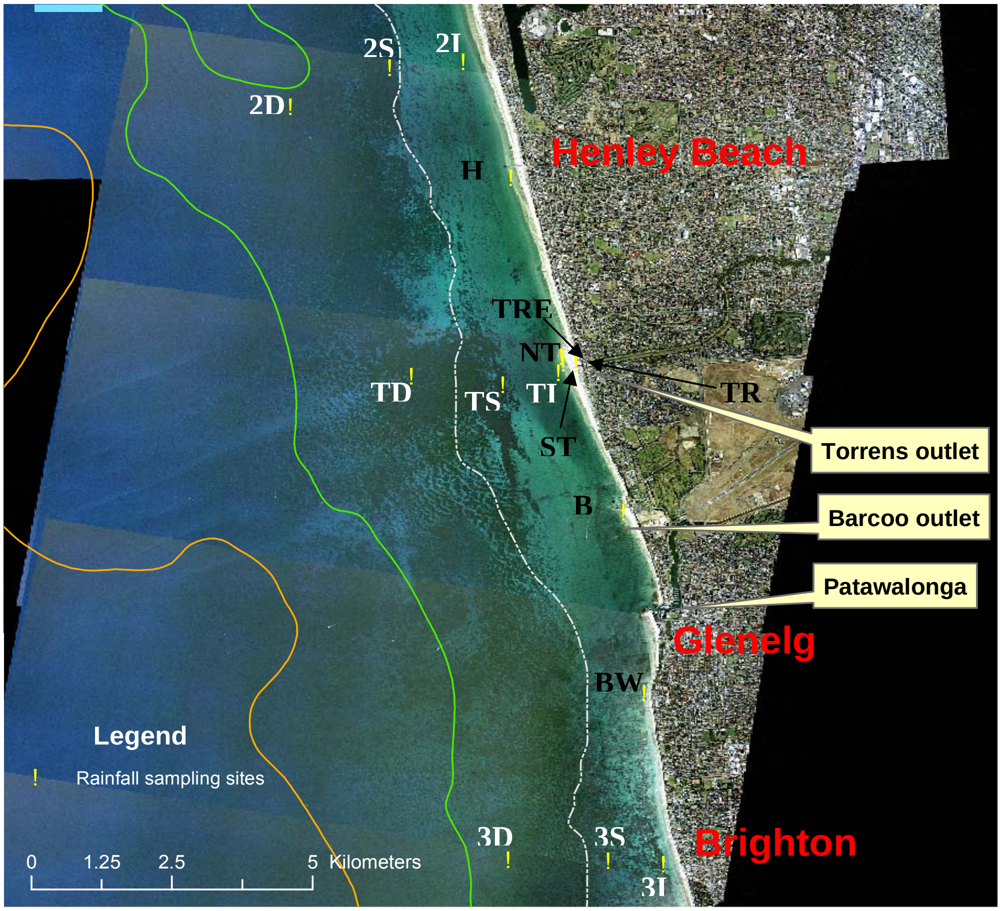


**Figure 8.** Map of Adelaide coast showing the rainfall event sampling sites used for water quality monitoring. Site labels in white (2I, 2S, 2D, TI, TS, TD, 3I, 3S, 3D) indicate ocean sampling sites. Site labels in black (HB, NT, TR, TRE, ST, BO, BW) indicate shore-based sampling sites. On the underlying aerial photograph, note the remaining seagrass meadows that appear as dark shading.

**Table 2.** Rainfall event survey dates for water samples collected for dissolved nutrient analyses.

| Site | Rainfall survey March 2004 | | May 2004 | | June 2004 | |
|---|---|---|---|---|---|---|
| | 29-3-04 | 30-3-04 | 19-5-04 | 21-5-04 | 9-6-04 | 10-6-04 |
| HB | 2 | | | | 2 | |
| NT | 2 | | | | 2 | |
| TRE | 2 | | | | 2 | |
| ST | 2 | | | | 2 | |
| BO | 2 | | | | 2 | |
| BW | 2 | | | | 2 | |
| 2I | | 3 | | | | 2 |
| 2S | | 3 | | 1 | | 2 |
| 2D | | 3 | | 1 | | 2 |
| TI | | | 1 | 1 | | 2 |
| TS | | 3 | 1 | 1 | | 2 |
| TD | | | 1 | 1 | | 2 |
| 3I | | 3 | | | | 2 |
| 3S | | 3 | | | | 2 |
| 3D | | 3 | | | | 2 |

See Figure 8 for codes and location of sample sites. Numbers within cells indicate the number of replicate samples collected.

Results from the rainfall surveys were compared against results from the ambient surveys for Zones 2 and 3 (Sites 2S, 2D, 3S, 3D), the ANZECC guidelines, and the EPA jetty monitoring data from Henley Beach, Glenelg and Brighton (Gaylard 2003). In analyzing the rainfall nutrient data it must be acknowledged that we did not collect comparable ambient data for the shore-based and inshore ocean sites. However, the EPA dataset provides long-term values for jetties at Henley Beach, Glenelg and Brighton which are very close to shore and which can be used for comparisons.

*Toxicants*

Water samples were collected for pesticide/herbicide analyses on the May and June 2004, and June 2005 surveys (Table 3). Pesticide/herbicide samples were collected from the surface waters in sterilized glass bottles (1 L) and kept on ice in the field. Bottles were placed in a freezer (-30° C) in the laboratory until they were sent away for analysis within two months of collection. Samples were analysed for a suite of organochlorine and organophosphate pesticides, a suite of triazine herbicides, and the herbicide Glyphosate (see Appendix D and E for detailed lists of compounds that were analysed). Analyses were conducted by the Australian Government Analytical Laboratories (and in some cases outsourced by them) using NATA-accredited testing procedures.

**Table 3.** Rainfall event survey dates for water samples collected for analyses of organochlorine pesticides, organophosphate pesticides, triazine herbicides, and Glyphosate. Samples collected on 10-6-05 and 14-6-05 were not analysed for glyphosate. TR, Torrens River; TRE, Torrens River estuary; TS, Torrens Shallow. See Figure 8 for codes and location of sample sites. Numbers within cells indicate the number of replicate samples collected.

| | Rainfall survey | | | | | |
|---|---|---|---|---|---|---|
| Site | May 2004 | | | June 2004 | June 2005 | |
| | 18-5-04 | 19-5-04 | 21-5-04 | 9-6-04 | 10-6-05 | 14-6-05 |
| TR | 1 | | | | | |
| TRE | | 1 | 1 | 1 | 2 | 2 |
| TS | | | 1 | | | |

### 2.1.3. Nitrogen isotope survey

Nitrogen exists in two naturally occurring stable isotopes, $^{14}N$ and $^{15}N$. The relative amount of the heavier isotope, $^{15}N$, can be used to differentiate the sources of nitrogen inputs in marine systems (Heaton 1986; McClelland *et al.* 1997) and is expressed through the notation $\delta^{15}N$ (in ‰) which is a measurement of the ratio of $^{15}N$ to $^{14}N$ in a sample relative to an internationally accepted standard (nitrogen in air):

$$\delta^{15}N = [(R_{sample} / R_{standard}) - 1)] \times 10^3$$

where R is defined as the atomic $^{15}N/^{14}N$ ratio of a sample and its relevant test standard.

During the sewage treatment process, animal wastes with a $\delta^{15}N$ of 5‰ that are high in ammonia are converted into nitrates via bacterial denitrification (Heaton, 1986). With a bacterial enzymatic preference for the lighter isotope $^{14}N$, treated waste becomes enriched in $^{15}N$. As a result, sewage derived from secondary or tertiary treatment facilities can be traced in the receiving environment via the enriched $\delta^{15}N$ signal (10-20‰), which is high relative to other sources such as fertiliser ($\delta^{15}N$ ~ 0‰) and soil (4-8‰) derived nitrogen (Heaton, 1986). However, recent work has indicated that nitrogen in the discharge waters from the Penrice soda factory in the Port River has an extremely high $\delta^{15}N$ signal of ca. 36‰ (R. Connolly, Griffith University, unpublished data). Based upon hydrodynamic modeling predictions for the region (Pattiaratchi *et al.* 2006) and an annual discharge of 1000 T nitrogen from Penrice into the Port River (which is equivalent to the combined annual discharge of nitrogen from Bolivar, Glenelg and Christies Beach WWTPs, Wilkinson *et al.* 2003), it is highly likely that nitrogen from Penrice reaches the offshore coast (and seagrasses) of Adelaide via Outer Harbour. Thus this potential source will need to be considered when interpreting results of the present nitrogen isotope survey.

In the present study, $\delta^{15}N$ values in seagrass were used to map the influence of anthropogenic nitrogen, most specifically derived from treated sewage, in Gulf St Vincent, following the framework developed by Costanzo *et al.* (2001). Seagrass samples were collected during July-October 2005 from 24 sites in ca. 5-10 m depth and <5 km offshore (from low water) around Gulf St Vincent. 16 sites were located within the ACWS area between Port Gawler and Sellicks Beach, with the remaining eight sites situated in western, northern and southeastern Gulf St Vincent. At each site, SCUBA divers collected four replicate seagrass samples consisting of three clumps of leaves and attached roots. Due to the moderate turnover rate of leaf material in *Posidonia* (leaf lifespan <1 year), leaf samples may provide an integrated signal from the previous year. In contrast, root material is longer-lived and may be indicative of nutrient conditions over a longer time period. Depending on the size of seagrass meadows, replicates were taken from four points a minimum of 2 m apart. *Posidonia* was sampled at all sites, except Site 23 where *Posidonia* was absent and *Amphibolis* was utilised instead. Samples were stored on ice in the field and transferred to a –30°C freezer upon returning to the laboratory.

Prior to processing, samples were thawed and rinsed thoroughly with water to remove excess surface salt and sediments. For each replicate, four green leaves were chosen at random and cleaned of epiphytes using the blunt edge of a scalpel. The underlying root structure beneath each of these leaves was also collected and analysed separately. All samples were freeze-dried and ground to a fine powder before being analysed for $^{14}N/^{15}N$ by Continuous Flow stable Isotope Ratio Mass Spectroscopy (CF IRMS) using a Europa Scientific ANCA-SL elemental analyser coupled to a Geo 20-20 Mass Spectrometer. Nitrogen isotopic abundances are reported as $\delta^{15}N$.

## 2.2. Sediment quality

Three replicate surface sediment samples (2 L) were collected from one station at each of the ambient water quality sites (except 1I which was not sampled; Figure 5) at the same time as the diver surveys (see below). Two replicate samples were also collected during a period of prolonged rainfall from each of two sites close to land-based inputs; adjacent to the Torrens River estuary in 2-3 m depth on 19 June 2004 and adjacent to the Barcoo Outlet in the same depth on 27 June 2004. Samples were kept on ice in the field and then placed in a -30° C freezer in the laboratory until they were sent away for analysis.

Initially only one sample from the eight ambient sites was analysed. If significant signals were detected in these samples, then the remaining replicates could be analysed. Both of the replicates from the Torrens and Barcoo sites were analysed. Samples were analysed for a suite of organochlorine and organophosphate pesticides, a suite of triazine herbicides, the herbicide Glyphosate, a suite of polyaromatic hydrocarbons, a range of total petroleum hydrocarbons, and a suite of trace elements (or heavy metals; see Appendix F and G for detailed lists of compounds that were analysed). Analyses were conducted by the Australian Government Analytical Laboratories (and in some cases outsourced by them) using NATA-accredited testing procedures.

## 2.3. Seagrass quality

Seagrass quality was investigated using two methodologies: a remote underwater video survey, and a SCUBA diver survey.

### 2.3.1. Video survey

Video footage was collected during December 2003 and March 2004 in order to characterise seagrass communities on a broad scale, and on October 19th and 20th 2004 to obtain an approximate current outer depth limit of *Amphibolis* and *Posidonia* and to search for the presence of *P. australis* in the nearshore of Holdfast Bay. A surface operated 450 line analogue camera, mounted on a small stainless steel frame suspended from the boat by rope and video cable, was used. Footage was recorded on a Sony digital video recorder along with positional data from a Koden differential GPS (recorded simultaneously with visual footage, via an audio encoder, to the audio track of the video tape). This made it possible to geo-reference data extracted from video footage.

Recordings were made of drift transects at two stations for each of the nine ambient sampling sites (Figure 5). Stations were generally separated by between 300 m and 600 m. Each transect was approximately 400 m although there was some variability in length due to prevailing conditions on the day (particularly at Site 1I where transects were closer to 250 m in length). In addition a series of transects were filmed at three points along the coastline to assess the outer depth limit. Drift transects were recorded (generally 100 –200 m) with every increase of 1 m depth moving offshore from approximately 12 metres.

Data were extracted from videotapes using a Visual Basic program designed in-house at SARDI. The user viewed the videotape and was able to select from a list of predetermined habitat categories (Table 4). The program combined the selected category with position information that was simultaneously downloaded from the audio track of the tape during viewing, into a text file that was then imported into a Microsoft Access database for processing. Accurate positional data made it possible to calculate the lengths of each section of homogeneous habitat. These lengths were then used to calculate percent cover data for each category at each location. In addition, habitat distribution (along transects) was plotted graphically.

**Table 4.** Video analysis habitat categories.

| Category | Description |
|---|---|
| Red/brown foliaceous algae | > 10% cover of foliaceous algae |
| *Amphibolis* dense | 90 - 100% cover |
| *Amphibolis* patchy | 40 - 90% cover |
| *Amphibolis* sparse | < 40% cover |
| *Halophila* | Any visible cover of *Halophila* |
| *Heterozostera* dense | 90 - 100% cover |
| *Heterozostera* patchy | 40 - 90% cover |
| *Heterozostera* sparse | < 40% cover |
| Mix *Posidonia* and *Amphibolis* | Any visible mixed stand |
| Mix *Posidonia* and algae | Any visible mixed stand |
| *Posidonia* dense | 90 - 100% cover |
| *Posidonia* patchy | 40 - 90% cover |
| *Posidonia* sparse | < 40% cover |
| Substrate | Bare substrate |

### 2.3.2. Diver survey

Due to logistical constraints, SCUBA surveys were conducted at only one station at each of the nine ambient sites (Figure 5) during April and June 2004. Eight of the nine sites were dominated by *Posidonia*, which was subsequently targeted for the sampling surveys. At Site 1I, *Heterozostera* dominated and was therefore targeted for more intensive sampling. At each station, a 50 m linear transect line was laid in a north-south direction across a continuous bed of seagrass. This transect was broken into five sets of five 0.25 x 0.25 m (0.0625 $m^2$) sample quadrats (each at 1 m intervals) with 5 m between each set (Figure 9) to give information at several spatial scales. At the two southern-most sites (4S, 4D), it was not possible to use this strategy as *P. coriacea* (which was the dominant species there) grows in small patches rather than meadows. For sites 4S and 4D, the 25 quadrats were haphazardly placed within patches of *P. coriacea*. Within each quadrat at the *Posidonia* sites, measurements were made of total leaf number and canopy height, and sub-samples of four shoots (with leaves intact) were removed from each corner of a quadrat for later analysis in the laboratory. Sub-samples were used to quantify various parameters (see below), and to discriminate different *Posidonia* species, as it was not possible to rapidly discriminate *P. angustifolia* from *P. sinuosa* in the field and a mixture of the two species was often sampled in the survey. For *Heterozostera* at Site 1I, *in situ* measurements were made of primary stem number and canopy height. No sub-samples were collected for analysis in the laboratory due to a lack of comparable sites within the present study and a lack of comparable historical data for *Heterozostera*.

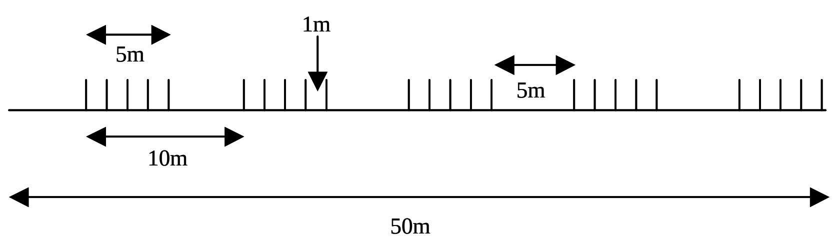


**Figure 9.** Sampling layout for diver surveyed transects (upright lines represent individual quadrats).

*Posidonia* material returned to the laboratory was used to record number of leaves per shoot to provide an estimate of the number of shoots per square metre using the leaf count data collected by divers. The length and width of each leaf, as well as the dry weight for a sub sample of leaves was also recorded. This information was then used to make estimates of average leaf length and width per quadrat and total standing dry biomass (per $m^2$) using the field data.

Estimates of epiphyte cover and coarse level composition were derived by randomly choosing one leaf from each quadrat, which was photographed under magnification in 3 cm sections (representing the base, middle and tip of the seagrass leaf). Photographs were processed through the use of a software package developed in-house at SARDI ('EpiMapper'). Images were imported to EpiMapper, which randomly overlays 20 points on the leaf area. The operator then assigns a value for each point to a taxonomic/descriptive class (e.g. fauna, encrusting coralline algae) from which an estimate of % cover can be generated for each image. Cover estimates for the base, middle and upper sections of the leaf were then averaged over the whole transect using the 25 quadrats. Due to the high diversity and uncertainty associated with epiphyte identifications (Johnson 2005), broad classifications were made.

Statistical analyses were generally not performed on data, as the objective of the study was not to look for differences between sites, but rather to describe the current status of existing meadows between Port Gawler and Sellicks Beach. Furthermore, any meaningful comparisons related to depth or position along the coast were confounded by seagrass species differences between sites and mixed species assemblages within sites and individual quadrats which made it impossible to analyse different seagrass species separately.

# 3. Results

## 3.1. Water quality

### 3.1.1. Ambient surveys

Results from the ambient surveys are presented in Figures 10 and 11 and summarised below for each of the water quality parameters. When interpreting the results, it must be acknowledged that any perceived seasonal differences may be an artifact of shorter-term temporal variation of which we are unaware due to a lack of within-season sampling. Nonetheless, results from the ambient surveys should provide a reasonable idea of the range of ambient conditions across a 1-year period. Values are compared against the ANZECC guideline values for marine waters in south central Australia (low rainfall area), and against a mean value derived from Gaylard (2004) for the Grange, Henley Beach, and Glenelg jetties where nutrients are known to be elevated relative to other jetty sites within the ACWS region.

*Total phosphorus*
A 2-way fixed factor analysis of variance (ANOVA) showed a highly significant interaction between site and season ($F_{24, 116} = 35.690$, $P < 0.001$) showing that patterns across sites and seasons were not consistent (Figure 10a). Nonetheless, mean levels were always highest at 1I for each season. Means ranged from ca. 0.015-0.13 mg P $L^{-1}$ at 1I but were generally $\leq$0.015 mg P $L^{-1}$ at all other sites (except for the winter sample at 1S = 0.045 mg P $L^{-1}$ which coincided with the maximum of 0.13 mg P $L^{-1}$ at the adjacent 1I). If site 1I is excluded, there are no strong indications of spatial patterns with location along the coast, but there is some evidence of a pattern with depth within Zones 1-3, with values in the shallow sites usually higher than the deep sites within each season. There are no strong indications of a seasonal pattern. Only at 1I during winter did a mean value exceed the ANZECC (2000) guideline value of 0.1 mg P $L^{-1}$ for marine waters in south central Australia (low rainfall area). Mean values for the 5 and 10 m depth sites were, with one exception (1S, winter), well below the mean value of 0.044 mg P $L^{-1}$ from the EPA data.

*Total nitrogen*
A 2-way fixed factor ANOVA showed a highly significant interaction between site and season ($F_{24, 116} = 4.175$, $P < 0.001$) showing that patterns across sites and seasons were not consistent (Figure 10b). Differences between sites were mainly due to Site 1I where mean levels were always highest in each season. Means ranged from 0.23-0.32 mg N $L^{-1}$ at 1I but were generally $\leq$0.17 mg N $L^{-1}$ at all other sites (except for the winter sample at 1S = 0.23 mg N $L^{-1}$). If Site 1I is excluded, there are no strong indications of spatial patterns with depth or location along the coast. There are no indications of a seasonal pattern. On no occasion did a mean value exceed the ANZECC guideline value of 1 mg N $L^{-1}$ for marine waters in south central Australia (low rainfall area). Mean values for the 5 and 10 m depth sites were well below the mean value of 0.340 mg N $L^{-1}$ from the EPA data.

*Total ammonia N*
A 2-way fixed factor ANOVA showed a highly significant interaction between site and season ($F_{24, 116} = 61.918$, $P < 0.001$) showing that patterns across sites and seasons were not consistent (Figure 10c). Mean levels ranged from 0.008-0.16 mg N $L^{-1}$ at the inshore site (1I) and from 0-0.06 mg N $L^{-1}$ at the deep and shallow sites. There is a strong indication of the same seasonal trend at all sites with highest levels in autumn and second highest levels in winter (except for Site 1I which also had a relatively high level in summer and was a major contributor to the significant interaction between site and season). The autumn values were not directly correlated with recent stormwater events (Figures 6 and 7). When excluding Site 1I, there were no strong indications of spatial patterns with depth or position along the coast. In four samples (all during autumn), mean values exceeded the ANZECC guideline value of

0.05 mg N $L^{-1}$ for marine waters in south central Australia (low rainfall area). Mean values for the 5 and 10 m depth sites were usually below the mean value of 0.042 mg N $L^{-1}$ from the EPA data; notable exceptions were Sites 1D, 1S, 2D, 2S, and 3D during autumn.

*Nitrate + nitrite*
A 2-way fixed factor ANOVA showed a highly significant interaction between site and season ($F_{24, 116}$ = 7.764, $P$ < 0.001) showing that patterns across sites and seasons were not consistent (Figure 10d). Mean levels ranged from 0 to 0.006 mg N $L^{-1}$. There was some evidence of a seasonal pattern with relatively low or undetectable levels during summer and autumn and peak levels during winter and spring. There are no strong indications of spatial patterns with depth or location along the coast. On no occasion did a mean value exceed the ANZECC guideline value of 0.05 mg N $L^{-1}$ for marine waters in south central Australia (low rainfall area). All mean values for the 5 and 10 m depth sites were well below the mean value of 0.033 mg N $L^{-1}$ from the EPA data, and were also well below mean values for all metropolitan jetties (see Gaylard 2004).

*All nutrient parameters*
Total nitrogen was generally about 10 times or more higher than total phosphorus at all sites and seasons (Figure 10). The inorganic component of total nitrogen (Total ammonia N, nitrate + nitrite) was relatively low (around 10-20%), except for autumn when it was around 40-60 % (Figure 10). Total ammonia N (ammonia + ammonium) was about 10 times higher than oxidized nitrogen (nitrate + nitrite) across the surveys (Figure 10).

*Chlorophyll-*a
A 2-way fixed factor ANOVA showed a highly significant interaction between site and season ($F_{24, 72}$ = 36.786, $P$ < 0.001) showing that patterns across sites and seasons were not consistent (Figure 11a). Mean values were generally ≤1.5 μg $L^{-1}$ across all sites and seasons, except for values of 4.4 μg $L^{-1}$ during summer at 1I and 2.4 μg $L^{-1}$ during winter at 3S. There are no clear seasonal trends, although winter tended to have peak values at most sites. There are no strong indications of spatial patterns with depth or location along the coast. On 10 occasions (six of those during winter), values exceeded the ANZECC guideline value of 1 μg $L^{-1}$ for marine waters in south central Australia (low rainfall area). Mean values for the 5 and 10 m depth sites were well below the mean value of 3.82 μg $L^{-1}$ from the EPA data.

