Environment Protection Authority

Nearshore marine aquatic ecosystem condition reports

Northern Spencer Gulf bioregional assessment report 2012

South Australia

Nearshore marine aquatic ecosystem condition reports: Northern Spencer Gulf bioregional assessment report

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Summary

Spencer Gulf contains more than half of South Australia's seagrass communities, with the northern section supporting extensive meadows. Seagrass provide vital functions:

- improve water quality and increase the clarity of water
- nursery habitat for many fish and invertebrates
- stabilise sediment helping to prevent erosion of beaches
- process wastes.

Rocky reef communities make up a small proportion of the Northern Spencer Gulf but they are significant. A small stretch of rocky coastline supports the world's largest breeding aggregation of giant Australian cuttlefish, which is known the world over for their colourful courting displays.

The nearshore habitats in the Northern Spencer Gulf are under pressure from numerous heavy industry, coastal discharges and development.

This assessment report details the results of the Environment Protection Authority's nearshore marine monitoring, evaluation and reporting (MER) program for the Northern Spencer Gulf bioregion (NSG) undertaken in 2012. The program assesses the seagrass, reef and unvegetated sediment habitats in nearshore waters across the NSG.

Winninowie

The Winninowie biounit is located in the uppermost part of the Gulf, north of Point Lowly and Ward Point. It is affected by high salinities and temperatures, making the biounit a unique environment. In 2012, 10 sites were sampled in Winninowie. Seagrass comprised 80% of the sites, bare sand 20%, and no reef was observed. Winninowie has dense and intact seagrass meadows generally in good condition, but there were areas where seagrass had disappeared. Throughout the biounit there were signs of nutrient enrichment, such as epiphyte on seagrass leaves and opportunistic algae (eg *Hincksia* spp).

Pressures affecting Winninowie include nutrient pollution discharged from the Port Augusta East wastewater treatment plant, and large volumes of thermal effluent from the power stations at Port Augusta. The region has a network of stormwater discharges from the urban centres and extensive shack communities that rely on septic tanks to treat sewage. Impacts to these habitats may be exacerbated by poor water exchange due to Winninowie's position at the top of the gulf and its unique temperature and salinity conditions. Overall Winninowie was in FAIR condition.

Yonga

Below Winninowie is the Yonga biounit extending south to Franklin Harbour and Point Riley. The EPA assessed 50 sites in autumn and spring 2012, and found seagrass was the dominant habitat across 86% of the sites, while 12% were sand, and only 2% reef. Generally seagrass habitats were flourishing, forming extensive, dense and continuous meadows, but closer to towns, epiphytes growing on seagrass leaves and nuisance opportunistic algae suggest nutrient enrichment.

Numerous discharges in Yonga may be effecting habitats, including ammonia rich wastewater from the Whyalla steelworks, metal laden discharges from the Port Pirie smelter, and nutrients from the Whyalla and Port Pirie wastewater treatment plants. Additionally, stormwater runoff from urbanised areas, and pollution from further south in Spencer Gulf flowing north into Yonga. The region has a legacy of historical metal pollution that has affected nearshore habitats, particularly around Port Pirie and Germien Bay. Yonga was considered to be in GOOD condition.

The evaluation of the impacts from metals to marine habitats is undertaken by the EPA through more specific and targeted programs, and not as a part of the EPA's aquatic ecosystem condition report (AECR) program[1].

[1] www.epa.sa.gov.au/data_and_publications/water_quality_monitoring/aquatic_ecosystem_monitoring_evaluation_and_reporting

This document is intended for the scientific and the interested public. It uses scientific language and complex statistical concepts to explain detailed findings and justify conclusions. Simplified report card style summaries can be found on the EPA website including the raw data and links to underwater video footage. The intention is to return to the NSG in five years to reassess the biounits to better inform whether habitat condition is changing.

1 Introduction

The Integrated Marine and Coastal Regionalisation of Australia (IMCRA) [Commonwealth of Australia 2006] uses a spatial framework delineating waters throughout Australia into bioregions, using biological data and inferred ecosystem patterns. The Spencer Gulf is separated into two bioregions (mesoscale) – lower (southern) Spencer Gulf and Northern Spencer Gulf (NSG) [Commonwealth of Australia 2006]. The assessment of the lower Spencer Gulf bioregion was conducted in 2010 and findings published in 2013 (Gaylard *et al* 2013a).

Spencer Gulf is a semi-enclosed sea approximately 325 km long and ~60 km wide, separating the Eyre and Yorke Peninsulas in South Australia. The mean depth is 22 m, however the NSG is considerably shallower with a mean depth of ~13 m, decreasing to ~7 m north of Point Lowly (Nunes Vaz & Lennon 1986). The shallow depth and limited water exchange leads to water temperatures ranging from 12–24°C, gradually increasing towards the head of the gulf (Nunes Vaz & Lennon 1986).

The region also experiences high salinity due to annual evaporation that exceeds freshwater inflow (land-based runoff and precipitation), creating an inverse estuary where the salinity gradually increases towards the head of the gulf (Nunes Vaz *et al* 1990). Salinity in NSG is regulated through natural flow of dense, hypersaline water along the seafloor on the eastern side of the Gulf in deeper waters during winter. The density driven current transports salt from the Gulf preventing salinity from increasing continually (Nunes Vaz & Lennon 1986). The temperature and salinity gradients create environmental niches or transition zones (ecotones) where ecological gradients reflect plants and/or organisms that have adapted to the local temperature and salinity conditions (BHPB 2009) and support some species with tropical and subtropical affinities (Shepherd 1983).

Of the 5,512 km^2 of seagrass in Spencer Gulf, over 75% (4,138 km^2) is in NSG with an estimated 25% of the total seagrass area in South Australia in the Yonga biounit alone (Edyvane 1999a). Seagrass meadows are particularly important, providing essential ecosystem services such as improving water quality and light availability, nursery habitat for fish and invertebrates, sediment stabilisation and nutrient cycling (Larkum *et al* 2006). Additionally, evidence suggests that the carbon storage of seagrasses are similar to that of forests, which is increasingly being recognised as important for climate change mitigation (Fourqurean *et al* 2012).

Although seagrass meadows are the dominant marine habitat in NSG, small areas of macroalgal dominated rocky reefs, and mangrove forests are also present (Edyvane 1999b). Together, the marine habitats support diverse assemblages of plants and animals that sustain some ecologically and economically important species. The mangrove and seagrass areas provide important nursery areas for (among others) Southern Sea Garfish (*Hyporhamphus melanochir*), King George Whiting (*Sillagnoides punctata*), Snapper (*Chrysophrys auratus*), Western King Prawn (*Melicertus latisulcatus*) and Blue Swimmer Crab (*Portunus pelagicus*).

Although reef areas constitute a relatively small proportion of the marine habitat of NSG, they are extremely valuable to the inhabitants. For instance, the giant Australian cuttlefish (*Sepia apama*), the largest cuttlefish in the world, aggregates annually to breed and spawn on a limited area of subtidal reef near Point Lowly (Gillanders and Payne 2014). The aggregated population has been the focus of a burgeoning ecotourism industry where tens of thousands of individuals could be easily observed in waters from 2–8 m (Gillanders and Payne 2014). In the mid 2000s a substantial population decline was observed with 183,000 animals in 1999 declining over several years to an estimated 13,500 in 2013 (Gillanders and Payne 2014). The cause of this decline is being investigated to ensure the populations longevity (Steer *et al* 2013).

NSG includes two biounits; at the southern extent, the Yonga biounit spans the width of the gulf from Victoria Point, south of Lucky Bay on the western side; to Point Riley, north of Wallaroo on the eastern side. Winninowie extends from the northern boundary of Yonga to Port Augusta at the top of the gulf (Edyvane 1999b, Figure 1).

Maintaining good water quality is vital for the wellbeing of marine ecosystems, with associated benefits for human health and the sustainability of industries (Bierman *et al* 2011). Understanding of the condition of habitats within nearshore waters is fundamental to ensuring that the actions of humans are not further degrading these habitats, resulting in losses of productivity and ecosystem services, and ultimately significant financial, social, cultural, and amenity losses (Costanza *et al* 1997, McArthur and Boland 2006, McDowell and Pfennig 2013).

The EPA Nearshore Monitoring, Evaluation and Reporting (MER) program aims to investigate broad ecological condition based upon the status of the dominant subtidal habitats in the nearshore marine waters across South Australia, typically seagrass, rocky reefs and unvegetated soft sediment (Edyvane 1999b). In this MER program, it is inferred that a healthy habitat will result in a healthy and biodiverse ecosystem.

1.1 Nearshore marine monitoring framework

The EPA report, The South Australian monitoring, reporting and evaluation program for aquatic ecosystems: Rationale and methods for the assessment of nearshore marine waters (Gaylard et al 2013a), details the framework and methods undertaken to assess broad ecological condition in South Australian nearshore marine environments. An overview of methods is provided in Appendix 1, but will not explain the methods in detail. The NSG report should be read in close association with Gaylard et al (2013a).

In summary, the nearshore marine MER program has been designed using a three-tier framework, which includes;

- **Tier 1** – a literature review and desktop threat assessment to study pressures on nearshore ecological communities in each biounit. This information is used to review and update conceptual models and if required, tailor the monitoring to address identifiable threats specific to each bioregion. A *predicted* condition for each biounit following the conceptual disturbance gradient (Appendix 2), is also developed combining data from the threat assessment and available published literature.
- **Tier 2** – a rapid field assessment program is undertaken to quantify the condition of the habitat. Condition monitoring for the nearshore MER program is undertaken throughout each biounit across two periods; autumn and spring, and the results are used to develop the *observed* condition.

The information collected from Tiers 1 and 2 is used to prepare an aquatic ecosystem condition report (AECR) for each biounit. The AECRs are designed to convey complex scientific information to the general public in an easily accessible format. This rating serves as a broad assessment of the *observed* ecological condition at the time of sampling, identifies the main pressures that are likely to be driving the observed condition, and lists the main management responses that are designed to address the pressures identified.

- **Tier 3** – where there are noteworthy differences between the *predicted* (Tier 1) and *observed* (Tier 2) conditions for a biounit, suggesting gaps in our understanding of threats and/or biological responses, the biounit may be highlighted as requiring further research. In such circumstances, the publication of the AECR will proceed with caution and a statement of limitations can be made, highlighting the need for further work in this location. Additional work may be undertaken by the EPA or another institution to investigate the drivers of the observed condition.

1.2 The structure of the current report

This document, in conjunction with Gaylard *et al* (2013b), details the science behind the assessment of habitat condition in Yonga and Winninowie biounits in the NSG. The report details the Tier 1 assessment including an overview of the threat assessment process leading to predicted condition of each biounit (Section 2). Section 3 provides a brief review of the methods of the field program (Tier 2) and how ecological condition has been evaluated. Sections 4 and 5 present the results of the Tier 2 program at the bioregional (NSG) and biounit scale (Yonga and Winninowie). These sections provide interpretation to some of the ecological processes within various strata that have been measured and whether there are observable biological gradients present that may aid in a determination of the condition, keeping in mind the broad-scale nature of the design of the MER program.

This information has been distilled to develop an AECR for each biounit aimed to communicate complex scientific information to the general public. In addition to the AECR, the raw data and a two-minute representative snapshot of the underwater video have been provided on the EPA website.

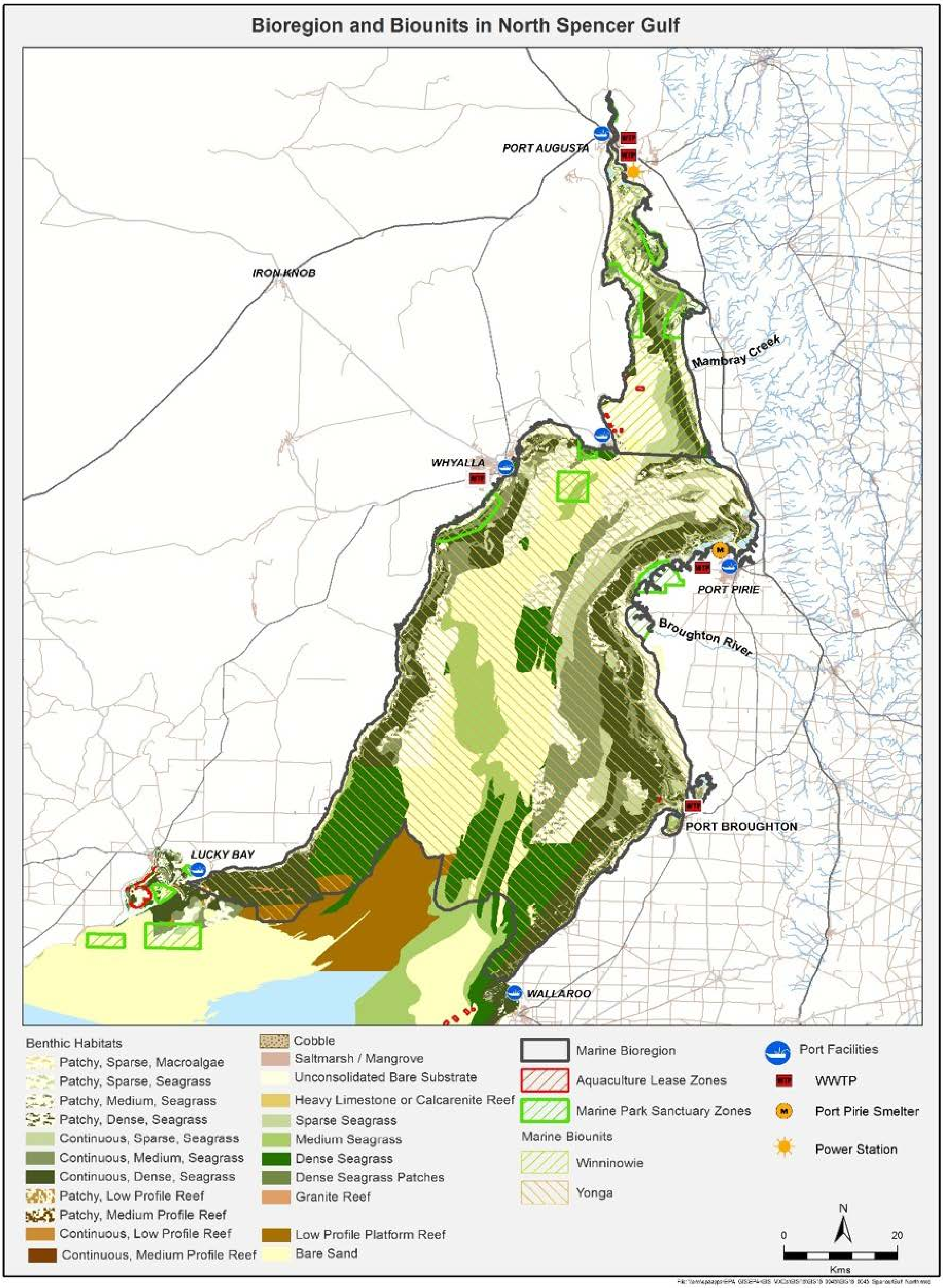


Figure 1 The Northern Spencer Gulf bioregion showing the Yonga and Winninowie biounits

2 Tier 1 – Assessment of threats to the Northern Spencer Gulf

A range of human activities can negatively impact on coastal marine systems and threaten ecological processes. Each human activity can be evaluated to have a probability or likelihood of causing a negative impact, which can be broadly estimated through knowledge of the process and pathways specific to that activity. The second consideration is the scale of negative impact that an activity might have (consequence). Using this framework, a threat assessment for each biounit has been developed. The threat assessment process for this purpose is detailed in Gaylard *et al* (2013b). A full quantitative risk assessment is outside the scope of this program.

2.1 Tier 1 – Winninowie

Description

Winninowie (Figure 2) is relatively shallow (mean depth 7 m) and more than 260 km from the mouth of the gulf, resulting in low wave energies, increased temperature/salinity and reduced water exchanges relative to waters further south. The complex geomorphology of the coastline includes numerous small bays and tributaries linking extensive mangrove flats to narrow, deeper channels where there are strong currents and large tides in excess of 3 m (BOM 2013).