*Other biological indicators*
Faecal coliforms, *E. coli*, and dissolved organic carbon were sampled only on the summer survey. Faecal coliforms were less than the detectable level (1 MPN. $100mL^{-1}$) at most sites, but were found at Site 1I and the shallow sites in Zones 1-3 (Figure 11b). The highest mean faecal coliform level of 3.3 MPN. $100mL^{-1}$ was at found at 1I. Dissolved organic carbon was less than the detectable level of 1 mg $L^{-1}$ for most samples (Figure 11c). The highest mean dissolved organic carbon level of 2.3 mg $L^{-1}$ was at found at 1I. *E. coli* was less than the detectable level (1 MPN. $100mL^{-1}$) at all sites and the data are therefore not graphed.

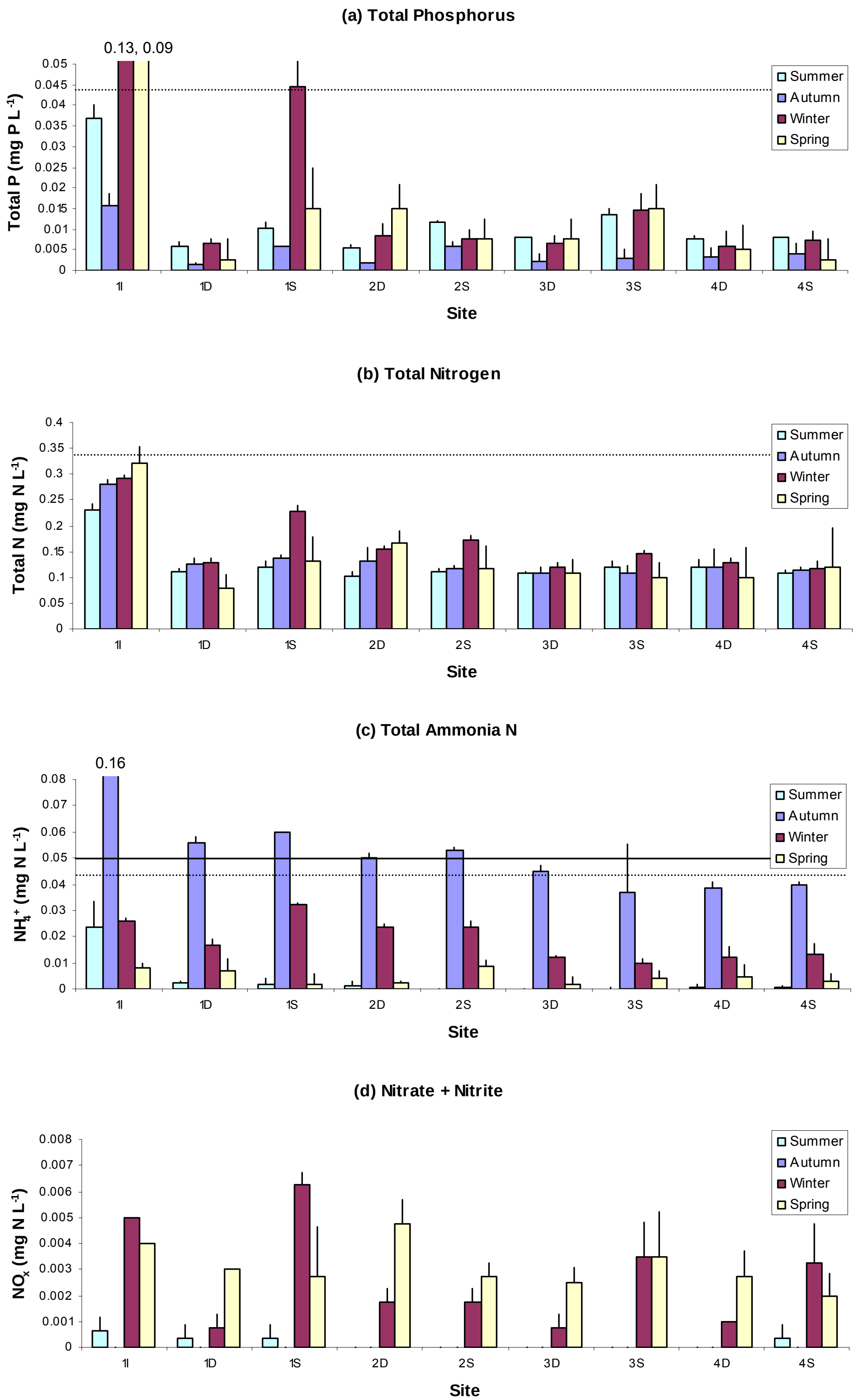


**Figure 10.** Mean (± SD) dissolved nutrient concentrations from the nine sampling sites used in the four seasonal ambient water quality surveys. Total ammonia N is ammonia + ammonium. n = 6, 3, 4, 4 for summer, autumn, winter, and spring, respectively. See Figure 5 for locations of sites. Solid horizontal lines indicate ANZECC water quality guidelines, and dashed horizontal lines indicate mean EPA values from Grange, Henley Beach, and Glenelg jetties, for those cases where values were within the ranges plotted.

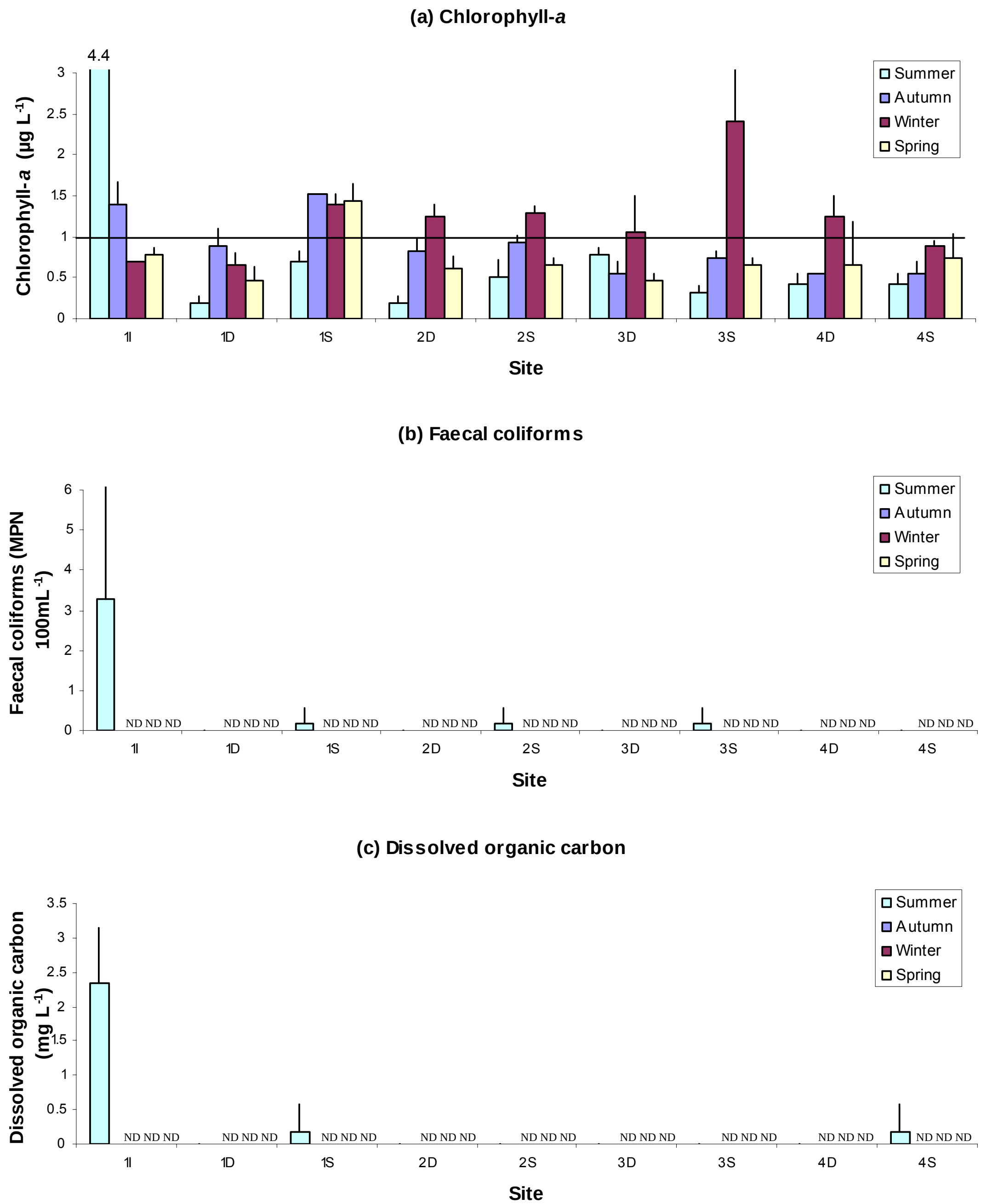


**Figure 11.** Mean (± SD) concentrations of (a) chlorophyll-*a*, (b) faecal coliforms, and (c) dissolved organic carbon from the nine sampling sites used in the four seasonal ambient water quality surveys. For chlorophyll-*a*, n = 3 (Station A only). For faecal coliforms and dissolved organic carbon, n = 6. See Figure 5 for locations of sites. ND = no data. Solid horizontal line indicates ANZECC water quality guideline for chlorophyll-*a*.

### 3.1.2. Rainfall event surveys

*Nutrients*
Following major rainfall events, there was some evidence of localized elevated nutrients in very shallow waters but little evidence for elevated nutrients in inshore waters where seagrasses have been lost or in offshore waters where seagrasses occur. The four shore-based sites in and around the Torrens outlet (HB, NT, TRE, ST) usually indicated a gradient of decreasing nutrient concentration away from the Torrens River estuary, and the shore-based sites usually had higher values than the ocean sites (Figures 12 and 13). Visual observations on all three surveys confirmed that turbid stormwater was entering the ocean from the Torrens River. On the June 2004 survey, it was apparent that detectable inputs were also coming from the Barcoo outlet (BO) and stormwater drains near the Broadway (BW, Figure 13a, c, d). On occasion there was some evidence of a decreasing nutrient gradient from inshore to offshore sites within a transect line (e.g. 2I, 2S, 2D for TP, Figure 12a; 3I, 3S, 3D for TP, Figure 13a). Despite the evidence of nutrient inputs from the Torrens River, nutrient levels at the shore-based sites away from the Torrens River (HB, BO, BW) and at the ocean-based inshore sites (2I, TI, 3I) mostly did not exceed the long-term means from the Henley Beach, Glenelg and Brighton jetties (Figures 12-14). Furthermore, nutrient levels at the ocean sites rarely exceeded the maximum values obtained from the ambient surveys (Figures 12-14). The levels of oxidized nitrogen detected in and around the Torrens River on the June 2004 survey (Figure 13d) are considered to be extremely high and are well in excess of the ANZECC guideline value of 0.05 mg N $L^{-1}$ for marine waters in south central Australia (low rainfall area). Other values from the rainfall surveys (excluding the Torrens River estuary site) rarely exceeded the ANZECC guideline values of 0.1 mg P $L^{-1}$, 1 mg N $L^{-1}$, and 0.05 mg N $L^{-1}$ for total phosphorus, total nitrogen and total ammonia N, respectively, in marine waters in south central Australia (low rainfall area).

*Toxicants*
Only one compound, simazine (a triazine herbicide), was detected in samples collected during three major rainfall events (Appendix D and E). Simazine was found in the Torrens River estuary during May and June 2004 at concentrations of 0.23 and 0.95 $\mu g\ L^{-1}$, respectively. These levels were greater than the ANZECC trigger value for freshwater of 0.2 $\mu g\ L^{-1}$ (for 99% species level of protection) but lower than the trigger value of 3.2 $\mu g\ L^{-1}$ (for 95% species level of protection). No equivalent values are available for marine waters. No simazine was detected in the marine samples collected adjacent to the Torrens River estuary.

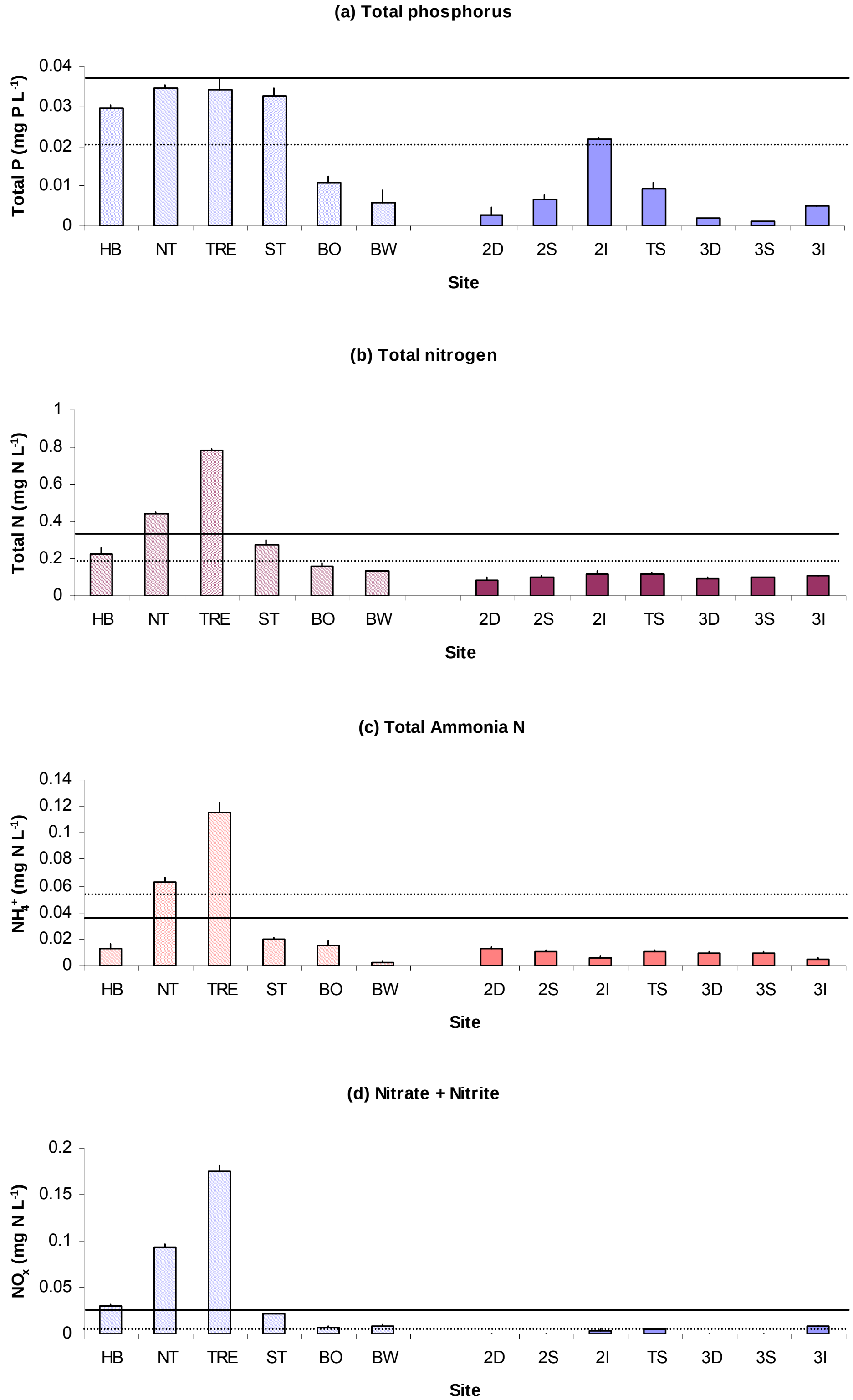


**Figure 12.** Mean (± SD) nutrient concentrations from the 13 sampling sites used in the March 2004 rainfall water quality survey. Total ammonia N is ammonia + ammonium. Hatched and solid bars indicate shore-based sites and ocean sites, respectively. n = 2 for shore sites. n = 3 for ocean sites. Values of zero are due to all replicates being < detectable limit. Solid horizontal line indicates mean EPA value from Henley Beach, Glenelg & Brighton jetties. Dashed horizontal line indicates maximum value from ambient water quality surveys.

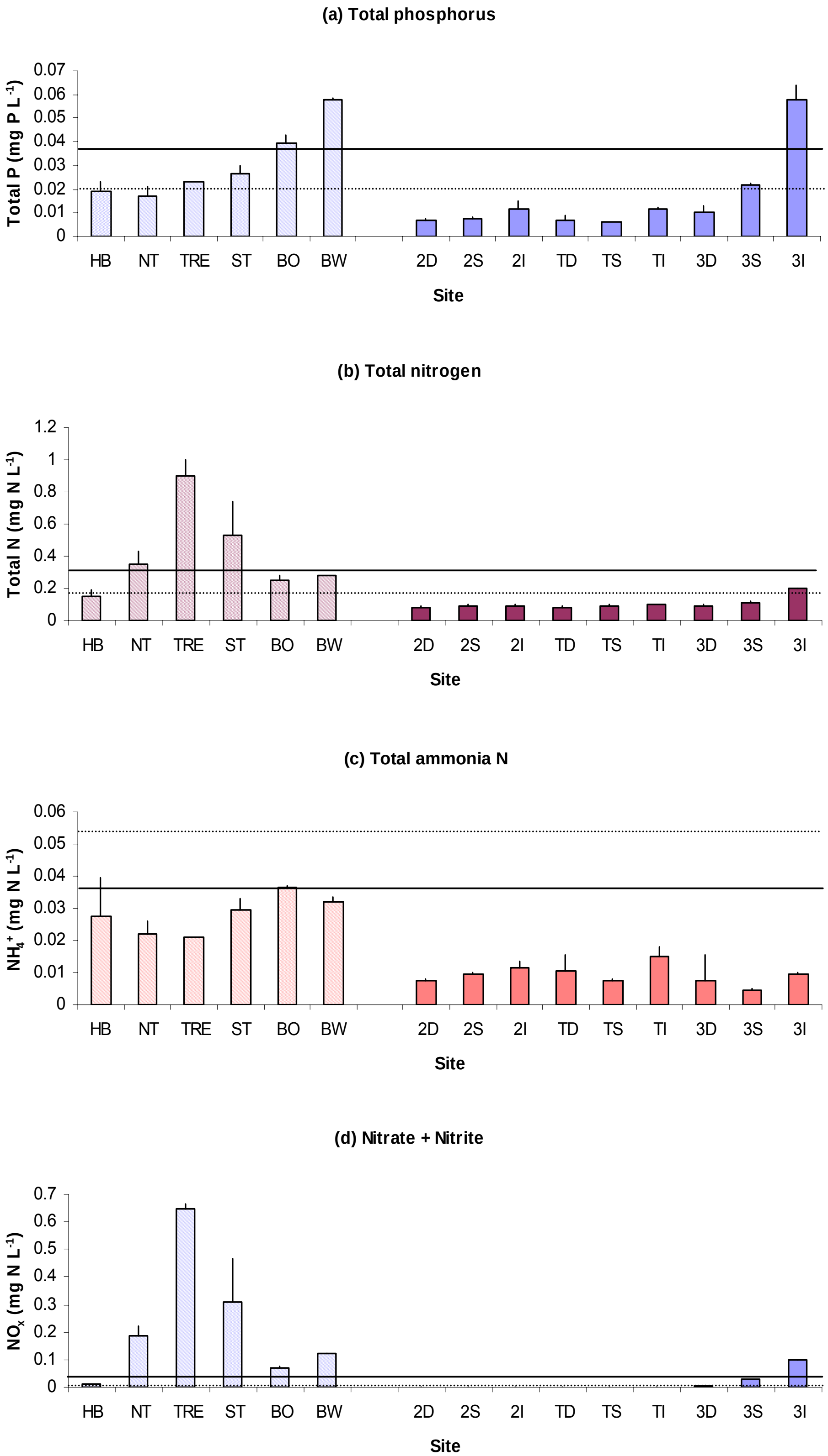


**Figure 13.** Mean (± SD) dissolved nutrient concentrations from the 15 sampling sites used in the June 2004 rainfall water quality survey. Total ammonia N is ammonia + ammonium. Hatched and solid bars indicate shore-based sites and ocean sites, respectively. n = 2 for all sites. Values of zero are due to all replicates being < detectable limit. Solid horizontal line indicates mean EPA value from Henley Beach, Glenelg & Brighton jetties. Dashed horizontal line indicates maximum value from ambient water quality surveys.

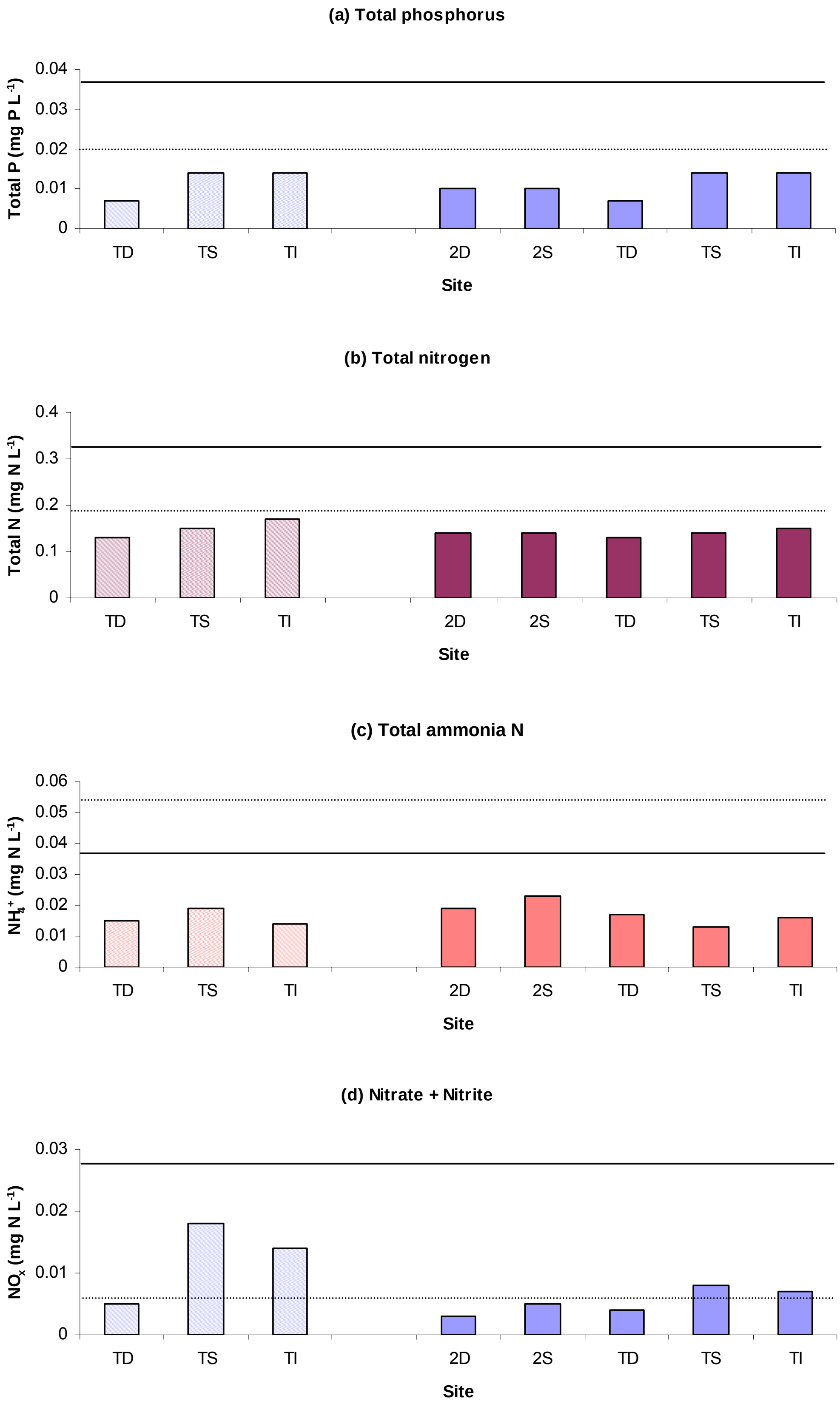


**Figure 14.** Dissolved nutrient concentrations from the 5 ocean sampling sites used in the May 2004 rainfall water quality survey. Total ammonia N is ammonia + ammonium. Hatched and solid bars indicate ocean sites sampled on 19 & 21 May 2004, respectively. n = 1 for all sites. Solid horizontal line indicates mean EPA value from Henley Beach, Glenelg & Brighton jetties. Dashed horizontal line indicates maximum value from ambient water quality surveys.