The combination of widespread mangrove forests becoming inundated at high tides, broad coverage of seagrass and unique oceanographic profile has formed important marine habitats in Winninowie (Edyvane 1999b, Figure 2). The State Benthic Habitat Mapping of Winninowie suggests that 66% of the benthos was seagrass, 33% sand and less than 1% was reef (DEWNR 2010). Seagrass meadows are dominated by *Posidonia australis* which sprawl from the intertidal to sandy subtidal platforms and banks to a depth of about 4 m before giving way to a predominance of *P sinuosa* with subordinate growth of *Heterozostera tasmanica*, *Amphibolis antarctica* and *Halophila ovalis* (Shepherd 1983). In some offshore waters there may be sparse *Posidonia* spp. as well as *Halophila australis* where sufficient light allows (Edyvane 1999b).

Threat assessment

The City of Port Augusta is the largest urban centre in the biounit with 13,808 people (ABS 2016). Up to the1860s, Port Augusta was the shipping gateway to the state's north, facilitating general import/export for goods for the Mount Remarkable region which was used for extensive sheep grazing and sporadic mining (Anderson 1988). Use of the port began to decrease with the introduction of diesel trains and the Transcontinental Railway line in the 1950s and the port was eventually closed in 1974 (Anderson 1988). More recently, Port Augusta has become an important road transport hub and tourist attraction.

Winninowie experiences low annual rainfall between 300–400 mm (BOM 2014) resulting in small and localised runoff. The changes in land use and catchment quality to accommodate extensive agriculture is likely to result in poor quality runoff from agricultural lands. While the annual volume discharged to the sea is likely to be low, the quality is also poor in the initial flush.

Port Augusta's sewage is treated through two wastewater treatment plants (WWTP) at Port Augusta West and Port Augusta East (Figure 2). Port Augusta East discharges nutrient rich effluent into the nearshore coastal waters near Hospital Creek just south of Port Augusta, while the vast majority of the sewage treated at the Port Augusta West plant is recycled and used to irrigate ovals and greenspaces.

Away from the city, shacks dependent on septic tanks to treat sewage, line much of the western side of Winninowie from Commissariat Point to Blanche Harbour and also Miranda on the eastern side (Figure 2). Septic tanks have been shown to introduce nutrients into the shallow groundwater at a load of between 5–10 kg/dwelling/year (Reay 2004), and nutrients are added into nearshore waters when groundwater moves towards the sea.

There were two coal-fired power stations in Port Augusta; Playford B and Northern power stations. The Playford B power station was commissioned in 1963 with a refit completed in 2005, while the Northern Power station was commissioned in 1985. For many years these power stations discharged very large volumes of thermal effluent to the marine environment south of Port Augusta with the potential to affect overall ecosystem health (Corbin & Wade 2004) such as changing the intertidal faunal composition (Thomas *et al* 1986), reducing seagrass growth, biomass and productivity (Ainslie et al 1994) and altering the growth and community structure of fish species (Jones *et al* 1996, Ralph 1998).

In 2012, Playford B was mothballed ceasing discharges, following by the Northern power station in 2016. It is expected that these closures will return the temperatures of the far northern gulf to a more natural regime, which is likely to reduce the stress on the ecosystem.

Flushing of Winninowie is restricted by the geomorphology at Point Lowly (Nunes Vaz 2014, Figure 2). The reduced water movement, especially in embayments, allows pollutants to accumulate and the water to become warmer than open coastlines (Harbison 1986). The conditions in Winninowie make nutrient pollution particularly noticeable through the rapid growth of opportunistic macroalgae and seagrass epiphytes.

Fitzgerald Bay, northeast of Whyalla (Figure 2), has been used for sea cage aquaculture of Yellowtail Kingfish (*Seriola lalandii*) or YTK. Juvenile fish were produced at a hatchery located at Arno Bay, and transported to sea cages offshore where they are grown to market size. The industry peaked at an annual production of 2,071 tonnes, but high mortalities in 2009–10 caused the end of farming in Fitzgerald Bay. If required, the *Aquaculture (Zones–Fitzgerald Bay) Policy 2008* still allows aquaculture development in Fitzgerald Bay.

The impacts of sea cage fish farming on the surrounding environment are well established (Naylor *et al* 2000, Islam 2005):

- organic loading resulting in directly smothering benthic habitats (Cheshire *et al* 1996)
- disruption to sediment chemistry in the direct vicinity (Lauer *et al* 2009)
- dissolved nutrients contributing to the decline of seagrass habitats in the nearfield (Delgado *et al* 1999)
- far-field eutrophication effects such as seagrass epiphyte loads (Gaylard *et al* 2013b) and algal blooms (Martinez-Porchas & Martinez-Cordova 2012).

Expected condition of Winninowie

The terrestrial environment adjacent to Winninowie is highly modified, which would typically result in rain washing pollutants into the sea, but low annual rainfall leads to relatively low and sporadic runoff. The nearshore waters receive numerous nutrient discharges and the biounit experiences low flushing that may exacerbate the impacts of pollutants in these locations.

Based on this threat assessment, the nearshore marine habitats in Winninowie are likely to be in Fair condition (Table 1). Habitats in fair condition are likely to be in variable condition compared to the reference condition; some areas in good condition with intact and dense seagrass, but also areas in poor condition. There are likely to be symptoms of nutrient enrichment such as epiphytes on seagrass and opportunistic macroalgae. These symptoms are likely to be worse in small embayments with low flushing.

Table 1 **Threat assessment for the Winninowie biounit 2012. For details of threat score weightings, see Gaylard *et al* 2013**

Threat	Threat score
Areas of restricted water movement or likely low flushing	3
Historical impacts	2
Agricultural runoff	2
Urban runoff	1
Dredging	0
Shipping	0
Industrial discharges	3
WWTP/CWMS	2
Septic tanks	3
Aquaculture – supplementary fed (historical)	2
Aquaculture – non-supplementary fed	0
Sum of threats	18
Expected condition	FAIR

Threat assessment scores

Consequence at a regional scale	Detail	Score
Insignificant	Localised impact with a short duration (days)	0
Low impact	Localised impact but with a moderate duration (weeks to months)	1
Moderate impact	Wide or long duration	2
High impact	Wide and long duration	3

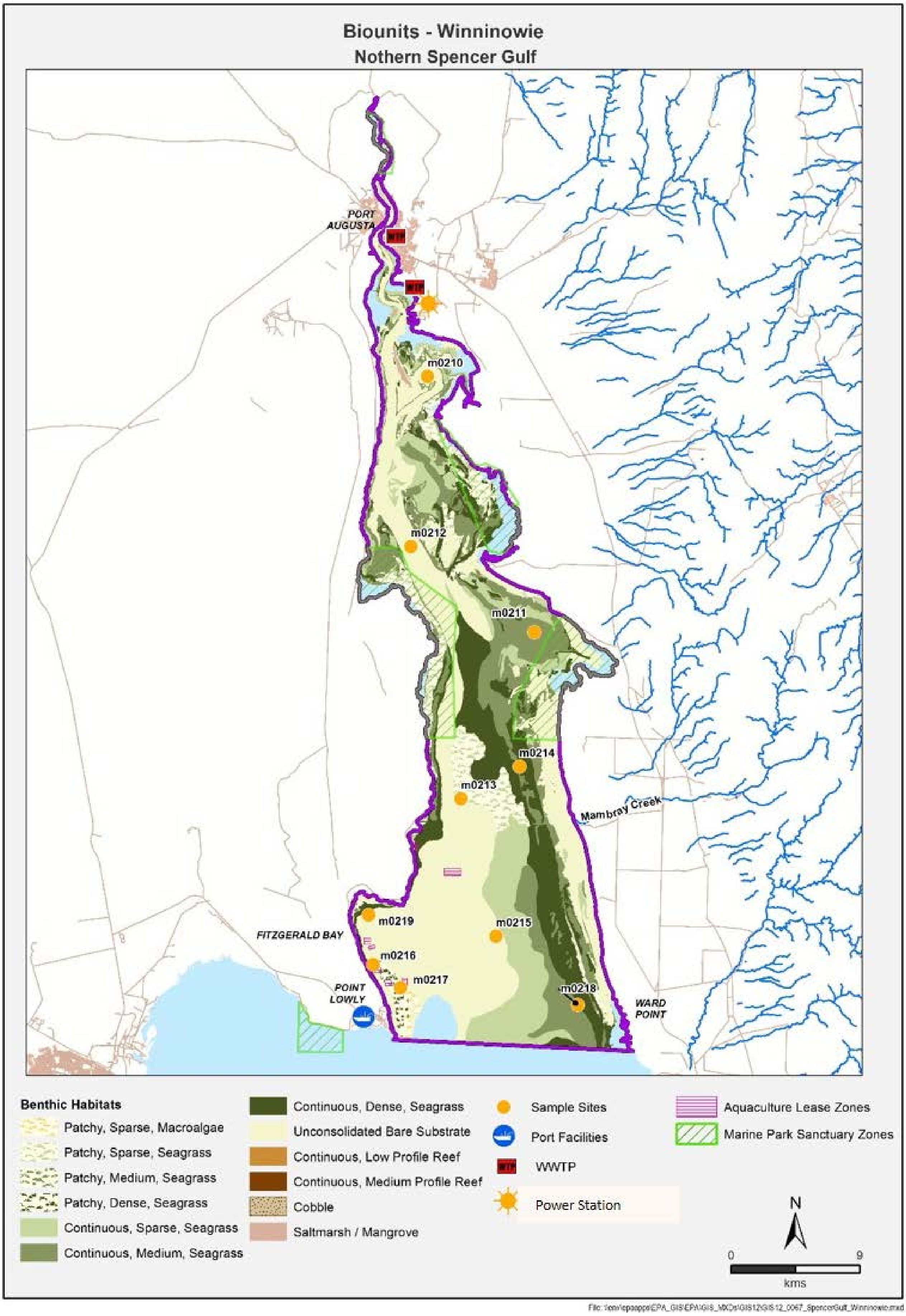


Figure 2 **The Winninowie biounit indicating benthic habitats from the state benthic habitat mapping layers (DEWNR 2010). EPA monitoring sites in 2012 are marked by the large dots.**

2.2 Tier 1 – Yonga

Description

Yonga occupies the central northern waters of Spencer Gulf, south of Winninowie, from Victoria Point (north of the mouth of Franklin Harbour) to Point Lowly on the Eyre Peninsula to Ward Point and Point Riley on the Yorke Peninsula (Edyvane 1999a, Figure 3). Yonga predominantly experiences low wave energy, with slightly higher energy south of Shoalwater Point, where the coast faces the southwesterly winds and occasional ocean swell (Edyvane 1999a). Increased mixing from wind and swells, as well as greater overall water volume means that temperatures and salinities in Yonga do not reach the upper extremes experienced in Winninowie (Nunes Vaz 2014).

The nearshore (<15 m depth) benthic habitats in Yonga comprise seagrass, bare sand and a small amount of reef (Figure 3). Seagrass meadows cover an approximate area of 2,490 km^2 and represents the largest area in South Australia (Edyvane 1999a). The meadows are dominated by *Posidonia australis* and *Amphibolis antarctica* in shallower waters with *P sinuosa* and *P angustifolia* in deeper water (Edyvane 1999a, Shepherd 1983). There are discrete areas where more ephemeral species such as *Halophila ovalis* and *Heterozostera tasmanica* can also thrive (Shepherd 1983). Relatively small areas of rocky shore that give way to subtidal reef are commonly dominated by mixed macroalgal and invertebrate communities (Edyvane 1999a, Shepherd 1983).

Both seagrass and reefs provide important habitat for mating, spawning and migration of commercially and recreationally fished species including Snapper (*Chrysophrys auratus),* and King George Whiting (*Sillaginodes punctata*) [Bryars 2003] as well as other regionally important species such as the Giant Australian Cuttlefish (*Sepia apama*) [Gillanders & Payne 2014].

Tens of thousands of *S apama* aggregate to spawn on the shallow reefs of Point Lowly each winter (Steer *et al* 2013) attracting tourists who come to witness the globally unique spectacle. In recent years, there has been a decline in abundance of *S apama* in NSG (Gillanders & Payne 2014). The South Australian Government is investigating possible reasons for the decline including water temperature, weather conditions, pollution, predators, prey, habitat, disease, fishing pressure and tourism (Steer *et al* 2013). Analysis of the daily average temperature over the embryo development period has shown the strongest signal for explaining the changes in the abundance and biomass of *S apama*. However while temperature regimes are important, other factors including predator-prey relationships and water quality are also likely contributors (Steer 2015). Importantly, the lack of long-term observations meant that natural variability as a cause of decline could not be ruled out (Steer *et al* 2013).

Threat assessment

There are two major urban centres in Yonga; Port Pirie on the northeastern shore and Whyalla to the northwest (Figure 3). The City of Port Pirie is the smaller of the two centres with a population of 17,364, while Whyalla has a population of 21,828 (ABS 2016). A third centre, Port Broughton, with a permanent population of less than 1,225 is located on the southeastern shores of Yonga (ABS 2016) and is a popular holiday destination.

The Whyalla WWTP was commissioned in 1966 and is located to the south of the town near Mullaquana. The plant discharges nutrient rich effluent into a small mangrove lined tidal creek, which flows to the sea (Figure 3). A water reclamation plant was built in 2004 to treat a portion of the effluent for reuse on council parks and reserves, reducing the overall volume discharged to the sea. The Port Pirie WWTP was commissioned in 1971 and discharges nutrient rich effluent into Second Creek; a small mangrove lined tidal creek connected to the sea through tidal action (Figure 3). In 2004 a sequencing batch bioreactor was installed with the aim of reducing the concentration and load of the nutrients being discharged to the creek.

Sewage in smaller townships is managed through a community waste management systems (CWMS) at Port Broughton and a number of small shack communities (eg Cowleds Landing, Murninnie Beach, Cowell) are dependent on septic tanks (Figure 3). In high densities septic tanks can introduce nutrients into shallow groundwater which may flow towards the sea (Reay 2004).