### 3.1.3. Nitrogen isotope survey

There was significant spatial variability in $\delta^{15}N$ levels for both seagrass leaves and roots collected around the coast of Gulf St Vincent (Figures 15 and 16). Leaves and roots showed broadly similar patterns of $^{15}N$ enrichment, with $\delta^{15}N$ levels at the southern, western and northern Gulf sites all within 'background' ranges (leaf = -3.36 to 1.40‰, root = -3.36 to -0.43‰). In contrast, leaf and root samples collected along the metropolitan coastline from Port Gawler to Port Noarlunga (Sites 7-20), were found to be enriched in $\delta^{15}N$ (leaf = 5.1 to 14.4‰, root = 4.1 to 12.7‰) and were significantly greater than all other sites (as shown by Tukey HSD tests following separate 1-way ANOVAs of leaves: $F_{23, 71} = 158.696$, $P < 0.001$, and roots: $F_{23, 91} = 88.330$, $P < 0.001$). Values were markedly higher at sites adjacent to Barker Inlet (leaf = 9.1 to 14.4‰, root = 9.1 to 12.7‰) followed by sites adjacent to Glenelg (leaf = 7.1 to 9.0 ‰, root = 5.1 to 7.0 ‰; Figure 16). $\delta^{15}N$ levels were consistently higher in leaves than roots (particularly at Sites 7-20 which had the greatest enrichment), with the obvious exceptions of Sites 6, 21 and 22 (Figure 16).

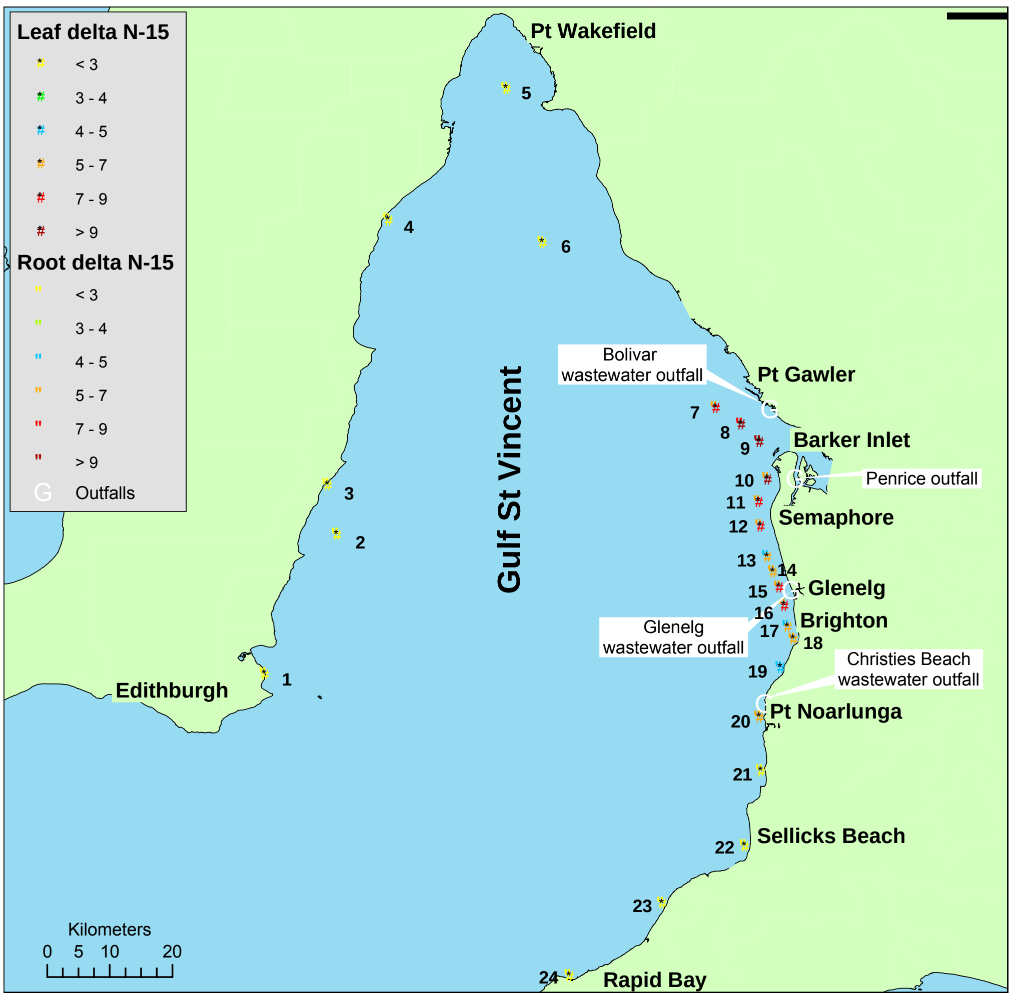


**Figure 15.** Spatial distribution of $\delta^{15}N$ in seagrass leaf and root samples collected from 24 sites around Gulf St Vincent.

The greatest enrichment of $\delta^{15}N$ occurred at sites within the vicinity of the WWTP and Penrice outfalls, with the degree of this enrichment significantly inversely related to distance from the nearest outfall (Figures 16 and 17; linear regression, $r^2 = 0.58$, $F_{1, 23} = 30.361$, $P < 0.001$). $\delta^{15}N$ in the upper northern reaches of the Gulf, taken at sites ≥ 20 km northeast of Barker Inlet, returned to background levels (leaf = 0.9-1.4‰, root = 0.7-1.2‰; Figure 16). Sites further south of Brighton begin to decrease with distance from the Glenelg WWTP outfall (minimum leaf = 4.9‰, root = 4.1‰) but again rise in the vicinity of the Christies Beach WWTP outfall (leaf = 6.6‰, root = 6.8‰; Figure 16). Sites >10km south of the Christies Beach WWTP outfall drop to background levels (leaf = -0.8 to 2.14‰, root = 0.6-1.61‰; Figure 16).

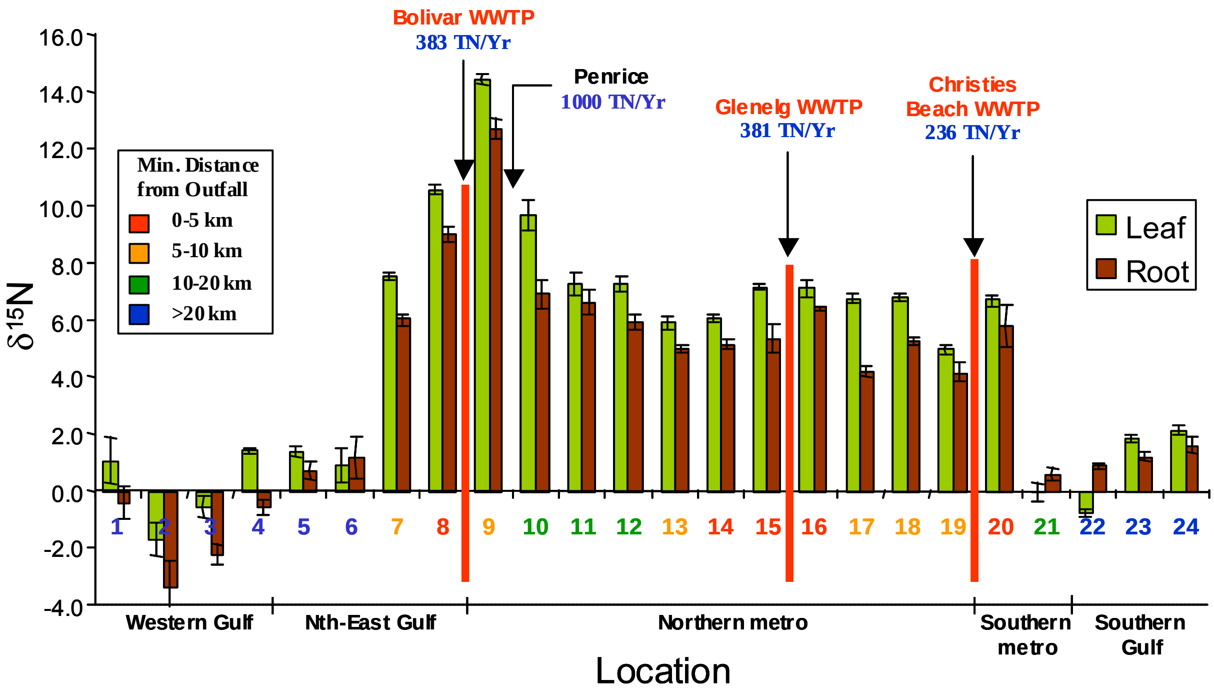


**Figure 16.** Mean $\delta^{15}N$ (± SE) in seagrass leaves (green) and roots (brown) as a function of location around the coast of Gulf St Vincent. Sites are labelled 1-24 in colour categories representing their relative distance from an outfall. Vertical red lines indicate the locations of each of the three coastal wastewater treatment plant (WWTP) outfalls; Bolivar, Glenelg and Christies Beach. n = 4 for all sites and tissues, except n = 2 for roots at Sites 15 and 22, and n = 3 for leaves and roots at Site 21.

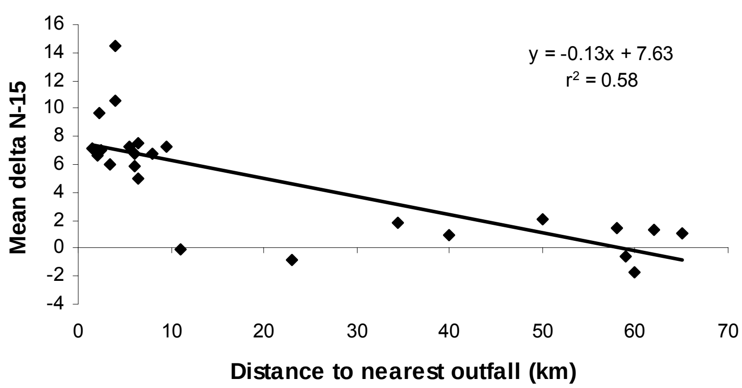


**Figure 17.** Relationship between mean leaf $\delta^{15}N$ and distance to the nearest coastal wastewater outfall, for each of the 24 sites sampled around Gulf St Vincent. Outfalls included are Bolivar, Glenelg, and Christies Beach WWTPs, and Penrice. Distance to the Penrice outfall was calculated from the mouth of the Port River at Outer Harbour. Fitted line is a linear regression.

## 3.2. Sediment quality

Concentrations of organochlorine and organophosphate pesticides, triazine herbicides, Glyphosate, polyaromatic hydrocarbons, and total petroleum hydrocarbons were all below detectable limits in the 12 sediment samples that were laboratory tested (Appendix F and G). The heavy metals chromium and zinc were detected at all 10 sites tested, while lead was detected at eight of the 10 sites but was not detected at the two Zone 3 sites (Appendix F and G; Figure 18). Copper was detected only at the two sites within Zone 1 (Appendix F and G; Figure 18). Concentrations of the four heavy metals were low, with ranges of 0.6-11, 0.5-23, 0.96-4.3, 0.73-0.85 mg/kg dry wt for chromium, zinc, lead, and copper, respectively. These levels are all well below the ANZECC (2000) recommended sediment quality guideline trigger values of 80, 200, 50, and 65 mg/kg dry wt, for chromium, zinc, lead, and copper, respectively. While there was little or no replication within sites (thus requiring caution with any conclusions drawn from between-site comparisons), there were no clear spatial patterns related to depth, except in Zone 4 where levels of chromium, lead, and zinc were all markedly higher at the deep site compared to the shallow site (Figure 18). In fact, the deep site at Zone 4 (4D) had the highest levels of chromium, lead, and zinc out of all 10 sites (Figure 18). The two sites closest to land-based discharges (i.e., Torrens and Barcoo) had similarly low levels of chromium, lead, and zinc to the other more offshore sites in Zones 1-3 where those compounds were also detected (Figure 18).

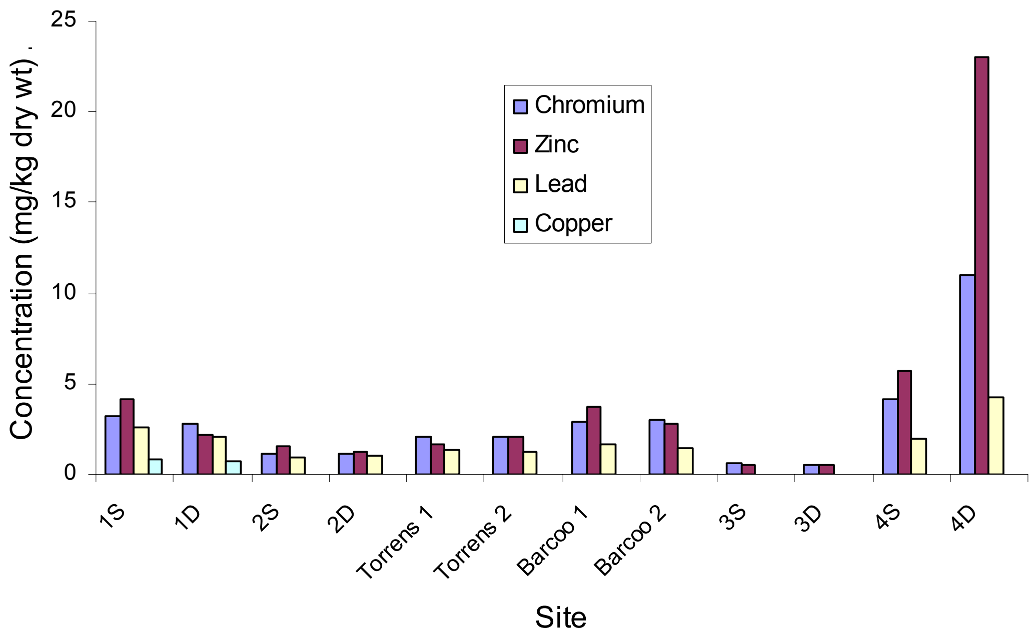


**Figure 18.** Concentrations (mg/kg dry wt) of the heavy metals chromium, zinc, lead and copper from sediment samples collected at 10 sites across the Adelaide metropolitan coast. The Torrens and Barcoo sites had two replicate samples (1 and 2); at all other sites n = 1. Values less than the detectable limits of analysis are presented as zero. Raw data are shown in Appendix F and G.

## 3.3. Seagrass quality

### 3.3.1. Video survey

*Seagrass cover and composition*
Seagrass cover and composition changed markedly from north to south and from inshore to offshore. While *Posidonia* dominated the deep sites in Zones 1-3, the shallow sites (particularly in Zones 2 and 3) were more diverse in nature, with *Amphibolis* being a major feature in Zone 2 and both algae and *Amphibolis* featuring in Zone 3 (Figure 19). The shallow sites also had greater cover of bare substrate than the deep sites in Zones 1-3 (Figure 19). The intertidal inshore site in Zone 1 was characterised by a dense cover of *Heterozostera* with some *Posidonia* present (Figure 19). The two sites in Zone 4 were markedly different from all others sites with a low cover of seagrass (Figure 19), comprised of occasional isolated clumps of *P. coriacea* that were generally only metres in diameter.

In terms of cover alone, the three northern deep sites (1D, 2D, 3D) appear quite similar (Figure 19). However, when the spatial distribution of habitats along transects is examined, Site 3D is quite different from the other two sites (Figure 20). The deep sites in Zones 1 and 2 consist of relatively dense and consistent meadows of *Posidonia* while in Zone 3, the meadows are considerably fragmented. Similarly, while the shallow sites in Zones 2 and 3 appear similar in terms of cover (Figure 19), clear differences in spatial distribution can be seen (Figure 20). The patchy nature of *P. coriacea* at the Zone 4 sites is clearly shown in Figure 20. The seagrass beds at the intertidal inshore site of Zone 1 were comprised mainly of continuous monospecific sections of dense *Heterozostera* and *Posidonia* (Figure 20).

*Outer depth limit of* Posidonia *and* Amphibolis
The outer depth limit of *Posidonia* varied along the coast (Figure 21). In Zone 1, off Barker Inlet, *Posidonia* was found to approximately 15 m depth, while *Halophila* was also present at this depth and deeper. In Zone 2, off Henley Beach, significant beds of *Posidonia* were also present to 15 m depth, but with thinner sparse cover to between 16-18 m depth. Similarly, in Zone 3 off Brighton, *Posidonia* became sparse at approximately 16 m depth but was still present, albeit in very small amounts, at 17-18m depths. Past 18 m depth, *Posidonia* was not observed.

*Amphibolis* was not detected in the Zone 1 transect. However, on the Zone 2 and 3 transects it was found both nearshore and out to depths of around 10-12 m (Figure 21).

*Presence of* Posidonia australis *in Zone 2*
A search in the nearshore region of Largs Bay failed to detect the presence of the previously recorded *P. australis*. The survey region was characterized by sparse *Posidonia* (probably *P. sinuosa*), with extensive areas of dead root mat.

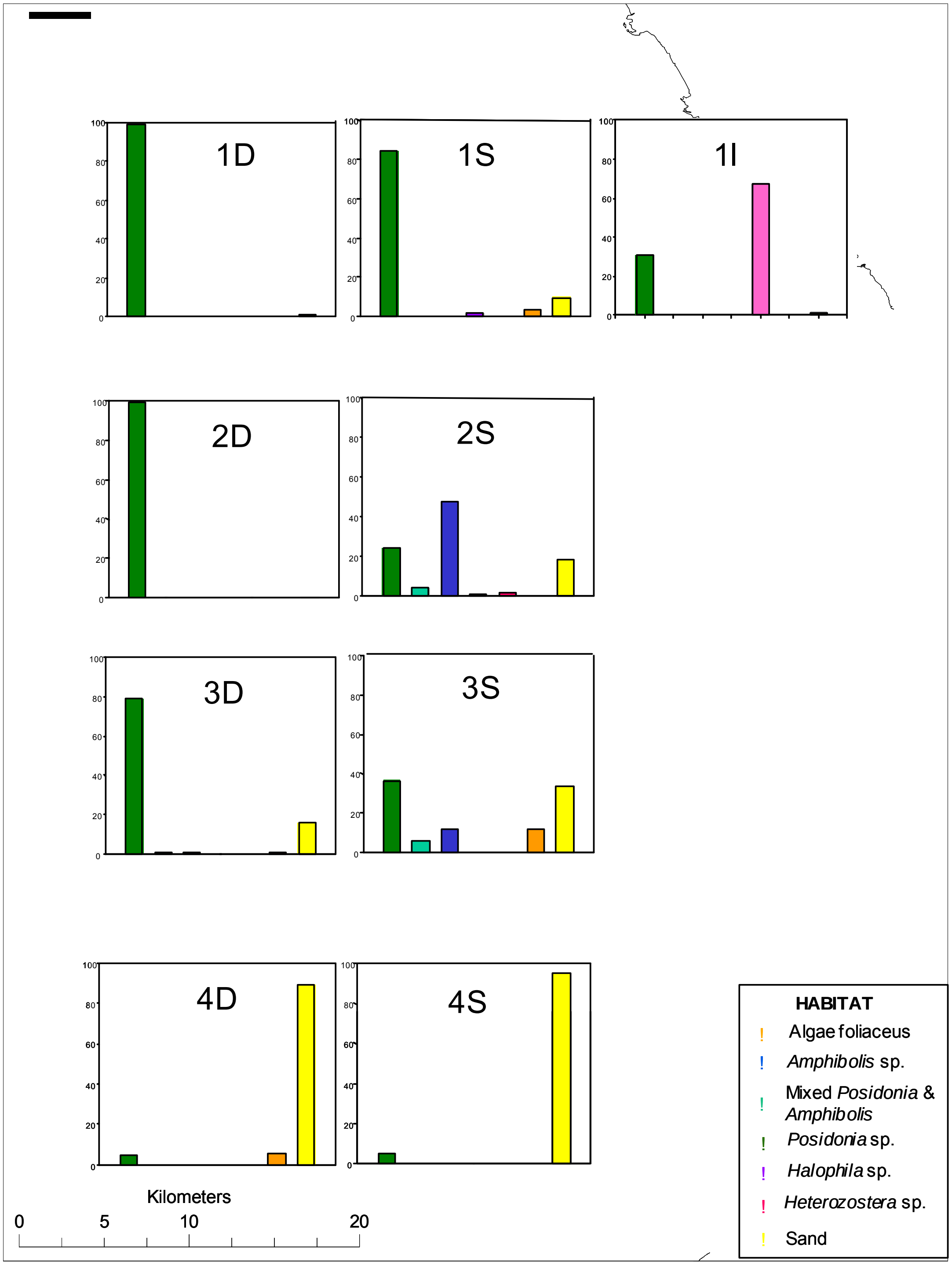


**Figure 19.** Benthic cover (%) of seagrasses from the pooled video data taken at two stations for each of the nine ambient monitoring sites during December 2003 (deep and shallow sites) and March 2004 (inshore site).