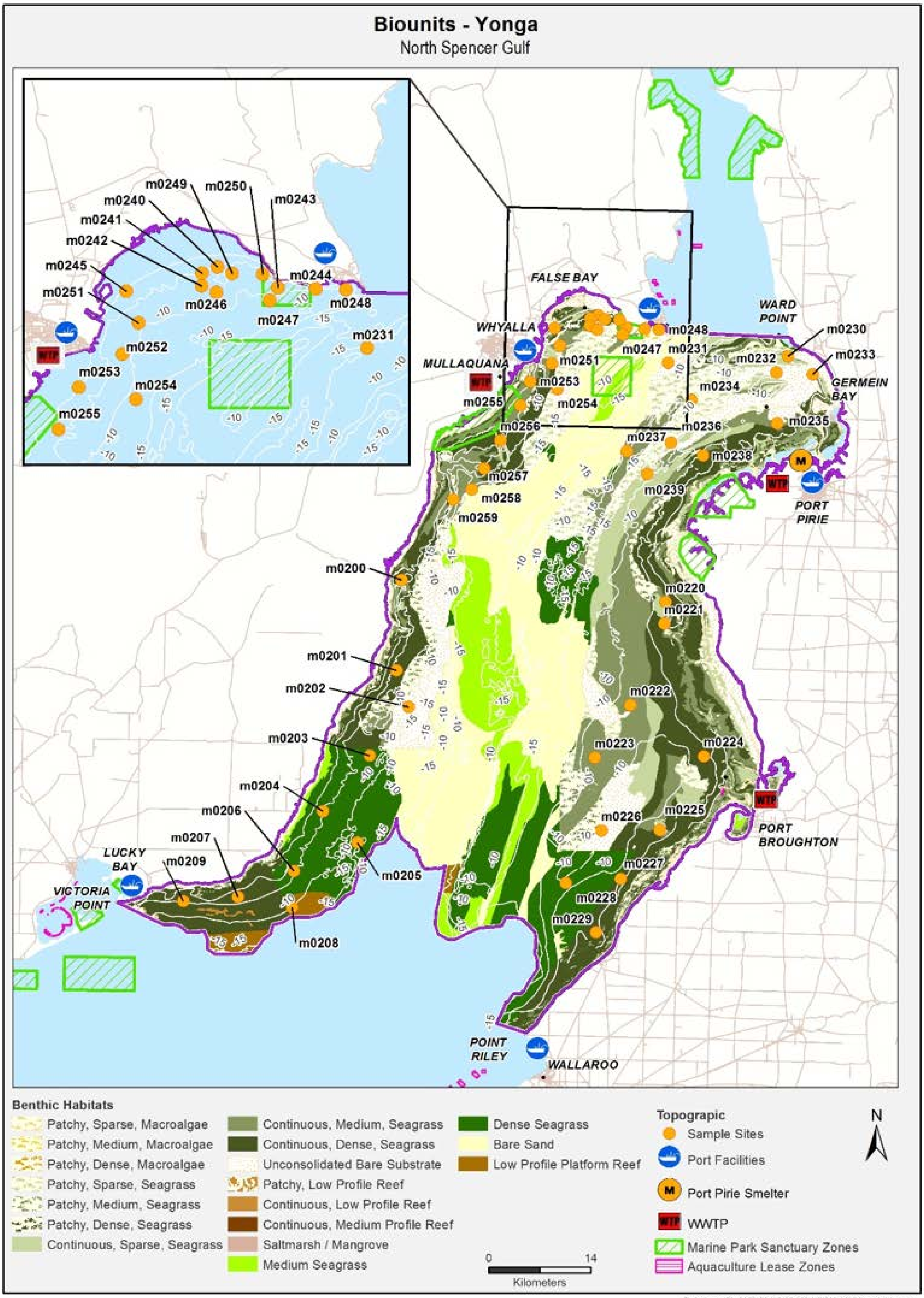


Figure 3 The Yonga biounit indicating benthic habitats from the State Benthic Habitat Mapping layers (DEWNR 2010). EPA monitoring sites in 2012 are marked by the large dots.

Yonga experiences between 200–400 mm of rainfall each year, with the eastern side receiving slightly more than the west. While rainfall is relatively low, large rain events will result in runoff from the urban areas in Whyalla, Port Pirie and surrounding small townships (eg Port Broughton) reaching the sea. Agriculture of cereal crops and livestock are key industries throughout the Yorke and Eyre peninsulas and reduces catchment runoff quality. The main drainage into NSG is through the Broughton River which lacks a coastal discharge under baseflow conditions, but connects to the Broughton estuary under regular seasonal flows and mid-flow events. Agricultural runoff into the Broughton River will transport sediments, nutrients and organic matter to the coast (Favier *et al* 2004). Large floods overflow the river onto low-lying samphire marshes and discharges through numerous tidal streams along the coast (Favier *et al* 2004).

In 1889, the Port Pirie lead and zinc smelter commenced operation on the shores of Port Pirie Creek processing ore from Broken Hill to refined metals and quickly became one of the world's largest lead smelters. Initially, environmental controls for the smelter were minimal and metal rich effluent was discharged directly into the harbour at Port Pirie (Gaylard 2014).

Metals in the marine environment can have profound consequences on biota with long-lasting effects. Contamination from metal discharge is not necessarily limited to the immediate vicinity of the point source. Harbison (1986) reported that depositional areas with low water flow in NSG, many miles from the point of discharge, can accumulate contaminants. Studies investigating metal levels in sediments of NSG showed that within a ~30-km radius of Port Pirie (ie ~600 km^2) sediments contained elevated levels of metals originating from the smelter, which decreased with distance from First Creek (Ward *et al* 1986).

Similarly, seagrasses growing near the Port Pirie smelter have been shown to have elevated levels of cadmium, lead and zinc in the leaves and were less productive than those well away from the smelter (Ward 1987). Additionally, metals reduced or eliminated 20 of the most common fish species that lived among the seagrasses in the contaminated area (Ecos 1983). More recent work by the EPA has shown that metal levels around the discharge site, and in seagrass have reduced substantially since the 1980s (Gaylard 2014). This work also showed that the Port Pirie Harbour area is likely to be acting as a source of pollution with large metal burden in the sediments being released and flowing into Germain Bay on outgoing tides (EPA unpublished data)

The long history of industrialisation in the Yonga biounit has resulted in extensive areas of metal contamination and substantial impacts on the condition of the ecology in some locations, suggesting a widespread ecological effect from metal contamination. While the MER program does not specifically investigate metal contamination, it is possible that metal contamination is contributing to the observed condition of the NSG marine environment. The evaluation of risks from historic and current metal contamination in the marine environment are specifically targeted in other work by the EPA (eg Gaylard *et al* 2011, Corbin & Wade 2004), and are not included in this statewide AECR program.

Expected condition of the Yonga biounit

Yonga has a long legacy of historical metal contamination throughout sediments. Recent work suggests the impacts from metals have decreased over the last 30 years, but there may still be ecosystem scale effects. Additionally, Yonga has numerous large nutrient discharges, substantial cities with stormwater runoff, and agricultural runoff that reach the nearshore environment. The biounit is large and well flushed, aiding the dispersal and dilution of pollution. This threat assessment predicts Yonga to be in Good condition (Table 2). Habitats are generally intact, but show initial symptoms of nutrient enrichment with increased epiphytic algae on seagrass leaves. There may be some changes to ecosystem function and resilience compared to reference condition.

Table 2 **Threat assessment for the Yonga biounit 2012. For details of threat score weightings, see Gaylard et al (2013)**

Threat	Threat score
Areas of restricted water movement or likely low flushing	1
Historical impacts	3
Agricultural runoff	1
Urban runoff	1
Dredging	1
Shipping	2
Industrial discharges	3
WWTP/CWMS	2
Septic tanks	0
Aquaculture – supplementary fed (Historical)	0
Aquaculture – non-supplementary fed	0
Sum of threats	14
Expected condition	GOOD

Threat assessment scores

Consequence at a regional scale	Detail	Score
Insignificant	Localised impact with a short duration (days)	0
Low impact	Localised impact but with a moderate duration (weeks to months)	1
Moderate impact	Wide or long duration	2
High impact	Wide and long duration	3

3 Tier 2 methods

Full details of methods, conceptual models and metrics used in the nearshore MER field program can be found in Gaylard *et al* (2013a). This section will briefly provide a summary and describe any specific aspect for the NSG bioregion with more detail in Appendix 1.

At each site, 10 random underwater 50 m video transects are used to establish habitat type and condition based on a multiple lines of evidence approach. Throughout the site, water is sampled to generate a composite sample representative of the site. While this is acknowledged to be only one point in time, it provides a detailed snapshot to assist in interpreting the habitat condition observed and provides a point of comparison to water quality guidelines (ANZECC 2000).

The habitat condition and water chemistry findings are then interpreted using the conceptual models and biological condition gradient.

3.1 Conceptual models and condition gradient

As well as detailing the methods undertaken to broadly assess ecological condition for the nearshore MER program, Gaylard *et al* (2013a) describes the development of generic conceptual models that have been used to suggest processes of degradation based on established literature (Appendices 2 and 3). They establish a biological condition gradient in response to nutrient enrichment and reduction in water clarity for seagrass, rocky reef and unvegetated sediment habitats in shallow (2–15 m) nearshore waters in South Australia. The condition gradient assumes that habitat condition deteriorates as the degree of human disturbance in the surrounding and adjacent environment increases and conversely, the best condition occurs where there is little to no human disturbance (Appendix 3).

As our understanding of the nearshore marine environment increases these conceptual models will be refined.

4 Tier 2 ecological condition – NSG bioregion

In this section, results are presented for the entire bioregion, discussing broad compositions of habitats monitored and large-scale patterns in habitat composition, condition and water quality.

4.1 Habitat

In 2012, 60 sites were sampled in NSG in autumn (11 April–9 May) and spring (24 September–30 October). Seagrass was the dominant habitat in 83.3% of sites in NSG, while bare sand comprised 15.1% of sites and reef 1.6%. These proportions are consistent with the nearshore regions of the state benthic habitat mapping which had 79.4% seagrass, 19.8% sand and 0.7% reef (DEWNR 2010) suggesting that this is a reasonable representation of the bioregion.