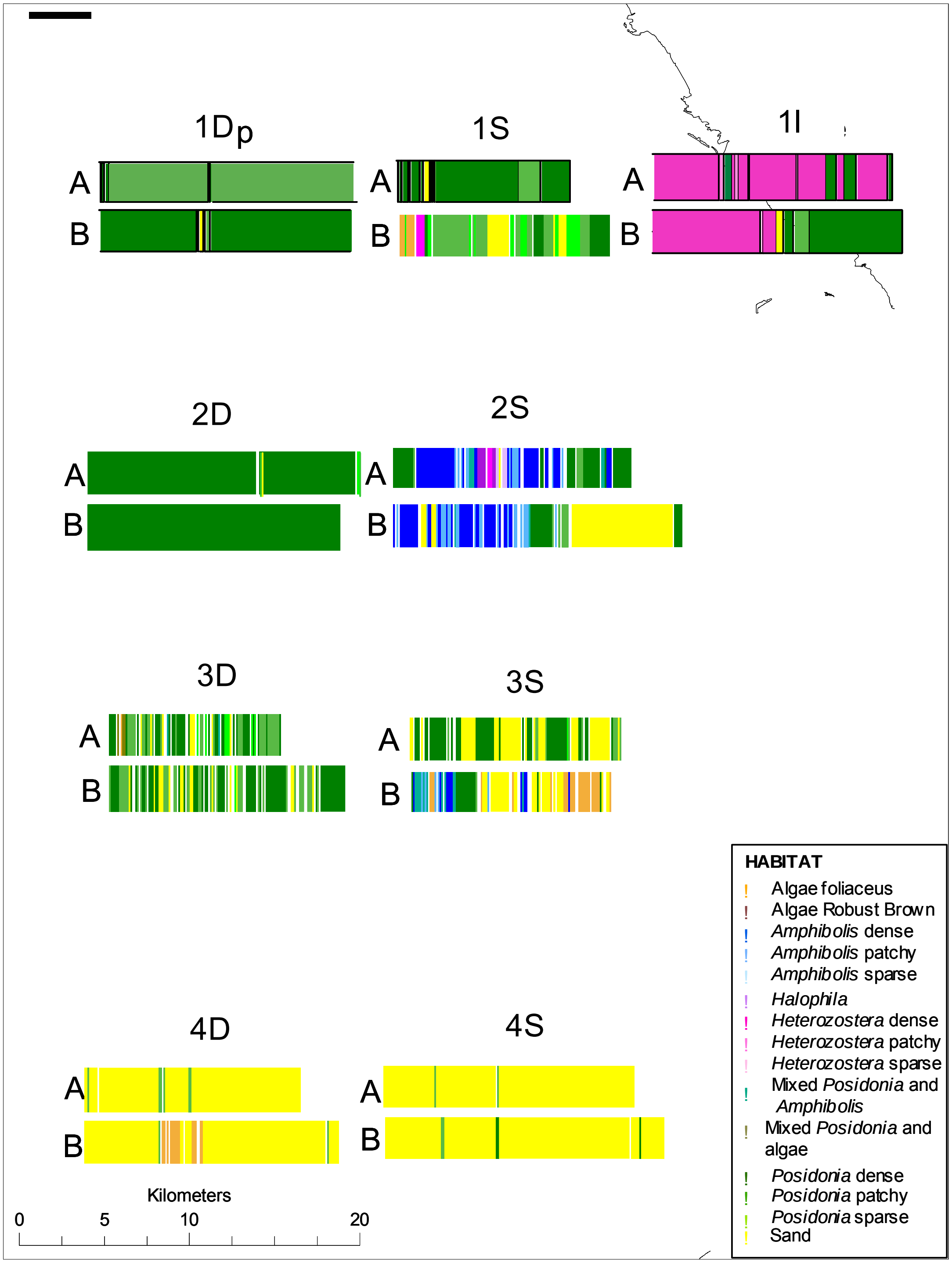


**Figure 20.** Species distribution and composition along ~300-400 m long video transects taken at two stations for each of the nine ambient monitoring sites during December 2003 (deep and shallow sites) and March 2004 (inshore site).

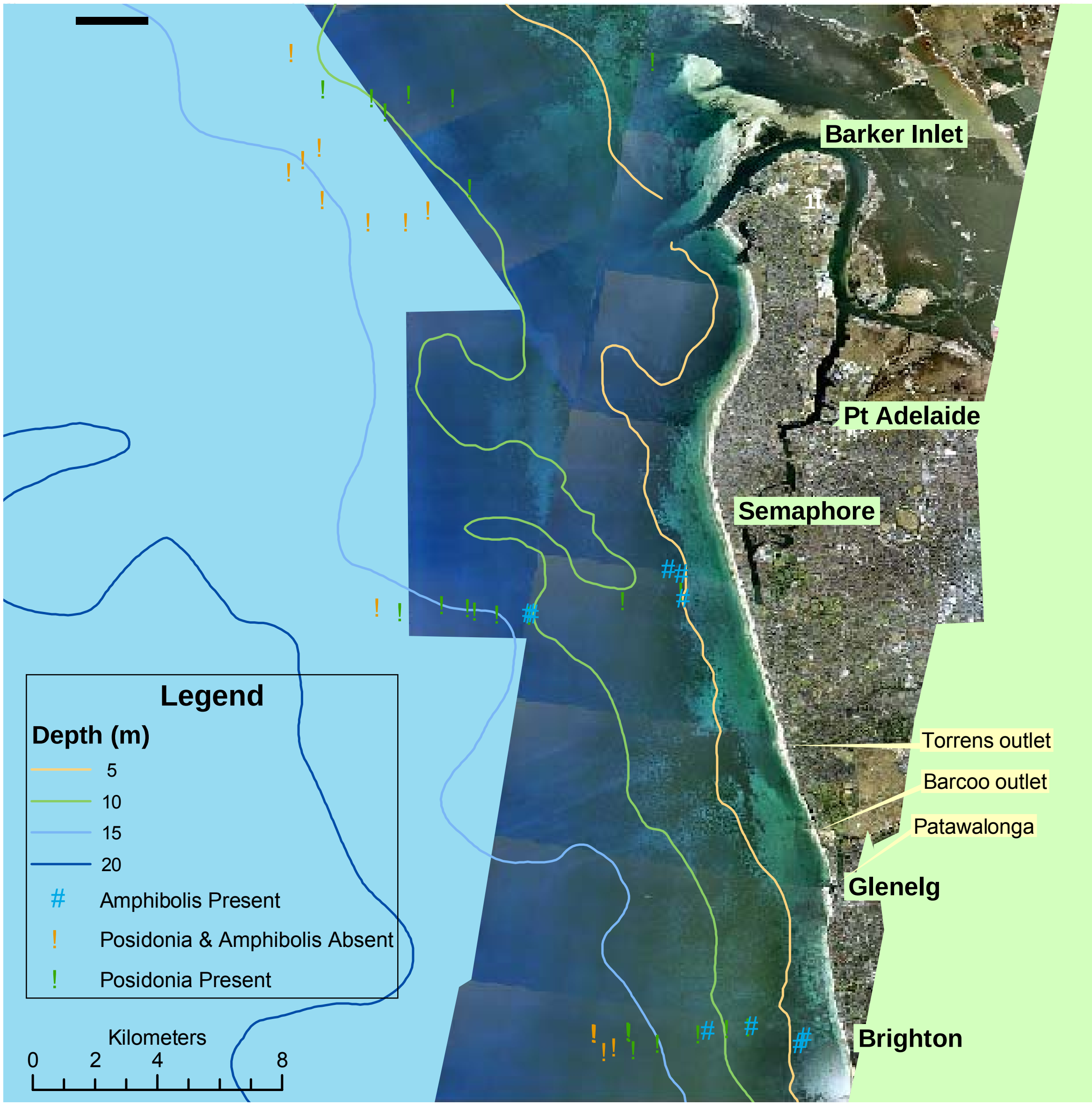


**Figure 21.** Occurrence of *Amphibolis* and *Posidonia* along three inshore to offshore transects between Barker Inlet and Brighton during October 2004. Linear lines in 10-20 m depths indicate the offshore limits of aerial photography.

### 3.3.2. Diver survey

*Species composition*

The inshore site of Zone 1 was exclusively comprised of *Heterozostera* (Figure 22). Despite limited large-scale spatial replication, patterns of species change in *Posidonia* are evident within Zones 1-3 where *P. sinuosa* was dominant in the shallow sites and *P. angustifolia* was dominant in the deep sites of Zones 2 and 3. *P. coriacea* was the only *Posidonia* species found at the Zone 4 sites, where patches of *P. coriacea* had to be deliberately targeted for quadrat sampling.

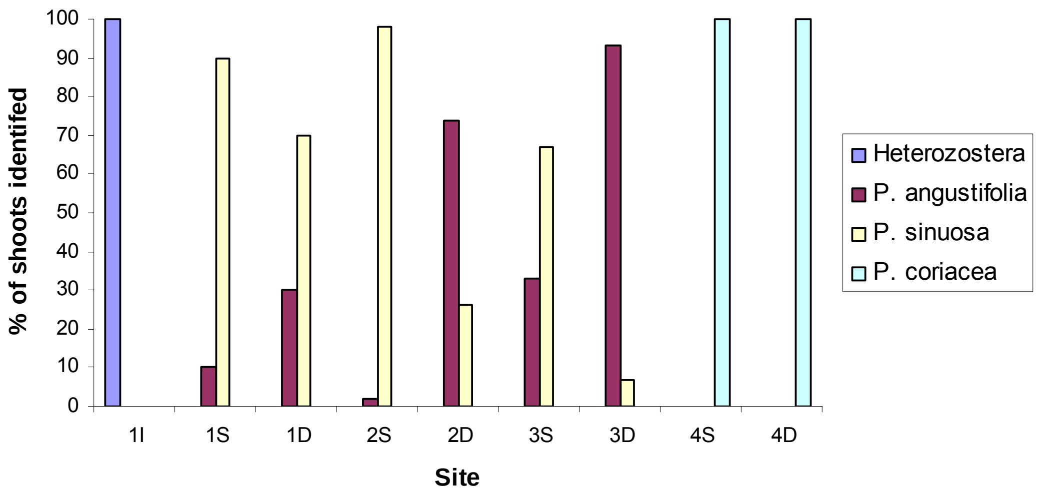


**Figure 22.** Species composition at each of the nine ambient monitoring sites determined from identified shoot sub-samples.

*Seagrass quality*

While patterns of seagrass quality across zones and depths (Figures 21 and 22) were probably influenced strongly by differences in species composition (Figure 22), some patterns may be physiologically driven. For example, aboveground biomass (which is an integration of leaf density, shoot density, leaf length, and leaf width) was consistently higher at the shallow sites than the deep sites in Zones 1, 2 and 3 (Figure 23a). This pattern was due to longer and wider leaves in the shallow sites of Zones 1-3 (Figure 24 b, c), despite generally lower leaf and shoot densities (Figure 23 b, c). These patterns were probably influenced by the occurrence of *P. angustifolia* at the deep sites of Zones 1-3 (Figure 22), as *P. angustifolia* generally has narrower leaves than *P. sinuosa* (Cambridge and Kuo 1979). No clear patterns across depth were seen for *P. coriacea* in Zone 4 (Figures 23 and 24). None of the sites displayed consistently low values across the array of seagrass quality parameters (Figures 23 and 24) that may have suggested an unhealthy meadow. There was considerable variation within sites for several of the parameters. For example, a 1-way nested ANOVA of leaf density (with grouping of adjacent blocks of five quadrats within each of the *Posidonia* sites) showed a significant difference within ($F_{1, 32}$ = 3.607, $P$ < 0.001) and between sites ($F_{1, 7}$ = 13.067, $P$ < 0.001). Such variability has implications for sampling in any research or monitoring program, and we suggest that around 10 quadrats is the minimum number required to quantify leaf density in *Posidonia angustifolia/sinuosa* meadows off Adelaide (Figure 25). At Site 1I, the *Heterozostera* appeared healthy with a stem density of 1061 ± 433 stems/m$^2$ (mean ± SD, n = 25) and a canopy height of 40.3 ± 9.2 cm (mean ± SD, n = 25). While not quantified, epiphyte load did not appear excessive.

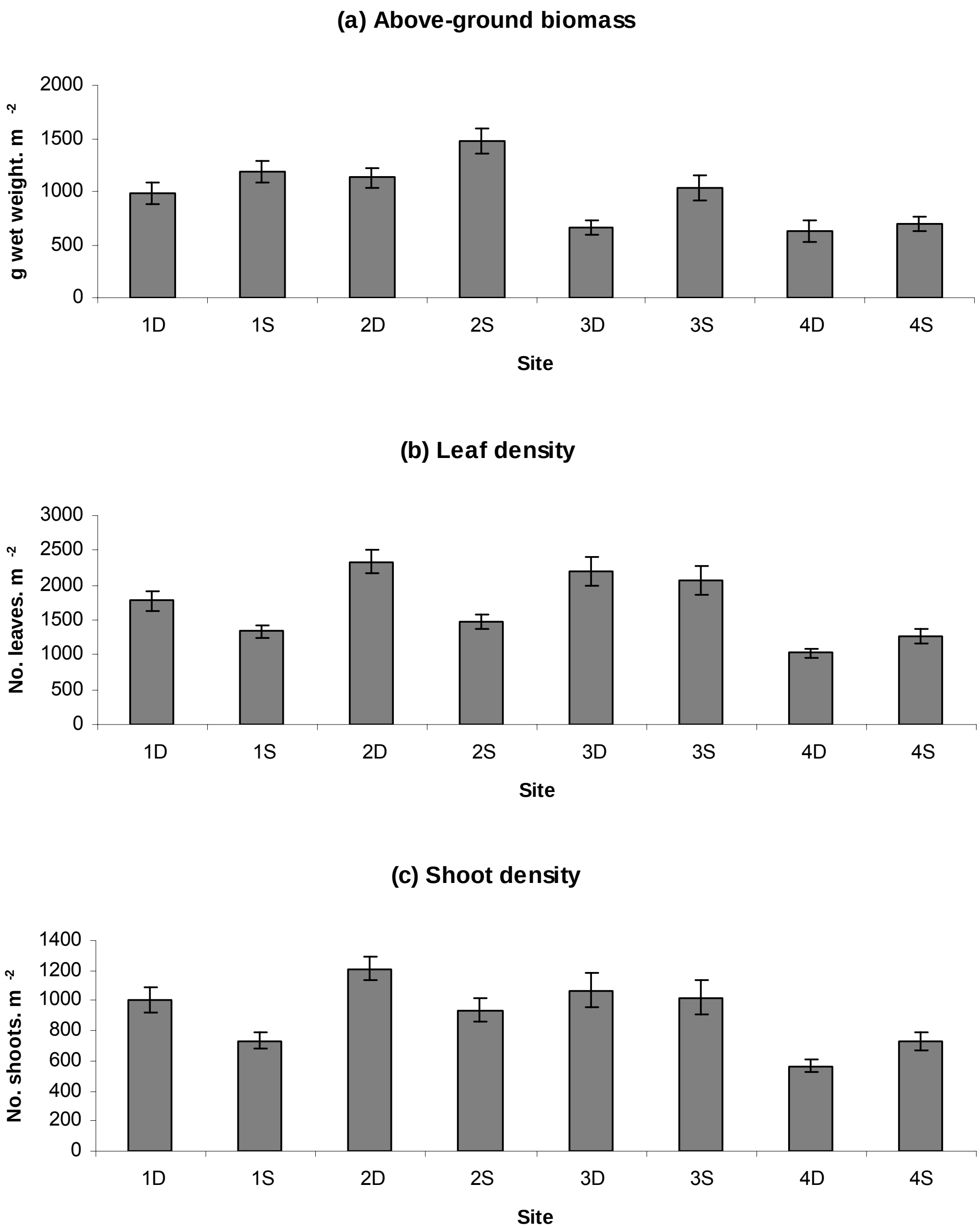


**Figure 23.** Seagrass quality parameters of (a) aboveground biomass, (b) leaf density, and (c) shoot density at deep (D) and shallow (S) sites in Zones 1-4. Values are mean ± SD (n = 5 for all sites).

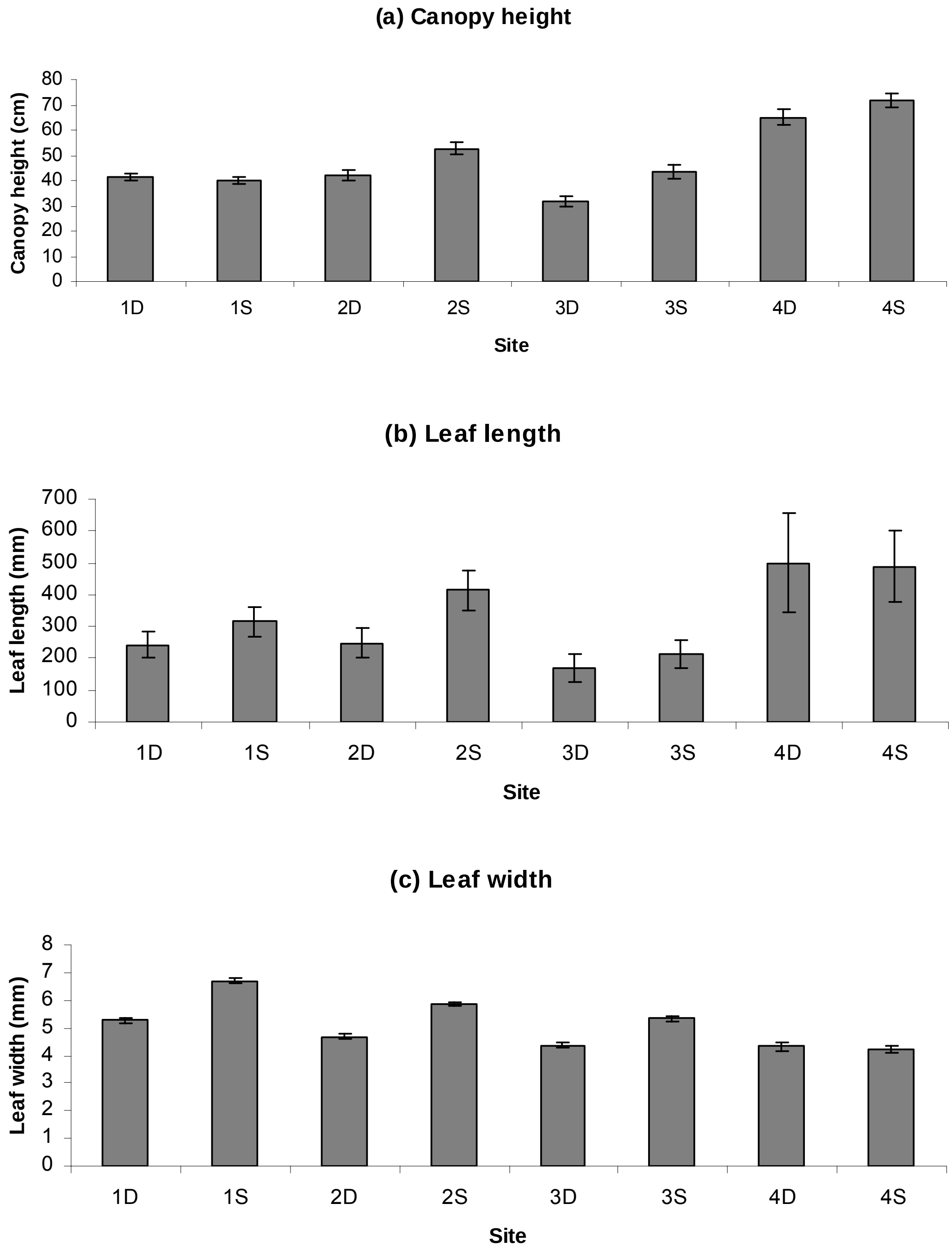


**Figure 24.** Seagrass quality parameters of (a) canopy height, (b) leaf length, and (c) leaf width at deep (D) and shallow (S) sites in Zones 1-4. Values are mean ± SD (n = 5 for all sites).

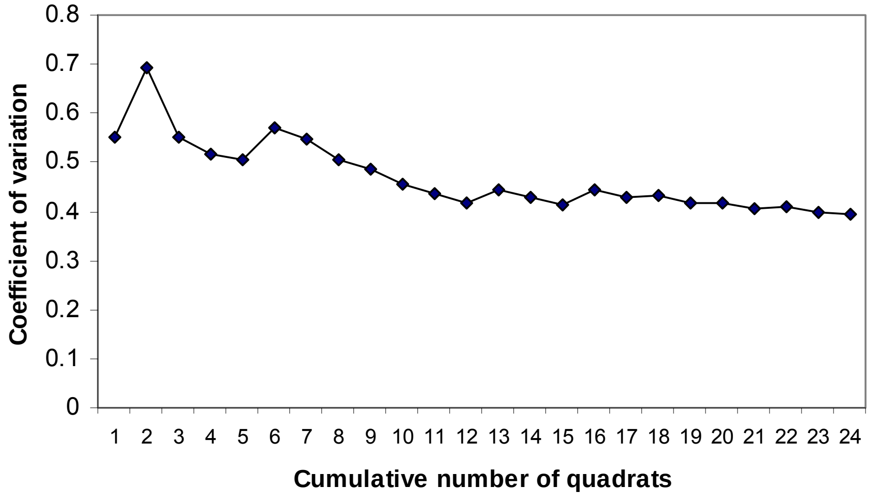


**Figure 25.** Effect of increasing number of 0.0625 $m^2$ quadrats on the coefficient of variation for leaf density in a mixed *Posidonia angustifolia*/*sinuosa* meadow. Data were randomly re-sorted from the 25 quadrats collected at Site 1D in 10 m depth during April 2004.

*Epiphyte cover*

Within sites there was a trend for increased epiphyte cover of each functional group from the base to the tip of the leaf (Figure 26). Total epiphyte cover at all locations on the leaf ranged from about 40-70% across all sites, with about twice the cover at 1S compared to 4S (Figure 26 a). Epiphyte cover at all sites was dominated by foliaceous and encrusting algae with a clear shift in the relative contribution of the two groups from north (Zone 1) to south (Zone 4; Figure 26 b, c); in the north, encrusting algae were dominant, while in the south, foliaceous algae were more dominant. There was very little cover of fauna at any of the sites (Figure 26 d). The greatest cover of fauna occurred at the shallow sites in Holdfast Bay (2S and 3S, Figure 26 d). Within zones, some trends in cover were apparent (Figure 26). While there was no clear pattern of total epiphyte cover with depth within zones (Figure 26 a), the cover of encrusting algae was greater at the deep sites compared to the shallow sites in Zones 2-4 (Figure 26 c). Conversely, there was a trend for increased faunal cover at the shallow sites over the deep sites in all four zones (Figure 26 d).

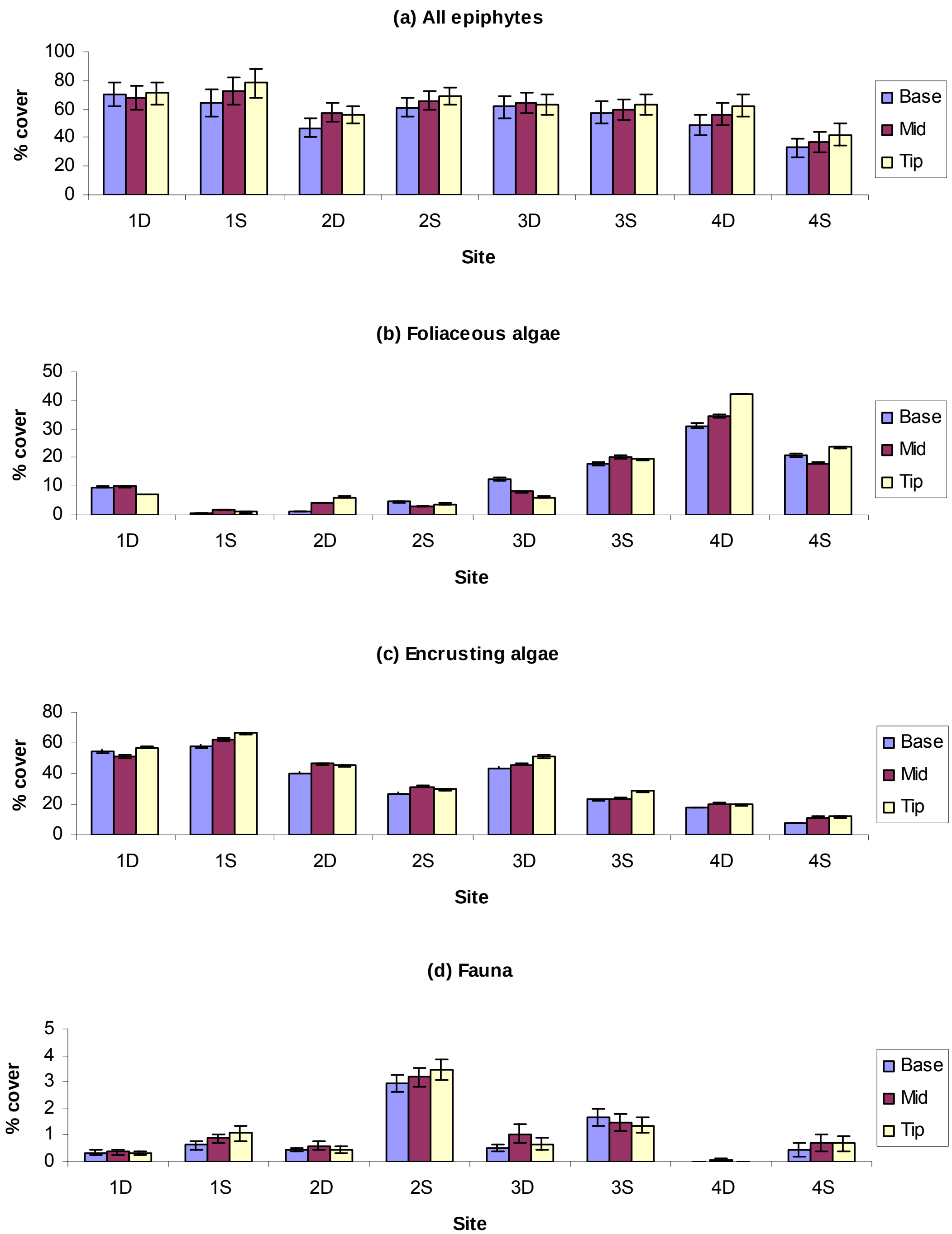


**Figure 26.** Mean (± SD) coverage of (a) all epiphytes, (b) foliaceous algae, (c) encrusting algae, and (d) fauna, on the base, mid and tip of *Posidonia* leaves at deep (D) and shallow (S) sites in Zones 1-4. n = 25 for all sites/leaf locations.