Seagrass throughout the NSG varied from dense, monospecific meadows of *Posidonia* sp. to complex and sometimes patchy beds comprised of two or more species of differing densities (Figure 4). As expected based on the geomorphology of the two biounits, Yonga (51.5%) had more seagrass than Winninowie (31.7%) and there was no difference in seagrass area between autumn and spring (PERMANOVA, p = 0.07), which is consistent with previous reports (Gaylard *et al* 2013b, Nelson *et al* 2013). The conceptual models suggest that in relatively shallow water (< 15 m) dense and continuous seagrass would be expected, where substrate allows. As such, sparse and patchy seagrass may indicate degradation from human activities or it could be due to natural losses from extreme weather events (Seddon 2000).

Hard substrates in NSG have largely been overlaid by sediments that have settled out in the less energetic waters (Gostin & Hill 2014). In a few locations where water movement is sufficient to expose underlying consolidated substrate, reef communities have developed. Rocky reef habitat made up a very minor component of the sites monitored in NSG and the dominant constituents comprised red macroalgae (46.8%) with smaller amounts of brown canopy algae (20.6%) and bare substrate (25.4 %).

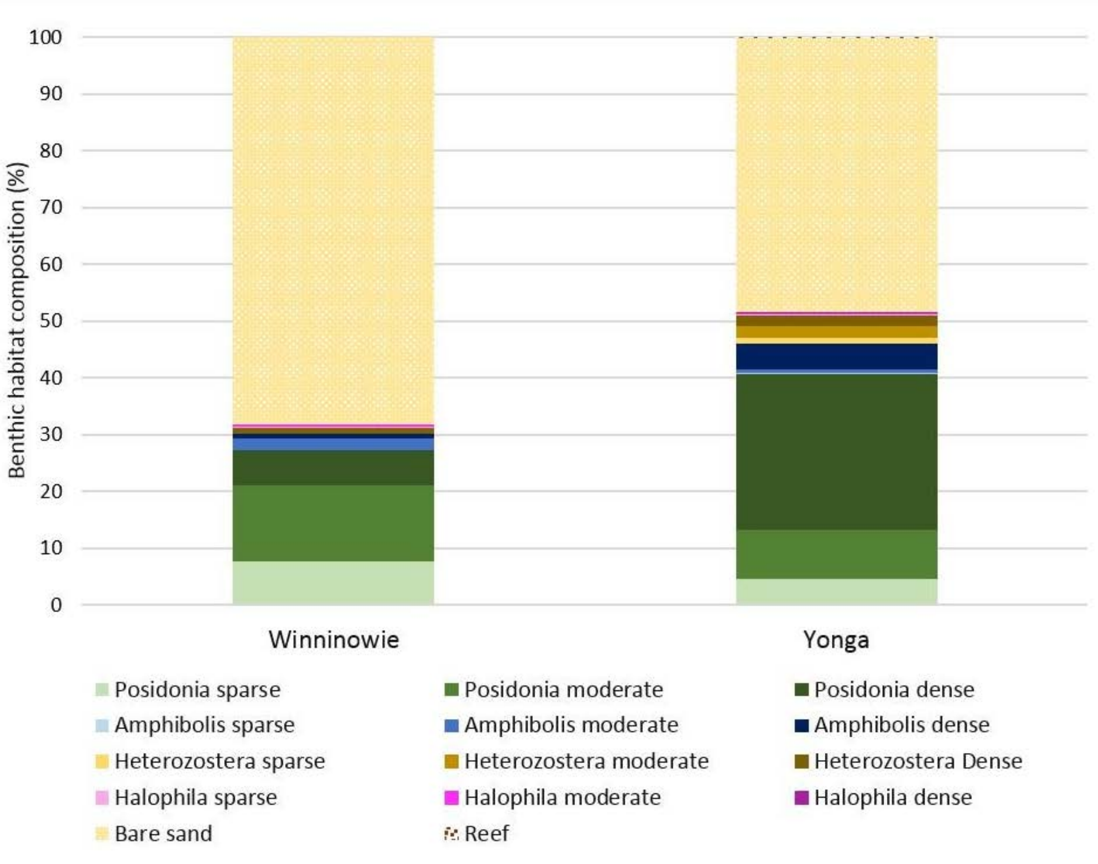


Figure 4 **Benthic habitat composition of Northern Spencer Gulf in 2012**

Figure 5 shows a nMDS plot of seasonally averaged benthic data for both biounits for comparison to reference points. The reference points were developed from the conceptual models show a range of habitats from between Excellent and Very Poor condition (Appendix 2). The spectrum of condition spans the plot with sites in Excellent condition on the left, Moderate condition in the middle of the plot and Very Poor condition on the right of the plot (Figure 5).

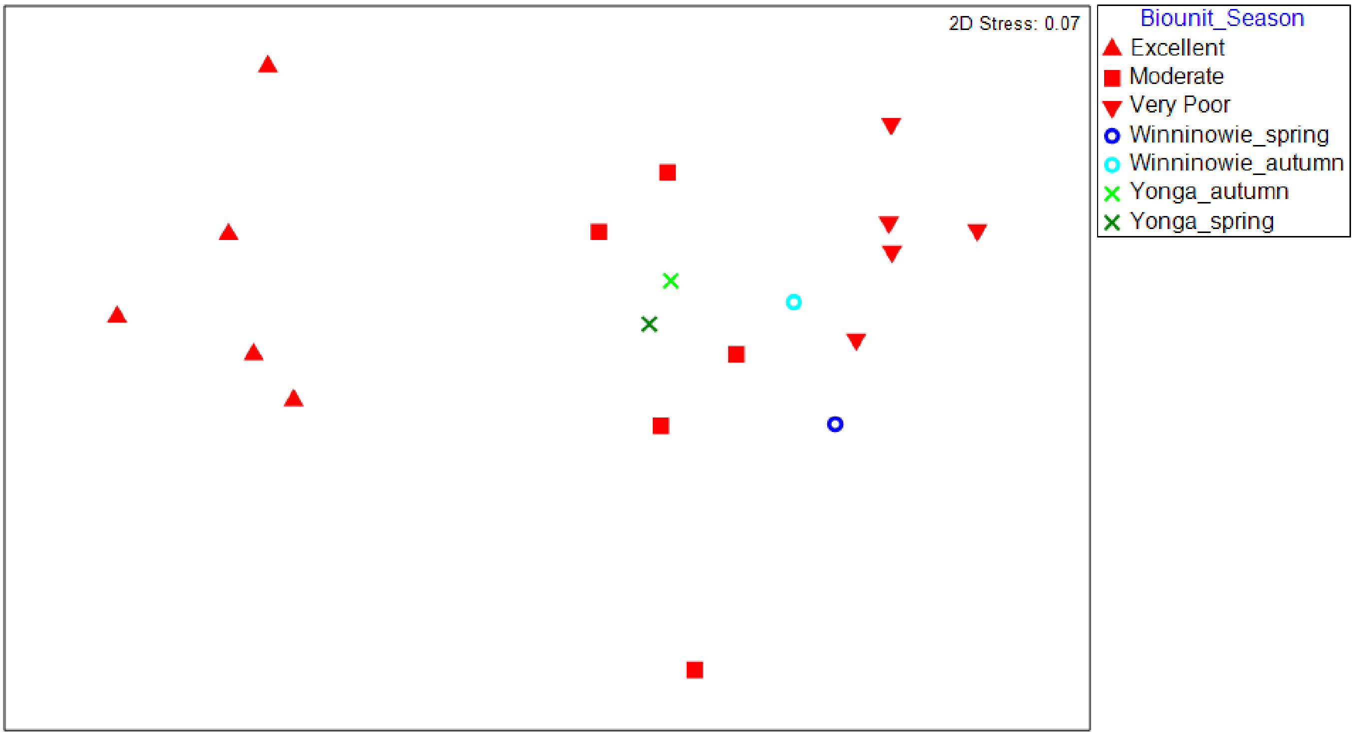


Figure 5 nMDS plot of seasonal benthic habitat composition of the Winninowie and Yonga biounits in 2012. Icons represent pooled, average data from each biounit. Hypothetical reference points are included in red for comparison.