# 4. Discussion

## 4.1. Water quality

### 4.1.1. Nutrients

Previous work along the Adelaide coast has consistently shown elevated nutrient levels in the nearshore (< 5 m depth) waters between Henley Beach and Glenelg, and in localized areas adjacent to WWTP outfalls (see Section 1.1.3). In the present study, nutrients were also found to be elevated in nearshore Holdfast Bay following major rainfall events. The shore-based sites (1 m depth) generally had higher nutrient levels than the ocean sites (3, 5 and 10 m depths) during both rainfall and ambient surveys. These differences partly reflect a general difference in water quality between nearshore and offshore waters, but also reflect the effect of stormwater. Nutrient levels at the shore-based sites were generally comparable to long-term ambient levels measured in the more comparable shallow waters off Henley Beach, Glenelg and Brighton jetties by the EPA (Gaylard 2004). Nonetheless, the pattern and level of nutrient concentrations in and around the Torrens River outlet, does indicate some elevation of nutrients in adjacent shallow nearshore waters (0-3 m depth) due to stormwater input and subsequent dilution away from the point source. In particular, levels of oxidized nitrogen were abnormally high in and around the Torrens outlet, and in nearby coastal waters. However, the offshore sites (> 5 m depth, where seagrasses still occur) showed little or no evidence of elevated nutrients following the three major stormwater events that were monitored. Thus, it would appear that the input of nutrients from major stormwater flows has a minor but localized influence on the ambient levels in Holdfast Bay that are already chronically elevated in the nearshore due to constant input from treated wastewater (Gaylard 2004). However, due to a rapid assimilation of nutrients (see below), the spatial impact of nutrients from stormwater may have been underestimated by the water quality monitoring technique used.

Ambient nutrient levels at offshore sites (5 and 10 m depth) where seagrasses still occur, showed little indication of elevation when compared to the ANZECC guidelines or EPA values for nearshore waters in the Grange to Glenelg region where nutrients are known to be elevated (Gaylard 2004). It is unclear why offshore Total ammonia N levels were elevated during autumn (March); the pattern was consistent in all four zones, there had been no significant rainfall events prior to sampling, and discharge volumes from the three WWTPs are relatively low at that time of year (Wilkinson *et al.* 2003). Furthermore, Gaylard (2004) reported no significant seasonal trends in any of the nutrients measured at the metropolitan jetties over the period 1995-2002, and without more frequent sampling in our study, it is possible that any observed ‘seasonal’ differences were due to short-term temporal variability. The one inshore (intertidal) site that was monitored during our ambient surveys was clearly different from the offshore (5 and 10 m depth) sites. The elevated nutrient levels at the inshore site may be due to the inherent difference between that site (intertidal over *Heterozostera*) versus the other sites (subtidal over *Amphibolis* and/or *Posidonia*) or to the close proximity of the site to the Bolivar WWTP outfall and/or the Penrice discharge via the Port River/Barker Inlet system.

Ambient nutrient levels at the offshore sites (5 and 10 m depth) were similar to historical values for offshore waters. Levels of total phosphorus detected in the present study (mean generally ≤ 0.015 mg P $L^{-1}$; maximum = 0.052 mg P $L^{-1}$) were similar to or slightly lower than those reported by Steffensen (1985) for the area from Barker Inlet to Brighton, and were also consistent with levels reported by Lewis (1975) south of Brighton (Table 5). While comparisons of oxidized nitrogen are complicated because the historical data are restricted to nitrate and the detection limits were higher, there is some evidence that mean levels observed in the present study (range 0-0.006 mg N $L^{-1}$) were slightly lower than during the period 1972-1983 (Table 5). In addition, Steffensen (1985) reported mean nitrate values for seabed samples taken 1 km offshore of 0.009, 0.02, 0.027, and 0.035 mg N $L^{-1}$, in spring, winter, summer, and autumn, respectively. No comparisons of total nitrogen or total ammonia N were possible due to a lack of historical data.

**Table 5.** Summary of dissolved nutrient data (mg $L^{-1}$) collected in Adelaide's offshore waters during the period 1972-1983. See Appendix A for further details of each study.

| Location | Nutrient | Mean | Max. | Study |
|---|---|---|---|---|
| Seabed samples at a site 2 km offshore from Semaphore in 5 m depth | Total phosphorus | 0.02 | 0.056 | Steffensen (1985) |
| | Nitrate | <0.01 | 0.01 | |
| Seabed samples at a site 1 km offshore from Henley Beach in 4 m depth | Total phosphorus | 0.031 | 0.155 | Steffensen (1985) |
| | Nitrate | <0.01 | 0.02 | |
| Seabed samples at a site 3.5 km offshore from Brighton in 10 m depth | Total phosphorus | 0.026 | 0.14 | Steffensen (1985) |
| | Nitrate | <0.01 | <0.025 | |
| Seabed samples at a site 1 km offshore from Brighton in 5.5 m depth | Total phosphorus | 0.026 | 0.145 | Steffensen (1985) |
| | Nitrate | 0.026 | 0.2 | |
| A site off Moana | Total phosphorus | ~ 0.005-0.01 (seasonal range) | ~ 0.025 | Lewis (1975) |
| | Nitrate | Mostly <0.01 (seasonal range) | ~ 0.03 | |

Despite the relatively low nutrient levels measured at the offshore sites during 2003/04 (and thus an apparent lack of influence from land-based discharges), offshore seagrass meadows along the entire Adelaide metropolitan coastline are characterised by $\delta^{15}N$ levels indicative of industrial effluent and sewage impact; with pronounced effects in waters adjacent to the Bolivar, Glenelg and Christies Beach WWTP outfalls, and the Penrice soda factory in the Port River. This impact diminishes, and seagrass $\delta^{15}N$ returns to background levels, at sites ≥10 km to the south of the Christies Beach outfall. Tracking the northern extent of enrichment is somewhat limited in the present study, with the northern samples collected approximately 25 km northeast from the last northern metropolitan site offshore from Port Gawler. These samples are characterised by background $\delta^{15}N$ levels, suggesting that terrestrially derived nitrogen is not a significant source in the northern Gulf. Future studies should be directed at sampling seagrass within this 25 km gap to the northeast to better characterise the northern extent of influence.

Annual nitrogen discharges are similar for both the Bolivar (383 tonnes. $yr^{-1}$) and Glenelg (381 tonnes. $yr^{-1}$) WWTPs (Wilkinson *et al.* 2003). Despite this, seagrasses located within the vicinity of the Bolivar WWTP off Barker Inlet (Sites 8 and 9), have $\delta^{15}N$ significantly higher than those found in Glenelg equivalents and have some of the highest levels reported for seagrass in sewage studies (present study high of $\delta^{15}N$ =14.43 ‰ in leaf *c.f.* 6‰ in study by McClelland *et al.* 1997 and 8.7‰ in study by Constanzo *et al.* 2001). Coastal waters off Barker Inlet not only receive sewage discharges from the Bolivar WWTP, but also inputs from the highly industrialised Port River (Pattiaratchi *et al.* 2006) and from sediment remineralization of deposits associated with the now decommissioned Port Adelaide WWTP (EPA, 2005). Of particular importance are discharges from the Penrice soda factory, which has a highly elevated $\delta^{15}N$ signal. In its production of sodium carbonate, the company produces an effluent largely consisting of highly bioavailable ammonia/ammonium, which contributes the highest effluent load of nitrogen to the Port River, an amount more than double that produced by the Bolivar WWTP (~1000 tonnes $yr^{-1}$, Jeremy Wilkinson, Flinders University, pers. comm., 2006). As a consequence of these inputs, the waters of the Barker Inlet-Port River system have nutrient concentrations much higher than trigger values indicative of environmental stress, and show signs of serious ecosystem decline (EPA 2005). While significant reductions in wastewater discharges have been achieved throughout the system, with the closure of the Port Adelaide WWTP and upgrades to the Bolivar WWTP, discharges from Bolivar and Penrice are thought to still be significant enough to affect the ecosystem. It appears that the peak levels of $\delta^{15}N$ detected off Barker Inlet in the present study were due to a combination of nitrogen from Bolivar WWTP and the Penrice factory, and that the elevated levels between Outer Harbour and Semaphore (Sites 10-12) are likely due mainly to Penrice. Importantly, there was a slight decrease in $\delta^{15}N$ between Semaphore and Glenelg (at Site 13); possibly an area in-between the major regions of influence from Penrice and Glenelg WWTP discharges. These observations are supported by predictions from hydrodynamic modelling of the region as part of the ACWS (Pattiaratchi *et al.* 2006).

Based upon the overall results from the $\delta^{15}N$ survey, it is apparent that the entire coast between Port Gawler and Port Noarlunga is affected by nitrogen from WWTP and industrial discharges.

### 4.1.2. Biological activity

Chlorophyll-*a* data from the ambient surveys support the findings of the $\delta^{15}N$ survey, which showed terrestrially derived nitrogen is reaching offshore waters along the AMC. On eight occasions, mean values at the offshore sites exceeded the ANZECC guideline value of 1 $\mu g\ L^{-1}$ for marine waters in south central Australia (low rainfall area). Significantly, mean values at the shallow site in Zone 1, which is in relatively close proximity to, and under strong influence from, Bolivar WWTP and Penrice (see $\delta^{15}N$ results), exceeded the ANZECC value on three of the four surveys. Again, the elevated levels of chlorophyll-*a* detected under relatively low water column nutrient concentrations reveal the inadequacies of water quality sampling in oligotrophic waters for detecting nutrient inputs.

The maximum mean chlorophyll-*a* value of 4.4 $\mu g\ L^{-1}$ was recorded at the inshore site in Barker Inlet, where maximum levels of faecal coliforms and dissolved organic carbon were also detected. These data suggest that the inshore site was being affected by anthropogenic inputs; most likely from the nearby Bolivar WWTP outfall and Penrice in the Port River. Significantly, the inshore sites in Zones 1-3 all had low levels of faecal coliforms, providing further evidence that land-based inputs are reaching the seagrasses at those sites.

### 4.1.3. Toxicants

Water-borne toxicants are unlikely to be a current threat to seagrasses in Holdfast Bay. Only one compound at a very low concentration (simazine) out of a wide range of organochlorine and organophosphate pesticides, triazine herbicides, and the herbicide Glyphosate was detected in estuarine and marine water samples collected on three occasions following significant rainfall and stormwater flows. Yet, it is during these peak run-off events (especially after dry periods) that we might expect to detect terrestrially derived chemicals if they are present. Nonetheless, while we failed to detect virtually any potential toxicants, this does not mean that they have not been discharged to the AMC in the past. Indeed, Wilkinson *et al.* (2005) documented a range of potential toxicants (including simazine) in stormwater that have historically been discharged to the marine environment. However, Wilkinson *et al.*

(2005) also reported that toxicants are spasmodically detected and when they are, they are in very low concentrations.

### 4.1.4. Turbidity

While neither turbidity nor light attenuation were measured during the ambient and rainfall surveys, it was apparent that the nearshore waters of Holdfast Bay were highly turbid following heavy rains and stormy seas during 2003-2004. Detailed investigations of underwater light conditions were subsequently carried out in Holdfast Bay during 2005/06 and are reported in Collings *et al.* (2006b).

## 4.2. Sediment quality

There is very little historical information on sediment quality and there was minimal investigation of sediment quality as part of the present study. Nonetheless, poor sediment quality does not currently appear to be an issue for seagrass health. Only very low or undetectable levels of potential toxicants were found in marine sediment samples collected adjacent to major stormwater outlets where they are most likely to occur, and at offshore sites where terrestrially derived sediments may potentially be transported. Steffensen (1981) also reported similarly low heavy metal values of around 3 mg. $kg^{-1}$ adjacent to the Port Adelaide WWTP sludge outfall when it was operating, but heavy metals were not considered to be the cause of the massive seagrass declines associated with the outfall (Neverauskas 1987a). In fact it has been shown that seagrasses can survive with very high levels of heavy metals (Ward 1987, 1989). It is unknown why levels of heavy metals were elevated in Zone 4 in comparison to Zones 1-3. It is possible that catchment inputs via the Onkaparinga River and/or the local coastal geology of the region have an influence on levels of heavy metals in Zone 4. Nonetheless, the concentrations detected in Zone 4 were well below ANZECC (2000) guideline trigger levels and were unlikely to be causing biological harm.

Nearshore Holdfast Bay has been eutrophic for some decades now (see earlier). It is possible that under conditions of elevated nutrients, nutrient-laden sediments and organic material accumulating on the substrate could result in increased levels of porewater nutrients. While sediment nutrient levels were not measured as part of the current study, results of benthic chamber work as part of Task EP 1 (see Nayar *et al.* 2006) indicate that for seagrass environments in Holdfast Bay, sediment nutrient levels (particularly ammonia/ammonium) are much higher than in the overlying water column (S. Nayar, personal communication). However, this is probably due to the naturally high organic content of sediments in seagrass meadows (Romero *et al.* 2006), and seagrasses seem to be more tolerant of higher levels of nutrients in sediments than in water (Ralph *et al.* 2006). Regardless, no comment can be made on historical levels of nutrients in nearshore sediments during the time of initial seagrass losses.

## 4.3. Seagrass quality

This section provides a synopsis of results from the video and diver surveys with respect to each of the seagrass quality indicators. While seagrass quality parameters for *Posidonia* such as leaf/shoot density and leaf length are naturally highly variable in both space and time (Wood and Lavery 2000), it is still useful to compare historical values for *Posidonia* from the AMC (see Appendix C) to see if any patterns emerge. No comparisons could be made for *P. coriacea* in Zone 4.

*Seagrass cover*
The patterns of seagrass cover that we recorded in the Section Bank to Seacliff region (Zones 1-3) are consistent with what can be seen in aerial photographs (e.g. Figures 4, 5

and 8) and with what has been documented previously. In the northern parts of the region, coverage is generally uniform and high, but in the southern parts, coverage is much more fragmented. The spatial differences in coverage are most likely due to the increased wave energy in southern Holdfast Bay that causes greater erosion leading to blowouts and scarps (Figure 8). During 1985, Thomas and Clarke (2002) reported no blowouts along a transect from 0 to 18 m depth out from Largs Bay, but observed blowouts on all three transects to the south between Point Malcolm and Brighton at depths of 3-15 m. While blowouts occur naturally in some locations (Clarke and Kirkman 1989), it is apparent that the fragmentation in southern and central Holdfast Bay is increasing (Hart 1997) and in this particular instance, the patchy cover of seagrass is indicative of unhealthy meadows. In contrast, the seagrass we documented off Maslin Beach in the southern part of the ACWS area is naturally patchy due to the nature of the *Posidonia coriacea* that we found there.

The outer depth limit of *Posidonia* is currently around 15-18 m in Holdfast Bay and appears to have changed little over the past 40 or so years (see Section 1.1.5). However, the data are spatially limited over that time period, and data are lacking for the period prior to the 1960s when *in situ* observations were first made. Nonetheless, it does not appear that the offshore margin has been regressing shoreward in conjunction with the observed seaward regression of the nearshore margin in Holdfast Bay.

*Species composition*
*Posidonia* was the dominant species at the sites chosen for video and diver surveys. As with previous surveys (e.g. Thomas and Clarke 2002), we found that *Posidonia* meadows in Holdfast Bay are comprised of both *P. angustifolia* and *P. sinuosa*, with a tendency for *P. angustifolia* to be more prevalent in deeper waters and in the southern part of Holdfast Bay. Indeed, Clarke and Kirkman (1989) postulated that *P. sinuosa* is being replaced by *P. angustifolia* in the central and southern parts of Holdfast Bay. In contrast, in the mid-part of Zone 4 we documented only *P. coriacea*, which prefers high wave energy environments. Nonetheless, during pilot work in Zone 4, we also documented *P. angustifolia* at 18 m depth directly offshore from the 5 and 10 m survey sites.

Of the nine sites surveyed, *Amphibolis* was documented at just the two 5 m depth sites in Zones 2 and 3. While *Amphibolis* was once a major component of the benthos to 12 m depth between Port Gawler and Seacliff (see earlier), it was not detected at the two sites out from the Section Bank or at the two 10 m depth sites in Holdfast Bay. Whilst the broad-scale spatial coverage of our survey was quite limited, other observations suggest that *Amphibolis* is now quite rare in Holdfast Bay. For example, as part of a seagrass rehabilitation project, researchers searched specifically during 2005 for *Amphibolis* beds between Largs Bay and Brighton in about 7-9 m depth, but found no *Amphibolis* between Henley Beach and Brighton (Rachel Wear, unpublished data). While there are still substantial *Amphibolis* meadows in nearshore waters just south of Brighton off Kingston Park and between Semaphore and Henley Beach, it appears that *Amphibolis* has largely disappeared from the nearshore area between Henley Beach and Brighton. Significantly, this area also contains the Glenelg WWTP outfall, and the Patawalonga and Torrens River outlets. It appears that *Amphibolis* has disappeared from the Section Bank area also. Again, this area is in close proximity to major land-based discharges: the Bolivar WWTP outfall and the Penrice soda plant outfall. It is believed that due to structural and physiological differences, *Amphibolis* is more susceptible than *Posidonia* to poor water quality (Shepherd *et al.* 1989, Ralph *et al.* 2006). Thus, in this particular instance, the absence of *Amphibolis* from areas where it was once abundant is probably indicative of an unhealthy seagrass system.

*Aboveground biomass*
Aboveground biomass in the present study (accepting that seasonal variability has not been accounted for) is within the range of previously recorded values for apparently healthy *Posidonia angustifolia*/*sinuosa* meadows, and indicates that the current status of the

*Posidonia* meadows sampled in Zones 1-3 is good. Average aboveground biomass of *Posidonia* (comprising *P. angustifolia* and *P. sinuosa*) varied from about 650 to 1450 g wet wt.$m^{-2}$ in depths of 5 and 10 m in Holdfast Bay (i.e. Zones 2 and 3). Shepherd (1970) reported values of around 800 g wet wt.$m^{-2}$ for *Posidonia* in depths of ca. 5 m in Holdfast Bay during 1968. However, at the blue-line of the time and in partially degraded areas, Shepherd (1970) reported values of just 100 g wet wt.$m^{-2}$. Lewis (1975) measured values of around 600 g wet wt.$m^{-2}$ for sites ca. 2 km offshore (equating to around 5 m depth) in Holdfast Bay between 1972-1975. As with Shepherd (1970), Lewis (1975) found lowest mean values (around 200-500 g wet wt.$m^{-2}$) along the blue-line of the time and at those sites 500 m from shore (Appendix C). Prior to the opening of the Port Adelaide WWTP sludge outfall, Lewis (1975) recorded values of between 600-700 g wet wt.$m^{-2}$ in ca. 10 m depth, and between ca. 1200-1600 g wet wt.$m^{-2}$ in 8-20 m depth off Glenelg during 1975.

*Leaf density*

While *Posidonia* leaf density does vary seasonally (e.g. Wood and Lavery 2000), results of the present study indicate that leaf densities at the sites surveyed were within the expected range of healthy *Posidonia* meadows in the Port Gawler-Marino region. Mean leaf density of *Posidonia angustifolia*/*sinuosa* varied from about 1300-2300 leaves.$m^{-2}$ at depths of 5 and 10 m across Zones 1-3. During the early 1980's, Neverauskas (1987a) recorded mean values of ca. 2000-2500 leaves.$m^{-2}$ across a range of depths from 5-13.5 m seaward of Brighton in Zone 3. Also during the early 1980s, Neverauskas (1988) found a mean leaf density of 1650 leaves.$m^{-2}$ in 11-12 m depth offshore from Port Gawler in Zone 1. In a study of the Port Adelaide WWTP sludge outfall during the 1980's, Neverauskas (1987b) reported mean values of ca. 1000-2300 leaves.$m^{-2}$ at control sites in 10-15 m depth away from the outfall, offshore from Point Malcolm in Zone 2.

*Shoot density*

As shoot density varies less seasonally than leaf density, it can be a better indicator of seagrass health at any given point in time. In the present study, mean shoot density of *Posidonia angustifolia*/*sinuosa* varied from about 700-1200 shoots.$m^{-2}$ in depths of 5 and 10 m across Zones 1-3. During the early 1980's, Neverauskas (1987a, 1988) recorded mean values of ca. 1000-1300 shoots.$m^{-2}$ across a range of depths from 5-13.5 m out from Brighton in Zone 3, and a mean value of 1100 shoots.$m^{-2}$ in 11-12 m depth offshore from Port Gawler in Zone 1. Our results indicate that shoot density at the shallow site in Zone 1 (mean = 735 shoots.$m^{-2}$) is below the expected range of healthy *Posidonia* meadows in the Port Gawler-Marino region. However, no conclusions can be made in the absence of additional historical data.