The icons for each season are offset vertically for both biounits, indicating that there is no substantial, seasonal change in condition (Figure 5). In Winninowie, spring data corresponds to lower cover of colonising seagrass species (ie *Heterozoztera* and *Halophila* spp) than autumn, and in Yonga, the spring data corresponds to less epiphyte and opportunistic macroalgae than in autumn. The findings are consistent with increased plant growth that occurs throughout spring and summer as daylight hours and water temperature increase.

4.2 Modifiers

A modifier in the context of the MER program describes indicators that may be transient, but can be symptomatic of stress on the habitat (Gaylard *et al* 2013a). Examples of modifiers include seagrass epiphytes and the presence of opportunistic macroalgae. Epiphytes are a natural part of seagrass meadows and play a role in primary production, nitrogen fixation, nutrient cycling and calcareous epiphytes in particular contribute to sediment formation (Borowitza *et al* 2006). However, seagrass epiphytes can cause seagrass loss when in excess or occur for prolonged duration (Neverauskas 1989, Bryars *et al* 2011). Additionally, rapidly growing algae such as *Hincksia sordida* may proliferate, shading seagrass and reef habitats leading to a decline in habitat condition.

Large amounts of opportunistic macroalgae and seagrass epiphytes typically indicate an excess of nutrients and can be used to make inferences about longer-term water quality (months-years). Differences in epiphyte and opportunistic macroalgal load are particularly useful for providing a seasonal, time integrated perspective of water quality as they respond to seasonal changes in light and nutrient availability (Borowitzka *et al* 2006).

Average epiphyte load (23.4) for the bioregion was considered to be low and was not different between biounits ($p > 0.8$, Figure 6a). Epiphytes displayed different seasonal patterns in each biounit, being higher in autumn for Winninowie, but higher in spring for Yonga. Despite the apparent seasonal difference, this is not statistically significant ($p > 0.5$, Figure 6a). The lack of seasonality in epiphyte load in NSG indicates that environmental factors do not appear to be a large influence on epiphyte growth at the bioregional scale, but site-specific differences may be evident.

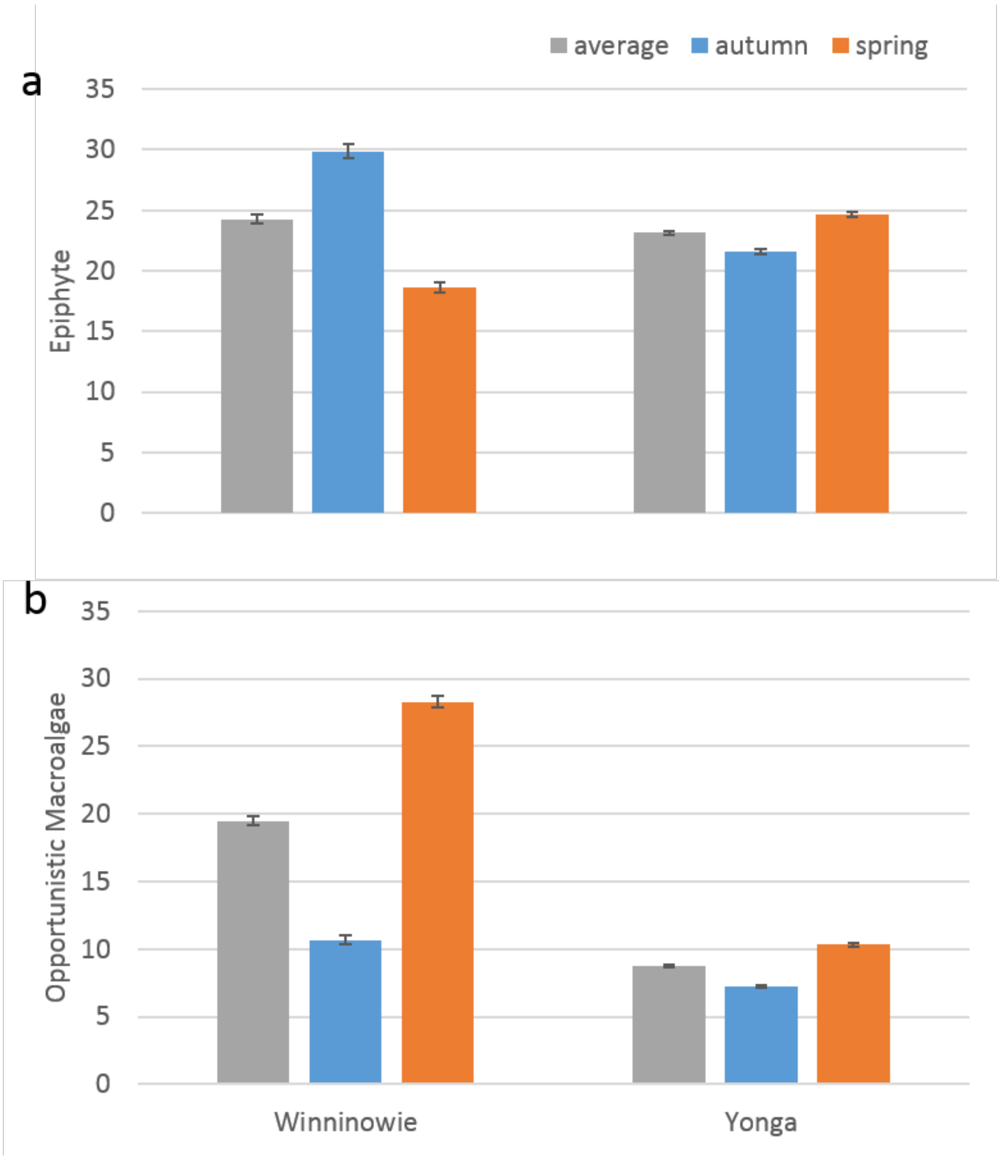


Figure 6 **Average annual and seasonal a) Seagrass epiphyte load (score out of 100) b) Opportunistic macroalgae (percent cover) for Northern Spencer Gulf during 2012. Error bars represent the standard error.**

Average opportunistic macroalgae cover (14.1) for the bioregion was low and significantly different between biounits ($p < 0.02$. Figure 6b). In both biounits, opportunistic macroalgae cover was significantly higher in spring, than autumn ($p < 0.03$, Figure 6b), and Winninowie had significantly more than Yonga ($p < 0.02$, Figure 6b). The seasonal difference in opportunistic macroalgae is consistent with a response to increased light and temperature that occurs from summer to autumn (Chavez *et al* 1999). The shallower depth, warmer temperature and reduced water flow in Winninowie compared to Yonga may help to explain the significant difference between the two biounits.

Water chemistry can provide a valuable insight into the nutrient availability that may allow epiphytes and opportunistic macroalgae to proliferate, while turbidity can impact habitat condition through decreasing light availability. Water chemistry can be highly variable through space and time and the snapshot of results provided as part of this report does not provide the same level of confidence as the longer term, integrated response to water quality of benthic habitat data. However, even limited water chemistry data adds an extra line of evidence that may be useful in the interpretation and conclusions about the marine habitat.

A principal component analysis (PCA) plot was used to describe the seasonal water quality for the biounits within NSG (Figure 7). The plot shows that there were differences between the biounits and that there were differences between seasons for both biounits. Chlorophyll *a* has a strong influence in Winninowie in autumn which may be driven by the restricted flushing and relatively shallow depth of the biounit leading to warmer water temperatures, creating ideal conditions for algal growth. Similarly turbidity is a strong influence in Yonga, with waters typically more turbid in autumn compared to spring, which may be due to the stronger afternoon breezes during summer.