*Leaf length and maximum leaf length*

While measures of leaf length are highly dependent on season (e.g. Wood and Lavery 2000), results for maximum and mean leaf lengths from the present study generally fall within the ranges of what are considered to be normal or healthy *Posidonia* meadows in Holdfast Bay. In the present study, canopy height (or maximum leaf length) of *Posidonia angustifolia*/*sinuosa* was about 30-50 cm in depths of 5 and 10 m across Zones 1-3. Shepherd (1970) reported a canopy height of ca. 60 cm in apparently healthy *Posidonia* meadows from ca. 5 to 15 m depth in Holdfast Bay during 1968. Shepherd (1970) also found that in degraded areas, *Posidonia* leaf length rarely reached 30 cm. There appear to be no other local reports of maximum leaf length. In the present study, mean leaf length of *Posidonia angustifolia*/*sinuosa* was about 17-41 cm in depths of 5 and 10 m across Zones 1-3. During 1974-1975, Steffensen (1985) recorded mean leaf lengths of 36-43 cm in ca. 5-7 m depth and mean leaf lengths of 25 and 23 cm in deeper sites of 10 and 12 m for *P. sinuosa* off Point Malcolm in Zone 2. Neverauskas (1987a) reported a mean leaf length of 32 cm for mixed *P. angustifolia*/*sinuosa* offshore from Port Gawler (i.e., Zone 1) in 11-12 m depth during the 1980's. During 1996, Harbison and Wiltshire (1997) measured mean leaf lengths of 17-21 cm for *P. sinuosa* in 3-7 m depth between Largs Bay and Brighton. The lowest value of 17 cm for mean leaf length in the present study was found at the deep site in Zone 3, possibly suggesting that this meadow was under some stress.

*Epiphyte composition and cover*

Increased epiphyte loads have previously been correlated with WWTP outfalls off Adelaide (e.g. Shepherd 1970, Glenelg WWTP; Shepherd *et al.* 1989, Bolivar WWTP; Neverauskas 1987b, Port Adelaide WWTP). It is postulated that the increased nutrients from the outfalls cause an increase in epiphyte load that can ultimately lead to the demise of the host seagrass (Shepherd *et al.* 1989, Ralph *et al.* 2006). Surveys in the present study showed no strong indication of increased epiphyte cover at the shallow sites in Zones 1-3 where the greatest influence from land-based inputs should potentially occur. Whilst epiphyte cover is poorly correlated with epiphyte load (Gonzalez 2005), qualitative observations from the video surveys and quantitative measures from other work (Lill 2005) also do not indicate excessive loads within Holdfast Bay. However, Gonzalez (2005) recently documented extremely high epiphyte loads on *Posidonia australis* adjacent to the Bolivar WWTP outfall. Thus it appears that the WWTP-derived nutrients reaching the remaining *Posidonia* meadows off Adelaide (as evidenced by the $\delta^{15}N$ work) are causing excessive epiphytic loads near Bolivar, but not in Holdfast Bay. This could be due to a number of factors including distance to outfalls, hydrodynamics, and depth (light availability). The effects of increased epiphyte load on seagrass health are being investigated as part of Task EP 1 (see Collings *et al.* 2006a).

While there was no apparent trend in total epiphyte cover between sites, several other trends were apparent. The observed trend of increased epiphyte cover from the base to the tips of *Posidonia* leaves can be explained by the basal growth pattern of *Posidonia* whereby the tips of the leaves are older than the bases and therefore have a greater amount of time to accumulate epiphytes (Borowitzka and Lethbridge 1989). There was also a greater cover of epiphytic fauna at the shallow sites in Holdfast Bay. Nonetheless, the cover was only very low (2-3 %). The shift from encrusting algae to foliaceous algae from north to south probably reflects the far greater coverage of nearby reefs in the south (Bryars 2003) and thus an increased epiphyte propagule supply. In summary, the epiphyte cover that we documented at the ambient sites does not indicate an anthropogenic impact. However, sites in closer proximity to land-based discharges were not specifically surveyed as part of our study, but may show increased epiphyte loads (e.g. Bolivar WWTP).

# 5. Conclusions and recommendations

*Water, sediment and seagrass quality*

Due to the various coastal inputs operating over the past 60 years, Adelaide's coastal waters are no longer pristine, with elevated levels of nutrients, toxicants, and turbidity regularly being reported over the last 30 years. In association with the decline in water quality, major losses of seagrass have occurred in several discrete locations along the AMC. While results from our surveys suggest that the remaining offshore seagrasses are healthy, subtle indicators such as meadow fragmentation and the absence of *Amphibolis*, suggest the contrary. In this context, it is apparent that offshore coastal water quality (where the remaining seagrasses occur) is still being affected by WWTP and industrial discharges. While the impact of any potential current or historical stressors on Adelaide's seagrass systems remains unclear, we can make some predictions about the likelihood of whether they contributed to historical and/or ongoing losses and thus require further investigation.

*Nutrients*

Historical data show that the nearshore (< 5 m depth) waters in Holdfast Bay have consistently had elevated levels of nutrients since at least the 1970s (when monitoring first began), and must be considered as eutrophic in the context of an oligotrophic system. The area of most pronounced levels of elevated nutrients, between Glenelg and Grange, coincides with the area of major nearshore seagrass losses in Holdfast Bay. Based on historical data for coastal inputs to Holdfast Bay from wastewater and stormwater (Wilkinson 2005, Wilkinson *et al.* 2003, 2005 a, b), it is apparent that nearshore nutrient levels would have been elevated prior to the 1970s also. While historical data and results from our ambient surveys show little indication of elevated water column nutrient levels offshore where seagrasses still occur, it is apparent from results of the $\delta^{15}N$ survey that substantial amounts of nitrogen from WWTPs and industrial discharges are reaching these seagrasses. This clearly demonstrates the inadequacies of only monitoring water column nutrients in oligotrophic environments, as a lack of elevated inorganic nutrients does not indicate a lack of available nutrients; the nutrients detected are those in excess of biological consumption and cycling. As an example, during the 1980s when the Port Adelaide WWTP sludge outfall was fully operational in 12 m depth off Semaphore, elevated nutrients could only be detected within close proximity (< ca. 500 m) of the sludge discharge point (Neverauskas 1987*a*; see Appendix A). Yet devastating effects on seagrasses due to elevated nutrients (rather than turbidity) from the discharge were detected many kilometres away (Neverauskas 1987a). Conversely, detection of elevated inorganic nutrients in an oligotrophic system, such as the ACWS region, is a clear indication that the system cannot assimilate the nutrient loads being delivered and is of particular concern.

The $\delta^{15}N$ survey indicates that seagrasses spanning the entire coastline of Adelaide from Port Gawler to Port Noarlunga are impacted by treated sewage and industrial discharges; the degree of impact being most severe in proximity to wastewater outfalls, with highest levels recorded adjacent to Barker Inlet where the combined discharges from Bolivar WWTP and Penrice soda factory are probably operating. Results of the $\delta^{15}N$ survey could be used to identify seagrass meadows of highest environmental risk and direct future management strategies aimed at reducing nitrogen and thus preventing future seagrass loss along the Adelaide metropolitan coastline. Perhaps the most surprising result of the $\delta^{15}N$ survey is that the remaining seagrasses in Holdfast Bay are still being affected by WWTP and industrial discharges; despite a reduction in annual nitrogen loads (Wilkinson *et al.* 2003) and their distance of several kilometres from nutrient inputs. Clearly then, the entire nearshore zone of denuded seagrass in Holdfast Bay would have been influenced by nutrient discharges from Glenelg WWTP and/or Penrice soda factory in the past.

While results of the $\delta^{15}N$ survey indicate that seagrasses along the entire AMC are receiving nutrients derived from wastewater outfalls, the *Posidonia* meadows sampled at the offshore ambient sites appeared healthy in terms of seagrass biomass and epiphyte load. Thus it would appear that levels of nutrients received from wastewater (and possibly stormwater) discharges are not having a detrimental impact on established *Posidonia* meadows at those sites. This conclusion is in agreement with recent mapping work indicating a stabilisation of the offshore regression of seagrasses in Holdfast Bay (Hart 1997, Cameron 2003). While fragmentation of seagrass beds is continuing in central and southern Holdfast Bay, this is probably linked to physical processes that may have been triggered by seagrass loss itself, rather than to land-based discharges. Nonetheless, current land-based discharges could be playing a major role in the fragmentation process if they negatively impact on seagrass meadow spreading and colonisation dynamics (see Clarke 1987).

In summary, it is clear that the effects of elevated nutrients on seagrass health warrant further investigation in the context of historical and ongoing nearshore seagrass losses in Holdfast Bay (for further work see Collings *et al.* 2006a).

*Toxicants*
Toxicants were unlikely to have been responsible for broad-scale historical seagrass losses for the following reasons:

- Toxicants have only been sporadically detected in very low concentrations in freshwater entering the AMC (Wilkinson *et al.* 2005a; results of present study).
- Concentrations required to affect seagrass physiological processes are relatively high (Westphalen *et al.* 2004), and due to rapid dilution in the marine environment, the historical levels detected in stormwater are unlikely to have reached levels capable of having an impact.
- We found all toxicants to be undetectable in the coastal waters off Adelaide following peak stormwater flows when it may be most likely to detect them.
- We found very low or undetectable levels of potential toxicants in marine sediment samples collected adjacent to major stormwater outlets where they may be most likely to occur and at offshore sites where terrestrially derived sediments may potentially be transported.

*Turbidity*
Further investigations of the effects of decreased light on seagrass health are presented in Collings *et al.* (2006b).

*Recommendations for future monitoring/research*
Based upon outcomes of the present study, the following activities are recommended:

- Experimentally test and model the effects of increased nutrients and turbidity on *Amphibolis* and *Posidonia*.
- Survey and commence long-term monitoring of seagrass quality at sites adjacent to land-based discharges and at suitable control sites.
- Survey and commence long-term monitoring of the outer depth margin of *Posidonia* meadows in Holdfast Bay.
- Survey and commence long-term monitoring of seagrass meadow fragmentation at a range of sites in Holdfast Bay.
- Conduct a detailed survey of the current distribution of *Amphibolis* between Port Gawler and Sellicks Beach.
- Conduct a spatially intensive $\delta^{15}N$ survey to determine the offshore and northern extents of nitrogen influence from wastewater outfalls along the AMC, and also characterise $\delta^{15}N$ signatures of potential nitrogen sources.

*Recommendations for management*

- Nutrient loads entering Adelaide's coastal waters need to be reduced in order for the system to have any chance of returning to its natural oligotrophic state.

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# 7. Appendices

**Appendix A.** Summary of studies investigating water quality along the Adelaide metropolitan coastline. Different sections show different parameters. AMC = Adelaide metropolitan coast from Port Gawler to Sellicks Beach; PAWWTP = Port Adelaide Wastewater Treatment Plant; GWWTP = Glenelg Wastewater Treatment Plant; BWWTP = Bolivar Wastewater Treatment Plant; Note that studies conducted in the Port River are not included, and that this list is not exhaustive.

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| **Nutrients** | | | | |
| Lewis (1975) | Middle Beach to St Kilda.<br>33 sites at 1000 and 2000m offshore. | Phosphate and Nitrate (µg $L^{-1}$) | June 1973 to August 1974, sampled approximately monthly. | Nutrient values generally low within the region with the exception of sites in close proximity to the BWWTP outfall. Nitrogen levels of up to 600 µg $L^{-1}$ and total phosphorus levels of up to 370 µg $L^{-1}$ recorded 500 m offshore from the outfall. |
| Lewis (1975) | Outer Harbor to Marino.<br>37 sites at 500, 1000, and 2000m offshore. | Phosphate and Nitrate (µg $L^{-1}$) | Summer 1972/73 to winter 1974, sampled approximately monthly. | Nitrogen levels generally low north of Henley Beach and south of Seacliff and during summer months. Higher levels of nitrate were recorded during winter months (generally between 40 and 120 µg $L^{-1}$) with the highest levels recorded in surface waters off Glenelg (up to 400 µg $L^{-1}$).<br>As with nitrate, phosphorus levels increased following significant storm water run-off (several peaks above 200 µg $L^{-1}$ during winter 1973) but were generally higher than nitrate levels. |
| Lewis (1975) | Marino to Sellicks Beach.<br>22 sites at 500 and 1000m offshore. | Phosphate and Nitrate (µg $L^{-1}$) | September 1973 to August 1974, sampled approximately 6-weekly. | Generally very low nutrient concentrations. Nitrogen generally less than 30 µg $L^{-1}$ and phosphorus generally less than 20 µg $L^{-1}$. Small peaks of 85µg $L^{-1}$ nitrogen and 67 µg $L^{-1}$ phosphorus recorded in winter 1974 at Christies Beach. Unusually high levels of nitrate (up to 170 µg $L^{-1}$) offshore from Sellicks Beach and Aldinga in Autumn 1974. |
| Lewis (1975) | Jetties: Largs Bay, Henley Beach, Glenelg, | Phosphate and Nitrate (µg $L^{-1}$) | 1973 and 1974, sampled monthly. | Generally higher levels in winter and during wet periods than in summer and other dry |

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| | Brighton, Port Noarlunga. | | | months. Glenelg and Brighton jetties contained high phosphorus and very high nitrogen levels (566 µg $L^{-1}$); both are adjacent a large number of storm water outlets. |
| Steffensen (1985) | Christies Beach. 14 sites at and around CBWWTP outfall | TKN (mg $L^{-1}$), Ammonia (mg $L^{-1}$), Nitrate (mg $L^{-1}$), Phosphorus (mg $L^{-1}$) | April 1978 | Elevated levels of nutrients at outfall (TKN=1.28, NO3=0.89, TP=0.25) and within at least 100m of outfall (TKN=0.64-1.15, NO3=0.33-0.61, TP=0.012-0.16). No perceptible increase in ammonia due to the outfall. |
| Steffensen (1985) | Outer Harbour to Marino. 37 locations; jetties plus 500 and 1000m offshore. Surface and bottom samples | Total phosphorus (mg $L^{-1}$), Nitrate (mg $L^{-1}$) | 1972 to 1983 Includes some results from Lewis (1975). | TP tended to be lower offshore and high immediately adjacent the GWWTP outfall. Higher values in bottom samples probably indicate sediment resuspension. Marked seasonal variation in nutrients: mean TP highest in winter (0.08 mg $L^{-1}$) and lowest in summer (0.02 mg $L^{-1}$); mean nitrate highest in winter (0.09 mg $L^{-1}$). TP and nitrate concentrations show peaks adjacent the GWWTP outfall and decline with distance to the north and south (particularly during summer). Maximum bottom concentrations of at least 0.14 mg $L^{-1}$ TP and 0.9 mg $L^{-1}$ nitrate. |
| Steffensen (1985) | Adjacent Glenelg WWTP outfall, 13-14 sites to examine dispersion of effluent | Ammonia, Nitrate, Nitrite, TKN, Soluble P, Total P | November 1980 (surface samples only) August 1982 (surface and bottom samples) | Dispersion patterns clearly evident in surface samples but much less so in bottom samples. In 1980: peak values (mg $L^{-1}$) at surface of ammonia 3.15, nitrate 1.18, nitrite 0.02, and TP 1.185. In 1982: peak values (mg $L^{-1}$) at surface of ammonia 0.254, nitrate + nitrite 1.48, and TP 0.597, and at bottom of ammonia 0.019, nitrate + nitrite 0.03, and TP 0.091. |
| Steffensen (1985) | Jetties between Brighton and Largs Bay. Offshore sites from the Patawalonga | Ammonia, Nitrate, TKN, Soluble P, | February 1976 March 1983 | In 1976: jetty data showed very little impact from stormwater (peak nitrate of 0.02 mg $L^{-1}$ Grange and Henley, peak TP of 0.02 mg $L^{-1}$ at Glenelg), but offshore sites (up to 100m) |

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| | and Torrens River. Surveys designed to detect coastal influence of two major stormwater events. | Total P | | showed some elevated values (peak ammonia of 0.16 mg $L^{-1}$, peak nitrate of 0.16 mg $L^{-1}$, peak TP of 0.099 mg $L^{-1}$).<br>In 1983: both jetty and offshore data showed very little impact of stormwater. |
| Neverauskas (1987a) | AMC - Port Adelaide WWTP sludge outfall | Ammonia | Two surveys during summers of 1984 and 1985. | Values as high as 590 µg $L^{-1}$ directly in the plume of sludge were recorded. The amount of ammonia generally decreased with distance from the outfall (300 m away, but still within the plume, mean values were 157, 19 and 25 µg $L^{-1}$, while 500 m away values dropped to 15 µg $L^{-1}$). |
| Neverauskas (1987a) | AMC - Port Adelaide WWTP sludge outfall | Oxidized nitrogen | Two surveys during summers of 1984 and 1985. | No elevation of oxidized nitrogen in the vicinity of the outfall. |
| Neverauskas (1987a) | AMC - Port Adelaide WWTP sludge outfall | Soluble phosphorus | Two surveys during summers of 1984 and 1985. | Values as high as 127 µg $L^{-1}$ directly in the plume of sludge. The amount of phosphorus decreased with distance from the outfall (300 m away, but still within the plume, mean concentrations varied between 29 and 4 µg $L^{-1}$, while 500 m away mean values were recorded as 16 µg $L^{-1}$). |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Total phosphorus (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean total phosphorus concentrations at all Adelaide jetties and the Port Hughes jetty varied between 0.025 mg $L^{-1}$ (Port Noarlunga) and 0.064 mg $L^{-1}$ (Port Hughes). A maximum reading of 0.68 mg $L^{-1}$ was observed off the Grange jetty. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Oxidized nitrogen (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean oxidized nitrogen concentrations at all Adelaide jetties and the Port Hughes jetty varied between 0.013 mg $L^{-1}$ (Largs Bay and Semaphore) and 0.037 mg $L^{-1}$ (Henley Beach). A maximum reading of 0.349 mg $L^{-1}$ was observed off the Port Noarlunga jetty. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port | Ammonia (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean ammonia concentrations at all Adelaide jetties and the Port Hughes jetty varied between 0.027 mg $L^{-1}$ (Brighton) and 0.044 mg $L^{-1}$ (Glenelg). Maximum readings at |

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| | Noarlunga (plus Port Hughes) | | | Adelaide jetties varied between 0.163 mg $L^{-1}$ (Semaphore) and 0.202 mg $L^{-1}$ (Largs Bay). At Port Hughes a maximum reading of 0.275 mg $L^{-1}$ was found. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Total nitrogen (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean total nitrogen concentrations at all Adelaide jetties varied between 0.252 mg $L^{-1}$ (Port Noarlunga) and 0.356 mg $L^{-1}$ (Henley Beach). A maximum value of 1.337 mg $L^{-1}$ was observed at Glenelg. |
| **Salinity** | | | | |
| Lewis (1975) | Middle Beach to St Kilda. 33 sites at 1000 and 2000m offshore. | Conductivity (µS/cm/25°C) | June 1973 to August 1974, sampled approximately monthly. | Predictable seasonal variations. Levels in summer and other dry periods generally ranged between 57000 (37.9 ‰) and 59000 µS/cm/25°C (39.4 ‰), in winter values ranged between 51000 (33.5 ‰) and 57000 µS/cm/25°C (37.9 ‰). |
| Lewis (1975) | Outer Harbor to Marino. 37 sites at 500, 1000, and 2000m offshore. | Conductivity (µS/cm/25°C) | Summer 1972/73 to winter 1974, sampled approximately monthly. | Large variation (up to 17%) between summer and winter reflecting increased evaporation during summer and the effects of stormwater run-off during winter. Very little spatial variability with bottom levels always exceeding 50000 µS/cm/25°C (32.7 ‰). |
| Lewis (1975) | Marino to Sellicks Beach. 22 sites at 500 and 1000m offshore. | Conductivity (µS/cm/25°C) | September 1973 to August 1974, sampled approximately 6-weekly. | Conductivity values were consistently high, generally between 56000 - 58000µS/cm/25°C (37.2-38.7 ‰). Increased flows during winter 1974 resulted in a 4% decrease in conductivity between the Onkaparinga River and Christies Beach. |
| Lewis (1975) | Jetties: Largs Bay, Henley Beach, Glenelg, Brighton, Port Noarlunga. | Conductivity (µS/cm/25°C) | 1973 and 1974, sampled monthly. | Little spatial or temporal variability in conductivity, however, lowest levels were recorded during spring 1974 when heavy rains resulted in strong storm water flows. |
| Lewis (1975) | Bolivar WWTP outfall | Conductivity (µS/cm/25°C) | Effluent dispersion study, 17 April 1974 | Minor reductions in salinity associated with effluent; 2% reduction ca. 500m offshore from outfall. |
| Lewis (1975) | Glenelg WWTP outfall | Conductivity (µS/cm/25°C) | Effluent dispersion studies, 2-3 May 1974 | Minor and localized reductions in salinity associated with effluent; maximum reduction |

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| | | | | was 13.6% at 3m from outfall in surface waters, with virtually no decrease in salinity in bottom waters. |
| Steffensen (1985) | Outer Harbour to Marino. 37 locations; jetties plus 500 and 1000m offshore. Surface and bottom samples | Salinity (TDS) | 1972 to 1983 Includes some results from Lewis (1975). | Surface samples were consistently lower than bottom samples indicating an influence from land-based discharges. |
| Steffensen (1985) | Adjacent Glenelg WWTP outfall, 13-14 sites to examine dispersion of effluent | Total dissolved salts (mg $L^{-1}$) | November 1980 (surface samples only) August 1982 (surface and bottom samples) | In 1980: very minor decrease in salinity at surface (33 000 compared to background of 38 000 mg $L^{-1}$). In 1982: very minor decrease in salinity at surface (ca. 35 000 compared to background of ca. 37 000 mg $L^{-1}$), with virtually no decrease in salinity at bottom. |
| Steffensen (1985) | Jetties between Brighton and Largs Bay. Offshore sites from the Patawalonga and Torrens River. Surveys designed to detect coastal influence of two major stormwater events. | Total dissolved salts (mg $L^{-1}$) | February 1976 March 1983 | In 1976: jetty and offshore data showed very little impact from stormwater (38 000 mg $L^{-1}$ compared to background of 39 000 mg $L^{-1}$) In 1983: jetty data indicated a minor impact of stormwater at Henley and Grange. |
| **Turbidity** | | | | |
| Lewis (1975) | Middle Beach to St Kilda. 33 sites at 1000 and 2000m offshore. | Turbidity (JTU) | June 1973 to August 1974, sampled approximately monthly. | Generally turbidity ranged between 2 and 6 JTU. The highest readings were recorded during winter 1974 when substantial quantities of muddy stormwater were discharged into the area (values reached 7 JTU). |
| Lewis (1975) | Outer Harbor to Marino. 37 sites at 500, 1000, and 2000m offshore. | Turbidity (JTU) | Summer 1972/73 to winter 1974, sampled approximately monthly. | Turbidity generally between 1 and 2 JTU, but higher between Glenelg and Marino during winter 1974 (3-6 JTU). |
| Lewis (1975) | Marino to Sellicks Beach. 22 sites at 500 and | Turbidity (JTU) | September 1973 to August 1974, sampled approximately 6- | Generally low turbidity, values generally ranged between 1 and 2 JTU, while no readings exceeded 3 JTU. |

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| | 1000m offshore. | | weekly. | |
| Lewis (1975) | Jetties: Largs Bay, Henley Beach, Glenelg, Brighton, Port Noarlunga. | Turbidity (JTU) | 1973 and 1974, sampled monthly. | Considerably higher than those values recorded offshore between Henley Beach and Marino with several mean values between 10 and 20 JTU off some jetties. |
| Steffensen (1985) | Christies Beach. 14 sites at and around CBWWTP outfall | Turbidity (NTU) | April 1978 | No pattern evident (range of 1-17 NTU). |
| Steffensen (1985) | Outer Harbour to Marino. 37 locations; jetties plus 500 and 1000m offshore. Surface and bottom samples | Turbidity (NTU), Suspended solids (mg $L^{-1}$) | 1972 to 1983 Includes some results from Lewis (1975). | Higher turbidity values in bottom samples indicates resuspension of sediments. Marked seasonal variation in mean turbidity with highest values in winter (12.7 NTU) and lowest in summer (0.5 NTU). Marked seasonal variation in mean suspended solids with highest values in winter (110.6 mg $L^{-1}$) and lowest in summer (50.7 mg $L^{-1}$). |
| Steffensen (1985) | Jetties between Brighton and Largs Bay. Offshore sites from the Patawalonga and Torrens River. Surveys designed to detect coastal influence of two major stormwater events. | Turbidity (NTU) | February 1976 March 1983 | In 1976: jetty data showed no impact from stormwater In 1983: offshore data showed no impact from stormwater. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Turbidity (NTU) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean turbidity levels were highest at Brighton where a value of 5.47 NTU was recorded. Other mean turbidity measurements off other jetties include: Largs Bay: 3.226 NTU, Semaphore: 2.367 NTU, Grange: 2.858 NTU, Henley Beach: 3.725 NTU, Glenelg: 3.613 NTU, Port Noarlunga: 0.879 NTU and Port Hughes: 1.363 NTU. A maximum of 72 NTU was recorded off the Brighton Jetty on 3 July 1995. |
| Lewis (1975) | Outer Harbor to Marino. 37 sites at 500, 1000, and 2000m offshore. | Suspended Solids (mg $L^{-1}$) | Summer 1972/73 to winter 1974, sampled approximately monthly. | Generally low (40-50 mg $L^{-1}$), but there was a moderate peak of 76 mg $L^{-1}$ observed off the Patawalonga Creek outlet in winter 1973. |

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| Lewis (1975) | Jetties: Largs Bay, Henley Beach, Glenelg, Brighton, Port Noarlunga. | Suspended Solids (mg $L^{-1}$) | 1973 and 1974, sampled monthly. | Suspended solids were considerably higher than those values recorded offshore between Henley Beach and Marino, with values during winter commonly exceeding 150 mg $L^{-1}$. |
| **Toxicants** | | | | |
| Steffensen (1981) | AMC - Port Adelaide WWTP sludge outfall | Cadmium (mg $L^{-1}$) | February 1979 | Cadmium concentrations in water at a range of depths surrounding the outfall varied between <0.0001 and 0.0005 mg $L^{-1}$. |
| Steffensen (1981) | AMC - Port Adelaide WWTP sludge outfall | Lead (mg $L^{-1}$) | February 1979 | The maximum lead concentration in water surrounding the outfall was 0.0002 mg $L^{-1}$, all other values were below detectable (<0.0002 mg $L^{-1}$). |
| Steffensen (1981) | AMC - Port Adelaide WWTP sludge outfall | Zinc (mg $L^{-1}$) | February 1979 | Zinc concentrations in water at a range of depths surrounding the outfall varied between 0.038 and 0.003 mg $L^{-1}$. |
| Steffensen (1981) | AMC - Port Adelaide WWTP sludge outfall | Copper (mg $L^{-1}$) | February 1979 | Copper concentrations in water at a range of depths surrounding the outfall varied between 0.004 and 0.015 mg $L^{-1}$. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Soluble aluminium (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean concentrations at all Adelaide jetties as well as Port Hughes varied between 0.032 mg $L^{-1}$ (Port Noarlunga) and 0.036 mg $L^{-1}$ (Port Hughes). A maximum level of 0.213 mg $L^{-1}$ was observed at Henley Beach. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Total aluminium (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean total aluminium concentrations at all Adelaide Jetties as well as Port Hughes varied between 0.046 mg $L^{-1}$ (Port Noarlunga) and 0.098 mg $L^{-1}$ (Brighton). A maximum level of 0.862 mg $L^{-1}$ was observed at Henley Beach. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port | Total nickel (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean total nickel concentrations at all Adelaide jetties and Port Hughes was 0.006 mg $L^{-1}$ with the exception of Port Noarlunga, which had a mean of 0.007 mg $L^{-1}$. A maximum of 0.075 mg $L^{-1}$ was observed at |

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| | Hughes) | | | Port Noarlunga. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Copper (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean copper concentrations at all Adelaide jetties and Port Hughes was 0.010 mg $L^{-1}$. A maximum copper concentration of 0.022 mg $L^{-1}$ was recorded at Brighton. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Lead (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean lead concentrations at all Adelaide jetties and Port Hughes was 0.005 mg $L^{-1}$. A maximum value of 0.013 mg $L^{-1}$ was observed at Glenelg and Brighton jetties. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Total chromium (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean total chromium concentrations at all Adelaide jetties and Port Hughes was 0.010 mg $L^{-1}$ with the exception of Henley Beach, which had a mean of 0.011 mg $L^{-1}$. A maximum of 0.09 mg $L^{-1}$ was observed at Henley Beach. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Total zinc (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean total zinc concentrations off Adelaide's jetties varied between 0.031 mg $L^{-1}$ (Largs Bay) and 0.037 mg $L^{-1}$ (Henley Beach). Maximum concentrations at each site varied between 0.174 mg $L^{-1}$ (Semaphore) and 0.288 mg $L^{-1}$ (Port Hughes). |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Soluble zinc (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean soluble zinc concentrations off Adelaide's jetties varied between 0.018 mg $L^{-1}$ (Largs Bay) and 0.022 mg $L^{-1}$ (Port Noarlunga). Maximum concentrations at each site varied between 0.162 mg $L^{-1}$ (Semaphore) and 0.064 mg $L^{-1}$ (Glenelg). |
| **Biological activity** | | | | |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Enterococci (cells/100mL) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean enterococci per 100mL at all Adelaide jetties varied between 0.886 (Port Noarlunga) and 2.188 (Brighton). A maximum count of 160 cells/100mL was found at Semaphore. Port Hughes had a mean count of 1.063 cells/100mL. |

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | *Escherichia coli* (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean *E. coli* concentration at all Adelaide jetties varied between 0.37 mg $L^{-1}$ (Semaphore) and 1.71 mg $L^{-1}$ (Glenelg). A maximum reading of 80 mg $L^{-1}$ was found at Largs Bay, while Port Hughes was found to have a mean of 0.85 mg $L^{-1}$ and a maximum of 120 mg $L^{-1}$. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Faecal coliforms (mg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean faecal coliform concentration at all Adelaide jetties varied between 0.495 mg $L^{-1}$ (Semaphore) and 2.125 mg $L^{-1}$ (Glenelg). A maximum reading of 310 mg $L^{-1}$ was found at Grange, while Port Hughes was found to have a mean of 0.805 mg $L^{-1}$ and a maximum of 510 mg $L^{-1}$. |
| Gaylard (2004) | Metropolitan jetties: Largs Bay, Semaphore, Grange, Henley Beach, Glenelg, Brighton, Port Noarlunga (plus Port Hughes) | Chlorophyll *a* (µg $L^{-1}$) | February 1995 to July 2002, samples taken every 2-4 weeks. | Mean chlorophyll-a concentrations at all Adelaide jetties varied between 4.54 µg $L^{-1}$ (Henley Beach) and 1.15µg $L^{-1}$ (Port Noarlunga). At Port Hughes, mean concentration was 0.80 µg $L^{-1}$. Maximum values between 6.13 µg $L^{-1}$ (Port Noarlunga) and 24.95 µg $L^{-1}$ (Henley Beach) were recorded. |

**Appendix B.** Summary of studies investigating sediment quality along the Adelaide metropolitan coastline. PAWWTP = Port Adelaide Wastewater Treatment Plant. Note that this list is not exhaustive.

| Author | Location | Parameters | Date | Main outcomes relevant to ACWS |
|---|---|---|---|---|
| Steffensen (1981) | PAWWTP sludge outfall | Cadmium (µg/g) | February 1979 | Ranged from <0.005 to 0.01 µg/g, with a mean concentration of 0.05 µg/g. |
| Steffensen (1981) | PAWWTP sludge outfall | Zinc (µg/g) | February 1979 | Ranged from 1.8 to 6.6 µg/g, with a mean concentration of 3.9 µg/g. |
| Steffensen (1981) | PAWWTP sludge outfall | Lead (µg/g) | February 1979 | Ranged from 1.5 to 6.3 µg/g, with a mean concentration of 2.8 µg/g. |
| Steffensen (1981) | PAWWTP sludge outfall | Copper (µg/g) | February 1979 | Ranged from 0.5 to 1.7 µg/g, with a mean concentration of 3.9 µg/g. |
| EPA (1997b) | Eight sites within the Port River/Barker Inlet system. | Heavy metals, Organochlorine (OC) pesticides, Polychlorinated biphenyls (PCBs), Organotins, Atrazine (a herbicide) | December 1995 to November 1996. Samples collected biannually. | Mean heavy metal concentrations ranged from 7.2 to 185.5, 30.0 to 402.7, 10.3 to 155.1, 0.7 to 1.1, 5.0 to 7.5, and 1.0 to 2.5 mg/kg for copper, zinc, lead, mercury, arsenic, and cadmium, respectively, with highest levels (except for mercury) in the upper Port River adjacent the PAWWTP outfall. The only OC pesticide detected was Chlordane at one site on one occasion. No PCBs were detected. Atrazine was not detected. |
| EPA (2000) | 26 sites in and around the Port River/Barker Inlet system, with 8 of the sites in the estuarine waters. | Heavy metals, PCBs | August 1999 | Heavy metal concentrations ranged from 41 to 3200, 120 to 7000, 30 to 1200, <0.2 to 4.6, <1.0 to 8.0, 16 to 240, and 8 to 64 mg/kg for copper, zinc, lead, mercury, cadmium, chromium, and nickel, respectively at the 8 estuarine sites. The PCB Arochlor 1260 was detected at a concentration of 95 µg/kg at one site. |

**Appendix C.** Summary of studies investigating the status of seagrass meadows along the Adelaide metropolitan coastline. Different sections indicate general technique of data collection. AMC = Adelaide metropolitan coast from Port Gawler to Sellicks Beach; PAWWTP = Port Adelaide Wastewater Treatment Plant; GWWTP = Glenelg Wastewater Treatment Plant; BWWTP = Bolivar Wastewater Treatment Plant; Note that studies conducted in the Port River are not included, and that this list is not exhaustive.

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| | | | **Aerial Remote Sensing** | | |
| Freeman (1982) | Outer Harbor to Kingston Park | Extent of nearshore seagrass line | Aerial photography from 1935, 1949, 1961, 1972, 1975, 1979 and 1981 | Substantial regression of the nearshore seagrass line between 1935 to 1981, particularly from Glenelg to Point Malcolm and just south of Outer Harbour. In 1935, seagrasses grew to within 200-300 m of shore at Glenelg and Henley Beach. Seagrass loss may have commenced as early as 1935. | No species discrimination |
| Steffensen (1985) | Outer Harbor to Kingston Park | Coverage of seagrass | Aerial photography from 1935, 1949, 1961, 1972, 1975 and 1981 | Between 1935 and 1981, substantial (1400 ha total) seawards regression of seagrass line, particularly from Semaphore to North Brighton. No loss detected between North Brighton and Kingston Park. In 1935, seagrasses grew to within 200-300 m of shore at Glenelg and Henley Beach. | No species discrimination. Data appear to be the same source as Freeman (1982). |
| Shepherd et al (1989) | Port Gawler to Barker Inlet | Coverage of seagrass | Aerial photography between 1965 and 1985. | Loss of intertidal *Heterozostera* and subtidal *Posidonia* meadows adjacent BWWTP outfall. Cause of losses postulated as increased epiphyte loads due to increased nutrients. | Posidonia loss included P. australis. |
| Clarke and Kirkman (1989) | Holdfast Bay | Movement of blowouts | Aerial photography in 1949 compared to 1981. | Blowouts move in a southwest direction. Over the 32-year period, movement of the erosion edge averaged 1.0 m/yr but movement of the colonizing edge averaged 0.5 m/yr. Thus, blowouts have expanded. | |
| Hart (1997) | AMC: Largs Bay to Aldinga | Coverage of seagrass and sand | Digital aerial orthophoto-graphy from 1949, 1965, 1971, 1977, 1983, | Between 1949 and 1995/6, estimated loss of 41 $km^2$ between Largs Bay and Aldinga. Most significant losses are nearshore between Largs Bay and Brighton, in | No species discrimination |

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| | | | 1988, 1995 and 1995/6 | northern Holdfast Bay around the PAWWTP sludge outfall, and offshore in the central and southern parts of Holdfast Bay. | |
| Edyvane (1999) | South Australia, including area of ACWS. | Coverage of seagrass and other benthic habitats | Landsat TM satellite imagery and aerial photography, with spot ground-truthing during 1994-1997. | Dense seagrass meadows offshore between Outer Harbour and Brighton. Dense seagrass patches nearshore in Largs Bay and nearshore from West Beach to Port Stanvac. Fragmented meadows in Bolivar/Section Bank region. No seagrasses mapped between Port Stanvac and Sellicks Beach. Dense seagrass meadows offshore from Sellicks Beach in Aldinga Bay. | No species discrimination. Coarse resolution (1: 100 000). Obvious errors in data. Mapping covers AMC out to ca. 20m depth. |
| Cameron (1999) | Gulf St Vincent, including area of ACWS. | Coverage of seagrass and sand in waters <12 m depth | Landsat TM satellite imagery from 1989/90 and 1997/98 | Over the time period examined (1989-1998), estimated loss of 551 ha seagrass, with losses off Henley Beach and Semaphore, but mainly off Bolivar. Extensive seagrass meadows mapped in region from Hallett Cove to Sellicks Beach. | No species discrimination possible. Pixel size (30 m) does not allow small-scale changes to be detected. |
| Kinhill (1999) | Adjacent to BWWTP outfall | Coverage of seagrass in 16km$^2$ area | Aerial photography between 1949 and 1997. | Estimated loss of 320ha of *Posidonia* between 1967 and 1997. Strong circumstantial evidence linking loss to commencement of, and ongoing, discharges from BWWTP. | Area of seagrass loss probably underestimates total area of loss in the region due to the restricted size of study area (16km$^2$) |
| Thomas and Clarke (2002) | Outer Harbor to Marino | Coverage of seagrass | Aerial photography between 1949 and 1985. | Estimated loss of 1,950 ha seagrass. | Lower depth limit assumed to be 15 m depth contour. |
| Cameron (2003) | AMC: North Haven to Sellicks Beach | Coverage of seagrass and sand | Digital aerial orthophoto-graphy | Estimated loss of 721 ha between 1995/6 and 2002. Appears to be substantial loss and fragmentation of deeper meadows in central and southern parts of Holdfast Bay. | No species discrimination. Amount of substrate gain estimated is unlikely to be due to seagrass recolonisation. |
| Blackburn | AMC: Port Gawler | Coverage of | Historical digital | In progress as part of the ACWS. | Attempting species |

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| (In progress) | to Sellicks Beach | seagrass and sand | aerial orthophoto-graphy. Airborne hyperspectral data during 2003. | | discrimination |
| | | | **Underwater Remote Sensing** | | |
| Cheshire *et al.* (2002) | Section Bank/Barker Inlet | Presence, cover, and shoot density | Remote underwater video and diver surveys from intertidal to shallow subtidal (< ca. 5 m depth) during 2001. | Presence of several species: *Posidonia australis*, *Heterozostera tasmanica*, *Zostera muelleri*, *Amphibolis antarctica*, and *Halophila australis*. Very little *Amphibolis antarctica*. | |
| Tanner (2004) | Outer Harbor shipping channel | Cover of dominant habitat types | Transect surveys using mostly remote underwater video & some SCUBA during 2004. | Presence of *Posidonia* and *Amphibolis* adjacent to shipping channel in depths of ~9-12 m, with *Halophila* present in deeper waters west of the channel. | No species discrimination |
| Tanner (2005) | Gulf St Vincent, including area of ACWS. | Distribution, species abundance and composition of benthic habitats | Remote underwater video surveys and spot dives during 2000/2001 | Substantial beds of *Posidonia* (and some *Amphibolis* and *Halophila*) recorded on AMC. *Posidonia* not recorded beyond 15 m depth. Disappearance of deepwater *Heterozostera tasmanica* beds in lower Gulf St Vincent and Investigator Strait potentially linked to increased turbidity from land-based discharges off Adelaide. | Low spatial resolution – sites 2 nm apart in E-W and 5 nm apart in N-S direction |
| | | | ***In Situ* Diver Surveys** | | |
| Shepherd (1970) | Port Gawler to Marino. 68 sites across range of depths. | Aboveground biomass, canopy height | Diver surveys of 100m radius around each site and harvest of 3-6 replicates of 0.167 m2 area to map "isodensities" of *Posidonia* during 1968. | *P. australis* found in depths of 2-18 m with canopy height of ca. 60 cm and aboveground biomass declining from 800 to 100 g/m2 (wet weight) with depth from ca. 5 to 15m in Holdfast Bay. At the nearshore margin aboveground biomass was 100g/m2 and in partially degraded areas adjacent north Glenelg, *Posidonia* had increased epiphyte load and leaf length rarely reached | References to *P. australis* (especially in deeper waters) probably include *P. sinuosa* and *P. angustifolia*, as the existence of these two species was unknown at the time. |

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| | | | | 30cm. *A. antarctica* present along entire near-shore margin from Outer Harbor to Marino with height of 30-40 cm. | |
| Lewis (1975) | Outer Harbor to Marino Rocks: 18 sites along shoreward edge of seagrass meadows; 37 sites inshore to offshore along coast (17 x 500 m offshore, 13 x 1000 m offshore, 7 x ~2000 m offshore) | Aboveground biomass, leaf length, and epiphyte load of seagrass. | Diver surveys using 0.5 x 0.5 m quadrats annually between 1972-1975. | *Posidonia* and *Amphibolis* present along weedline between Outer Harbour and Marino. Complete degradation evident at sites 500 m offshore between Outer Harbour and Glenelg. Relatively low epiphytic cover along coast. Average aboveground biomass of *Posidonia* along 18 sites between 1972-75 ranged from 386-469 g wet weight $m^{-2}$. Peak value of 1598 $g.m^{-2}$. Average aboveground biomass (g wet weight $m^{-2}$) of *Posidonia* between Outer Harbour and Marino at: 500 m offshore 220-264; 1000 m offshore 440-520; >1000 m offshore 567-661. Underwater excavations of old root mats revealed that seagrasses once grew to within 200m of the shore between Outer Harbour and Marino. | *Posidonia* data appear to be predominantly for *P. sinuosa* (referred to as broad-leafed variety) cf. *P. angustifolia* (thin-leafed variety). Data on leaf lengths not presented. |
| Lewis (1975) | Port Gawler to Outer Harbour | General observations | Diver surveys between 1972-1975. | Dense beds of *Posidonia* (broad-leaved form) from St Kilda to Port Prime at 0.5-15 m depths. | The broad-leaved form of *Posidonia* was probably *P. australis* (in shallow water) and *P. sinuosa* (deeper water). |
| Lewis (1975) | Area of 1000 m radius around CBWWTP outfall in 7 m depth. | Species composition over 3-year period. | Diver surveys between 1972-1975. | Sparse clumps of *Posidonia* with large bed of *Amphibolis* to north of O'Sullivan Beach. | No quantitative data. No species discrimination. |
| Lewis (1975) | 21 sites in ca. 8-20 m depth between Glenelg South and Henley Beach, covering ca. 10 km | Species composition (in 1961 just prior to opening of Glenelg sludge outfall, and | Diver surveys in 1961 and 1973-5. Quadrats used in 1975 only. | In 1961, generally dense *Amphibolis* and *Posidonia*, except for deepest (> ca. 15 m) sites but some *Posidonia* still present at 20 m depth. *Amphibolis* was the dominant seagrass species in 1961. In 1975, | Lewis (1975) states that *Posidonia* replaced *Amphibolis* during the period between 1961 and 1975. However, there |

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| | of coastline and up to 4.8 km offshore. | in 1973-5), and aboveground biomass (1973-5 only) | | generally dense *Posidonia* at most sites (~1200-1600 g wet weight $m^{-2}$), but complete lack of *Amphibolis* at all but one site (the inshore site furthest to the north, 1200 g wet weight $m^{-2}$). | appears to have been selective loss of *Amphibolis*. |
| Lewis (1975) | Proposed PAWWTP sludge outfall 4 km offshore from Semaphore. 13 sites near outfall in ~9-11 m depth. | Aboveground biomass of *Posidonia* prior to opening of outfall. | Diver survey during 1974. | Aboveground biomass values of 600-700 g wet weight $m^{-2}$. *Posidonia* found across the 13 sites. No *Amphibolis* detected. | |
| Shepherd and Sprigg (1976) | Gulf St Vincent, including area of ACWS. | Distribution and composition of dominant benthic habitats | Spot dives during 1964-1969, interpolation of data points to create benthic habitat map | Substantial beds of *Posidonia* recorded on AMC. Offshore limit of *Posidonia* meadows indicated at ca. 15 m depth along AMC. *Amphibolis antarctica* rarely occurs in depths > 12 m. | Low spatial resolution of dive sites (~2-3 km intervals between sites along traverses at intervals of 8 km). |
| Johnson (1981) | 5 ha area around proposed PAWWTP sludge outfall in 12 m depth, ca. 4.5 km west of Point Malcolm | Species composition and coverage | Diver transects during 1976-1978, interpolation of data to create seagrass habitat map. | 53% *Posidonia*, 2% *Amphibolis*, 30% mixed *Posidonia* & *Amphibolis*, 15% sand and trench. *Heterozostera* and *Halophila* also present in low densities but not included in survey. | This area was completely denuded due to the sludge outfall, but since the outfall was closed in 1993, there has been some recovery of *Posidonia* and *Halophila* (Bryars and Neverauskas 2004) |
| Steffensen (1981) | PAWWTP: area adjacent to sludge outfall pipeline. | Species composition and coverage | Diver transects and remote underwater TV camera during 1976. | Inshore areas: 40-60% cover *Posidonia sinuosa* with dense patches of *Amphibolis antarctica*, plus understorey of *Halophila* in more open areas. | Qualitative estimates of coverage only |
| Steffensen (1981) | PAWWTP: area adjacent to nearshore 1500 m of sludge outfall pipeline off Point | Species composition | Diver surveys during 1977. | *Amphibolis* was dominant species in first 400 m of nearshore seagrass meadow. | Descriptive only. |

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| | Malcolm | | | | |
| Steffensen (1981) | PAWWTP sludge outfall in 12 m depth, ca. 4.5 km west of Point Malcolm: 55 sites at outfall prior to its opening. | Species composition and coverage | Diver survey during 1978 using a 'needle point sampler' and harvesting of quadrats. | Highly variable coverage of *Posidonia* (up to 90%), *Amphibolis* (up to 40%) and *Halophila*. *Posidonia* aboveground dry weight up to ca. 60 $g.m^{-2}$ with most between 10-50 $g.m^{-2}$. | |
| Neverauskas (1985) | PAWWTP sludge outfall in 12 m depth, ca. 4.5 km west of Point Malcolm | Spatial extent of seagrass loss and degradation | Diver transects during 1982-1984. | Total loss of 365 ha seagrasses. Total area affected was ca. 1900 ha. Selective loss of *Amphibolis* cf. *Posidonia*. Suggested that if epiphyte:leaf biomass ratio reaches ca. 3:1 in *Posidonia* community at 10-12 m depth, then it will die-off. | Did not discriminate *Posidonia* spp. |
| Steffensen (1985) | 6 sites around the GWWTP sludge outfall in ca. 12 m depth. | Leaf density, leaf length, leaf width and epiphyte communities of *Posidonia sinuosa*. | Diver surveys during 1974-1975 but specific technique is unclear. | At shallow sites (~5-7 m), mean leaf density ranged from 515-583.$m^{-2}$ and mean leaf length from 36-43 cm. Density at deeper sites (10 & 12 m) was significantly lower at 100 and 73.$m^{-2}$, respectively, with mean leaf lengths of 25 and 23 cm. No consistent pattern in leaf width with overall range of 3-7 mm. Leaf density and leaf length decline with distance into a bed. Epiphyte composition documented. Noted isolated patches of *Heterozostera*. *P. angustifolia* only recorded as scattered individuals. | Unclear if differences between sites are due to effects of sludge outfall , effluent outfall, or depth/location. |
| Steffensen (1985) | GWWTP sludge outfall site | *Posidonia* degradation /loss | Diver surveys between 1977-1982 but specific technique is unclear. | In 1977, degradation covered an area of 100 m N-S and 40-50 m E-W of outfall. Further surveys in 1978, 1979, and 1982 indicated no further change. Clear correlation between area of loss and GWWTP sludge outfall. Based on initial surveys in Lewis (1975), seagrass loss was *Posidonia* and *Amphibolis*. | |

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| Steffensen (1985) | PAWWTP sludge outfall | - | - | - | Results in this report are documented in Neverauskas (1985, 1987b) |
| Connolly (1986) | Middle Beach to Barker Inlet | Presence of seagrass species, and epiphyte cover. Although surveys were principally designed to detect *Ulva*. | Diver surveys during 1985/1986 at 86 sites. | *Amphibolis* absent from nearshore subtidal area adjacent to BWWTP outfall (where patchy *Posidonia* still found) but present further offshore to 8 m depth. No clear pattern of epiphyte density although epiphytes often seen to be smothering *Posidonia*, and *Amphibolis* usually had heavy epiphyte load. Species present: *A. antarctica*, *P. australis*, *P. sinuosa*, *Halophila australis*, *Heterozostera*, and *Zostera*. | Photographs indicate a loss of *Amphibolis* (old stems evident) amongst surviving *Posidonia*, and the presence of *A. griffithii*. |
| Neverauskas (1987a) | Offshore from Brighton in 0-18 m depth | Shoot density, leaf density, leaf width, leaf length, number of leaves per shoot, aboveground biomass, and epiphyte load of *Posidonia*. Photosynthetically active radiation | Diver transect sometime between 1983-1985 with replicate 250 x 250 mm quadrats harvested at 5, 7, 10, 13.5 and 16.5 m depth intervals | Mean aboveground biomasses of ca. 180-250 g dry weight $m^{-2}$ at 5-13.5 m depths with max. at 10 m and significantly lowest at 16.5 m depth with ca. 60 g.$m^{-2}$. Mean shoot densities of ca. 1000-1300 $m^{-2}$ at 5-13.5 m depths, but significantly lowest at 16.5 m depth at ca. 600 $m^{-2}$. Mean leaf densities of ca. 2000-2500 $m^{-2}$ at 5-13.5 m depths, but significantly lowest at 16.5 m depth at ca. 900 $m^{-2}$. Mean total leaf length per shoot tended to decline with depth from ca. 1000 at 5 m to ca. 500 mm at 16.5 m. Leaf width did not change with depth, with mean values of ca. 4.5 to 5.5 mm. Meadows of *Posidonia* were absent at 18 m depth. | Did not discriminate *Posidonia* spp. |
| Neverauskas (1987a, 1988) | Single site ca. 10 km offshore from Port Gawler in 11-12 m depth. | Effects of reduced light on *Posidonia* shoot density, leaf density, leaf width, leaf length, number of leaves per shoot, | During the early 1980's, 250 x 250 mm quadrats harvested at 3 monthly intervals over 1 year period. | Reported presence of *Posidonia sinuosa* and *Posidonia angustifolia*. Initial mean values of: aboveground biomass 100 g dry weight $m^{-2}$, leaf density 1650 $m^{-2}$, shoot density 1100 $m^{-2}$, leaf width 5.6 mm, number leaves per shoot 1.5, leaf | Did not discriminate *Posidonia* species in analyses. |

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| | | aboveground biomass, epiphyte load | | length 320 mm. | |
| Neverauskas (1987b) | 14 sites around and away from PAWWTP sludge outfall (approx. depth range 10-15 m). | Aboveground biomass, number of leaves, and epiphyte load of *Posidonia*. | Harvesting of 250 x 250 mm quadrats for each of 3 summers between 1983-1985. | High epiphyte loads associated with sludge outfall: mean epiphyte/leaf ratios of ca. 0.5-1.9. Aboveground biomass and leaf number were both negatively correlated with epiphyte load. At control sites, ranges of mean values were: aboveground biomass ca. 70-190 g.m$^{-2}$; leaf density ca. 1000-2300 m$^{-2}$; epiphyte/leaf ratio ca. 0.3-0.4. | No discrimination of *Posidonia* species. |
| Sergeev *et al.* (1988) | 2 sites 1.4 km off Henley Beach in 6-7 m depth; 1 site 2.6 km off Brighton in 10-11 m depth. | Qualitative descriptions of sites | Diver survey during 1985. | Henley Beach sites: mixed meadows of *P. angustifolia* and *P. sinuosa,* with smaller patches of *Amphibolis antarctica*. Brighton site: dominated by *P. angustifolia* with small patches of *Amphibolis antarctica*. | Study was designed to examine motile epifaunal communities and not seagrasses. |
| Clarke and Kirkman (1989) | Holdfast Bay | General observations of blowouts | Diver surveys during early 1980's. | Few localities where *Heterozostera* and *Amphibolis* found. Dominance of *P. sinuosa* and P. *angustifolia*. Evidence that *P. sinuosa* is being replaced by *P. angustifolia* in central and southern parts of Holdfast Bay. | |
| Shepherd et al (1989) | Adjacent BWWTP outfall | Leaf density, leaf length, leaf width, and epiphyte abundance. | Diver surveys between 1973-1976. Specific techniques unspecified. | Over the 3-year period, leaf density, leaf length, and leaf width of *Posidonia australis* declined and epiphyte abundance increased. | No values given for parameters of seagrass quality |
| Cheshire and Miller (1999) | Port Stanvac to Christies Beach | Shoot density | Diver surveys from 1992 to 1999 in 12-18 m depth. | Significant *Posidonia* in southern sites at 15 & 18 m depth. | Shoot density data not presented |
| Cheshire and Miller (1996) | Outer Harbor to Brighton | Species composition, shoot density of *Posidonia* | Diver surveys during early 1990's along 7 transect lines perpendicular to the shoreline in 3-7 m depth and 1 km | Presence of *Posidonia*, *Amphibolis* and *Heterozostera* with significant amounts of *Posidonia* (> 100 shoots m$^{-2}$) on several lines. | No species discrimination |

| Author, Year | Location | Parameters | Technique | Main outcomes relevant to the ACWS | Comments |
|---|---|---|---|---|---|
| | | | offshore. | | |
| Harbison and Wiltshire (1997) | Largs Bay to Brighton: 5 nearshore sites (all <5 m depth) plus 1 at Port Hughes | Cover, leaf length, leaf growth, and productivity of *Posidonia* | Diver surveys during April and October 1996 using 20 x 20 cm quadrats, measuring length of longest 25% of leaves. | Cover (%) of *P. sinuosa* was Brighton (50), Grange (5), Largs Bay (5) and Glenelg (1). *P. australis* was only species at St Kilda site. Mean leaf length of *P. sinuosa* for AMC sites ranged from 173-206 mm. | |
| Thomas and Clarke (2002) | Outer Harbor to Brighton | Species ID, % cover, shoot density, and standing crop. | Diver surveys during 1985 along 4 transects perpendicular to the shoreline from 0 m out to the depth limit of *Posidonia* in the area (~18 m) | Presence of *P. australis* in inshore margin of seagrass meadows in Largs Bay. Vegetated areas dominated by mixed *P. sinuosa* and *P. angustifolia* with *P. sinuosa* dominant in north of Holdfast Bay and *P. angustifolia* dominant in south. *A. antarctica* and *H. australis* also noted. Percent cover declines with depth from 95-100% in shallow to 0% at 17-18 m depth. Dense area of *Amphibolis* noted at ~9 m depth off Largs Bay. No blow-outs observed on transect off Largs Bay. Blow-outs observed on all 3 transects between Point Malcolm and Brighton at depths of 3-15m. From 1935 aerial photography: seagrasses grew within ca. 150m of shore at West Beach. | |
| SA DEH (unpubl. data) | Outer Harbour to Sellicks Beach. | A series of rod-lines out to ca. 7-9 m depth along the AMC. | Diver surveys measuring sediment levels | Data collected include qualitative estimates of seagrass species composition and density. However, seagrass data are not currently in a collated form that enables analysis. | |

**Appendix D.** Test results for pesticides and herbicides in water samples collected from the Torrens River, the Torrens River estuary, and offshore from the Torrens River after major rainfall events in May and June 2004. All values are µg $L^{-1}$. TR, Torrens River; TRE, Torrens River estuary; TRO, Torrens River offshore; <, value was less than detectable limit of testing procedure. See text for further details.

| Compound | TR 18/5/04 | TRE 19/5/04 | TRE 21/5/04 | TS 21/5/04 | TRE 9/6/04 |
|---|---|---|---|---|---|
| **Organochlorine Pesticides** | | | | | |
| HCB | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Lindane | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Heptachlor | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Aldrin | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| BHC Total(other than lindane) | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Heptachlor epoxide | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Chlordane | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| DDE | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Dieldrin | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Endrin | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| DDD | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| DDT | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Methoxychlor | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Endosulfan Total | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| **Herbicides** | | | | | |
| Glyphosate in water | <10 | <10 | <10 | <10 | <10 |
| **Triazine Herbicides** | | | | | |
| Atrazine | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Hexazinone | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Metribuzine | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Prometryne | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Simazine | <0.1 | <0.1 | **0.23** | <0.1 | **0.95** |
| **Organophosphate Pesticides** | | | | | |
| Demeton-S-Methyl | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Diazinon | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Dimethoate | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Pirimiphos-Methyl | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Chlorpyrifos | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Parathion | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Malathion | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Fenthion | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Ethion | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |
| Azinphos-Methyl | <0.1 | <0.1 | <0.1 | <0.1 | <0.1 |

**Appendix E.** Test results for pesticides and herbicides in water samples collected in the Torrens River estuary after a major rainfall event in June 2005. All values are µg $L^{-1}$. TRE, Torrens River estuary; <, value was less than detectable limit of testing procedure. See text for further details.

| Compound | TRE 10/6/05 | TRE 10/6/05 | TRE 14/6/05 | TRE 14/6/05 |
|---|---|---|---|---|
| **Organochlorine Pesticides** | | | | |
| HCB | <0.010 | <0.010 | <0.010 | <0.010 |
| Heptachlor | <0.010 | <0.010 | <0.010 | <0.010 |
| Heptachlor epoxide | <0.010 | <0.010 | <0.010 | <0.010 |
| Aldrin | <0.010 | <0.010 | <0.010 | <0.010 |
| gamma-BHC (Lindane) | <0.010 | <0.010 | <0.010 | <0.010 |
| alpha-BHC | <0.010 | <0.010 | <0.010 | <0.010 |
| beta-BHC | <0.010 | <0.010 | <0.010 | <0.010 |
| delta-BHC | <0.010 | <0.010 | <0.010 | <0.010 |
| trans-Chlordane | <0.010 | <0.010 | <0.010 | <0.010 |
| cis-Chlordane | <0.010 | <0.010 | <0.010 | <0.010 |
| Oxychlordane | <0.010 | <0.010 | <0.010 | <0.010 |
| Dieldrin | <0.010 | <0.010 | <0.010 | <0.010 |
| p,p-DDE | <0.010 | <0.010 | <0.010 | <0.010 |
| p,p-DDD | <0.010 | <0.010 | <0.010 | <0.010 |
| p,p-DDT | <0.010 | <0.010 | <0.010 | <0.010 |
| Endrin | <0.010 | <0.010 | <0.010 | <0.010 |
| Endrin Aldehyde | <0.010 | <0.010 | <0.010 | <0.010 |
| Endrin Ketone | <0.010 | <0.010 | <0.010 | <0.010 |
| alpha-Endosulfan | <0.010 | <0.010 | <0.010 | <0.010 |
| beta-Endosulfan | <0.010 | <0.010 | <0.010 | <0.010 |
| Endosulfan Sulfate | <0.010 | <0.010 | <0.010 | <0.010 |
| Methoxychlor | <0.010 | <0.010 | <0.010 | <0.010 |
| **Triazine Herbicides** | | | | |
| Atrazine | <0.10 | <0.10 | <0.10 | <0.10 |
| Hexazinone | <0.10 | <0.10 | <0.10 | <0.10 |
| Metribuzine | <0.10 | <0.10 | <0.10 | <0.10 |
| Prometryne | <0.10 | <0.10 | <0.10 | <0.10 |
| Simazine | <0.10 | <0.10 | <0.10 | <0.10 |
| **Organophosphate Pesticides** | | | | |
| Demeton-S-Methyl | <0.10 | <0.10 | <0.10 | <0.10 |
| Dichlorvos | <0.10 | <0.10 | <0.10 | <0.10 |
| Diazinon | <0.10 | <0.10 | <0.10 | <0.10 |
| Dimethoate | <0.10 | <0.10 | <0.10 | <0.10 |
| Chlorpyrifos | <0.10 | <0.10 | <0.10 | <0.10 |
| Chlorpyrifos Methyl | <0.10 | <0.10 | <0.10 | <0.10 |
| Malathion | <0.10 | <0.10 | <0.10 | <0.10 |
| Fenthion | <0.10 | <0.10 | <0.10 | <0.10 |
| Azinphos Ethyl | <0.10 | <0.10 | <0.10 | <0.10 |
| Azinphos Methyl | <0.10 | <0.10 | <0.10 | <0.10 |
| Chlorfenvinphos (E) | <0.10 | <0.10 | <0.10 | <0.10 |
| Chlorfenvinphos (Z) | <0.10 | <0.10 | <0.10 | <0.10 |
| Ethion | <0.10 | <0.10 | <0.10 | <0.10 |
| Fenitrothion | <0.10 | <0.10 | <0.10 | <0.10 |
| Parathion (Ethyl) | <0.10 | <0.10 | <0.10 | <0.10 |
| Parathion Methyl | <0.10 | <0.10 | <0.10 | <0.10 |
| Pirimiphos Ethyl | <0.10 | <0.10 | <0.10 | <0.10 |
| Pirimiphos Methyl | <0.10 | <0.10 | <0.10 | <0.10 |

**Appendix F.** Test results for hydrocarbons, pesticides, herbicides, and trace elements in marine sediment samples collected adjacent the Torrens River and the Barcoo Outlet. All values are mg/kg dry weight. <, value was less than detectable limit of testing procedure. See text for further details.

| Compound | Torrens 1 | Torrens 2 | Barcoo 1 | Barcoo 2 |
|---|---|---|---|---|
| **Organochlorine Pesticides** | | | | |
| HCB | <0.01 | <0.01 | <0.01 | <0.01 |
| gamma BHC ( Lindane ) | <0.01 | <0.01 | <0.01 | <0.01 |
| Heptachlor | <0.01 | <0.01 | <0.01 | <0.01 |
| Aldrin | <0.01 | <0.01 | <0.01 | <0.01 |
| BHC(other than g-BHC) | <0.01 | <0.01 | <0.01 | <0.01 |
| Heptachlor epoxide | <0.01 | <0.01 | <0.01 | <0.01 |
| Chlordane (trans and cis) | <0.01 | <0.01 | <0.01 | <0.01 |
| DDE | <0.01 | <0.01 | <0.01 | <0.01 |
| Dieldrin | <0.01 | <0.01 | <0.01 | <0.01 |
| Endrin | <0.01 | <0.01 | <0.01 | <0.01 |
| DDD | <0.01 | <0.01 | <0.01 | <0.01 |
| DDT | <0.01 | <0.01 | <0.01 | <0.01 |
| Methoxychlor | <0.01 | <0.01 | <0.01 | <0.01 |
| Total Endosulfan | <0.01 | <0.01 | <0.01 | <0.01 |
| **Herbicides** | | | | |
| Glyphosate in water | <0.05 | <0.05 | <0.05 | <0.05 |
| **Triazine Herbicides** | | | | |
| Atrazine | <0.1 | <0.1 | <0.1 | <0.1 |
| Hexazinone | <0.1 | <0.1 | <0.1 | <0.1 |
| Metribuzine | <0.1 | <0.1 | <0.1 | <0.1 |
| Prometryne | <0.1 | <0.1 | <0.1 | <0.1 |
| Simazine | <0.1 | <0.1 | <0.1 | <0.1 |
| **Organophosphate Pesticides** | | | | |
| Demeton-S-Methyl | <0.1 | <0.1 | <0.1 | <0.1 |
| Diazinon | <0.1 | <0.1 | <0.1 | <0.1 |
| Dimethoate | <0.1 | <0.1 | <0.1 | <0.1 |
| Pirimiphos-Methyl | <0.1 | <0.1 | <0.1 | <0.1 |
| Chlorpyriphos | <0.1 | <0.1 | <0.1 | <0.1 |
| Parathion | <0.1 | <0.1 | <0.1 | <0.1 |
| Malathion ( Maldison ) | <0.1 | <0.1 | <0.1 | <0.1 |
| Fenthion | <0.1 | <0.1 | <0.1 | <0.1 |
| Ethion | <0.1 | <0.1 | <0.1 | <0.1 |
| Azinphos-Methyl | <0.1 | <0.1 | <0.1 | <0.1 |
| **Total Petroleum Hydrocarbons** | | | | |
| TPH C6 - C9 | <25 | <25 | <25 | <25 |
| TPH C10 - C14 | <50 | <50 | <50 | <50 |
| TPH C15 - C28 | <100 | <100 | <100 | <100 |
| TPH C29 - C36 | <100 | <100 | <100 | <100 |
| **Poly Aromatic Hydrocarbons** | | | | |
| Naphthalene | <1.0 | <1.0 | <1.0 | <1.0 |
| Acenaphthylene | <1.0 | <1.0 | <1.0 | <1.0 |
| Acenaphthene | <1.0 | <1.0 | <1.0 | <1.0 |
| Fluorene | <1.0 | <1.0 | <1.0 | <1.0 |
| Phenanthrene | <1.0 | <1.0 | <1.0 | <1.0 |
| Anthracene | <1.0 | <1.0 | <1.0 | <1.0 |
| Fluoranthene | <1.0 | <1.0 | <1.0 | <1.0 |
| Pyrene | <1.0 | <1.0 | <1.0 | <1.0 |
| Benz(a)anthracene | <1.0 | <1.0 | <1.0 | <1.0 |

| Compound | Torrens 1 | Torrens 2 | Barcoo 1 | Barcoo 2 |
|---|---|---|---|---|
| Chrysene | <1.0 | <1.0 | <1.0 | <1.0 |
| Benzo(b)&(k)fluoranthene | <1.0 | <1.0 | <1.0 | <1.0 |
| Benzo(a)pyrene | <1.0 | <1.0 | <1.0 | <1.0 |
| Indeno(1,2,3-cd)pyrene | <1.0 | <1.0 | <1.0 | <1.0 |
| Dibenz(ah)anthracene | <1.0 | <1.0 | <1.0 | <1.0 |
| Benzo(ghi)perylene | <1.0 | <1.0 | <1.0 | <1.0 |
| **Trace Elements** | | | | |
| Cadmium | <0.5 | <0.5 | <0.5 | <0.5 |
| Chromium | **2.1** | **2.1** | **2.9** | **3** |
| Copper | <0.5 | <0.5 | <0.5 | <0.5 |
| Lead | **1.3** | **1.2** | **1.7** | **1.5** |
| Mercury | <0.2 | <0.2 | <0.2 | <0.2 |
| Zinc | **1.7** | **2.1** | **3.7** | **2.8** |

**Appendix G.** Test results for hydrocarbons, pesticides, herbicides, and trace elements in marine sediment samples collected at eight of the nine ambient sampling sites. All values are mg/kg dry weight. <, value was less than detectable limit of testing procedure. See text for further details.

| Compound | 1S | 1D | 2S | 2D | 3S | 3D | 4S | 4D |
|---|---|---|---|---|---|---|---|---|
| **Organochlorine Pesticides** | | | | | | | | |
| HCB | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Heptachlor | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Heptachlor epoxide | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Aldrin | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| gamma-BHC (Lindane) | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| alpha-BHC | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| beta-BHC | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| delta-BHC | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| trans-Chlordane | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| cis-Chlordane | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Oxychlordane | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Dieldrin | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| pp-DDE | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| pp-DDD | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| pp-DDT | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Endrin | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Endrin Aldehyde | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Endrin Ketone | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| alpha-Endosulfan | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| beta-Endosulfan | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Endosulfan Sulfate | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| Methoxychlor | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 | <0.010 |
| **Triazine Herbicides** | | | | | | | | |
| Atrazine | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Hexazinone | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Metribuzine | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Prometryne | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Simazine | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| **Organophosphate Pesticides** | | | | | | | | |
| Dichlorvos | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Demeton-S-Methyl | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Diazinon | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Dimethoate | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Chlorpyrifos | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Chlorpyrifos Methyl | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Malathion (Maldison) | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Fenthion | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Ethion | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Fenitrothion | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Chlorfenvinphos (E) | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Chlorfenvinphos (Z) | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Parathion (Ethyl) | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Parathion Methyl | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Pirimiphos Methyl | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Pirimiphos Ethyl | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Azinphos-Methyl | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| Azinphos Ethyl | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 | <0.10 |
| **Total Petroleum Hydrocarbons** | | | | | | | | |
| TPH C6 - C9 | <25 | <25 | <25 | <25 | <25 | <25 | <25 | <25 |
| TPH C10 - C14 | <50 | <50 | <50 | <50 | <50 | <50 | <50 | <50 |
| TPH C15 - C28 | <100 | <100 | <100 | <100 | <100 | <100 | <100 | <100 |
| TPH C29 - C36 | <100 | <100 | <100 | <100 | <100 | <100 | <100 | <100 |

| Compound | 1S | 1D | 2S | 2D | 3S | 3D | 4S | 4D |
|---|---|---|---|---|---|---|---|---|
| **Poly Aromatic Hydrocarbons** | | | | | | | | |
| Naphthalene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Acenaphthylene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Acenaphthene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Fluorene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Phenanthrene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Anthracene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Fluoranthene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Pyrene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Benz(a)anthracene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Chrysene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Benzo(b)&(k)fluoranthene | <2.0 | <2.0 | <2.0 | <2.0 | <2.0 | <2.0 | <2.0 | <2.0 |
| Benzo(a)pyrene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Indeno(1,2,3-cd)pyrene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Dibenz(ah)anthracene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| Benzo(ghi)perylene | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 | <1.0 |
| **Trace Elements** | | | | | | | | |
| Cadmium | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 |
| Chromium | **3.2** | **2.8** | **1.1** | **1.1** | **0.6** | **0.53** | **4.2** | **11** |
| Copper | **0.85** | **0.73** | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 |
| Lead | **2.6** | **2.1** | **0.96** | **1** | <0.5 | <0.5 | **2** | **4.3** |
| Mercury | <0.2 | <0.2 | <0.2 | <0.2 | <0.2 | <0.2 | <0.2 | <0.2 |
| Zinc | **4.2** | **2.2** | **1.6** | **1.2** | **0.5** | **0.52** | **5.7** | **23** |