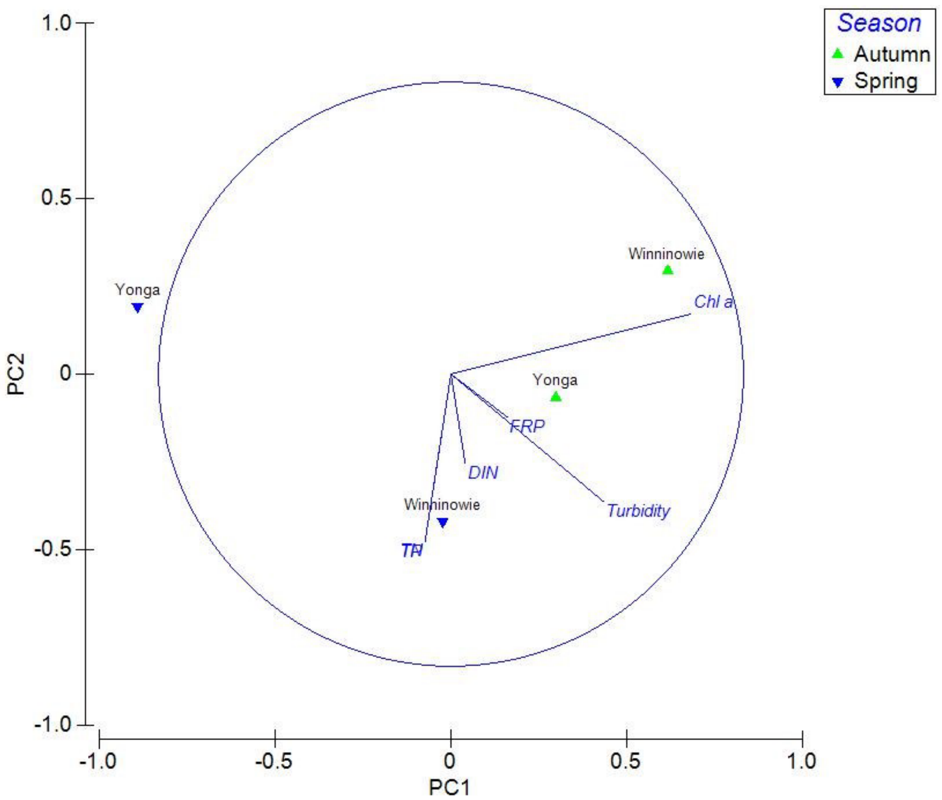


Figure 7 **PCA of average seasonal water chemistry results for biounits within the Northern Spencer Gulf bioregion in 2012. The first two principal components (PC1 and PC2) account for 86.8% of the variability. TN = total nitrogen, TP = total phosphorus, DIN= dissolved inorganic nitrogen, FRP = filtered reactive phosphorus, Turbidity, Chl a = Chlorophyll *a*.**

5 Tier 2 ecological condition – biounit

This section documents the two biounits within the Northern Spencer Gulf bioregion. The sections consider more detailed evaluation of the habitats monitored and patterns in composition, condition and water quality.

5.1 Winninowie biounit

Habitat

A total of 10 sites were monitored in Winninowie over autumn and spring in 2012 (Figure 2). Seagrass was present in 80% of sites, 20% bare sand and reef was not encountered (Figure 8). The habitat proportions are not substantially different than the 64.3% seagrass, 34.7% sand and < 0.8% reef for the nearshore area contained within the state habitat benthic mapping (DEWNR 2010) and these results are a good representation of the known major habitats in Winninowie.

Figure 8 **Annual average benthic habitat composition for sites within the Winninowie biounit. No epiphyte or opportunistic macroalgae was observed at Port Paterson.**

Seagrass cover, density and species composition were highly variable between sites within Winninowie. No seagrass was recorded in Fitzgerald Bay north (m0216) or Fitzgerald Bay south (m0217), and only a small area of moderate and sparse *Posidonia* was found at Fitzgerald Bay inner (m0219) in Figure 8.

Sites on the eastern side of the gulf (eg Miranda, Mount Gullet, Ward Point) have more than 80% seagrass cover (Figure 8), which is likely due to wide, relatively shallow gradient from the shore. However, the shallow waters provide good conditions for algal growth, with epiphyte and opportunistic macroalgal indices above 40 for the sites with greatest seagrass cover. Winninowie has numerous deep channels with seagrass largely confined to the shallower waters less affected by the high current speeds (Figure 2). Colonising seagrasses (*Halophila* sp and *Heterozostera* sp) at Port Paterson, Blanche Harbour and Baroota may be reflective of the high water currents and more mobile sediments that may preclude larger, longer-lived species observed at Miranda, Mount Gullet and Ward Point.

Appendix 2 outlines the conceptual models for seagrass habitats in South Australia. A fundamental part of this MER is an understanding of whether the environmental conditions are suitable for seagrass to grow at a particular location where it has been recorded as absent. Once a suitability assessment has been conducted, a measure of seagrass extent can be used to define its condition. The methods for habitat suitability assessment are detailed in Gaylard *et al* (2013a). The essence of the habitat suitability assessment is a literature search to consider variables including the depth of water, sediment particle size, an estimate of wave energy based on fetch, water flow (ie restricted water flow), proximity of seagrass, turbidity, other evidence of nutrient impacts, and the presence of stressors that may have contributed to loss. The sites at Fitzgerald Bay north and south (m0216 and m0217, respectively) had no observed seagrass on the site. In order to determine whether there are any obvious factors that may preclude seagrass from growing at that location, a seagrass habitat suitability assessment was conducted for both sites (Table 3).

Table 3 Seagrass habitat suitability assessment for sites in Winninowie biounit

Parameter	Fitzgerald Bay north (m0216)	Fitzgerald Bay south (m0217)
Depth	10–12 m	10–11 m
Particle size	Sand (63 µm–2 mm)	Sand (63 µm–2 mm)
Profile	Flat (< 25 cm)	Flat (< 25 cm)
Wave energy	Low	Low
Water flow	Moderate	Moderate
Adjacent seagrass	Very close (< 350 m)	Close (1,200 m)
State benthic habitat mapping	Patchy, dense seagrass	Patchy, dense seagrass
Significant stressors	Historical extensive finfish (YTK) aquaculture	Historical extensive finfish (YTK) aquaculture
Other evidence	High bio-turbation	High bio-turbation
Conclusion	Habitat appears suitable	Habitat appears suitable

The seagrass habitat suitability assessment suggests that the environmental conditions are well within the tolerances of seagrasses in South Australia. As such there is nothing obvious within the physical environmental factors that would preclude seagrass growth at the sites (Table 3). It should be noted that this assessment is coarse and conclusions considered qualitative.

Fitzgerald Bay has been used extensively for YTK aquaculture between 1999 and 2011. Previous studies have noted the presence of seagrass in Fitzgerald Bay (Tanner & Fernandes 2010, Parsons-Brinkerhoff & SARDI 2003, DEWNR, 2010), while Tanner and Fernandes (2010) have shown ambiguous results with respect to impacts on seagrass in Fitzgerald Bay from YTK aquaculture undertaken in 2004; a period when the industry was in its infancy. After this time the biomass of YTK aquaculture at Fitzgerald Bay substantially increased. It is possible that nutrient loading may have affected seagrass abundance.

Impacts from finfish aquaculture have been demonstrated to include elevated abundances of some benthic taxa, benthic microbial mat (Cheshire *et al* 1996), epiphytes (Rountos *et al* 2012) and seagrass loss (Ruiz *et al* 2010). At the time of monitoring, seagrass cover was very sparse and patchy, typically less than 9% in the sites associated with Fitzgerald Bay (Fitzgerald Bay inner, south and north) and the benthic habitat shows signs consistent with impacts from increased nutrient loading; Fitzgerald Bay north (m0216) had signs of elevated populations of burrowing taxa (Figure 9a), microbial mat was observed at Fitzgerald Bay south (m0217) in Figure 9b and heavy epiphyte loads were observed at Fitzgerald Bay inner (m0219) in Figure 9c. Additionally, evidence of high bio-turbation observed by Tanner and Fernandes (2010) may support the presence of residual nutrients in the sediment, and may impede seagrass recolonization.

Figure 9 Example of benthic habitat at (a) Fitzgerald Bay north (m0216) showing high bioturbation, (b) Fitzgerald Bay south (m0217) showing micro-phytobenthic mat and (c) Fitzgerald Bay inner (m0219) showing dense epiphyte loads on seagrass

5.1.1 Modifiers

There was a higher cover of seagrass epiphytes in autumn than spring for Winninowie (Figure 6a) whereas opportunistic algal cover (eg *Ulva* sp) was higher in spring than autumn (Figure 6b). The epiphyte load varied considerably, from no epiphytes at Port Paterson (m0210) through to moderate epiphyte load (55 out of 100) at Miranda (Figure 8). Epiphytes, by definition will only grow on other plants so will not be present at sites where seagrass was absent. Rather, these sites can show nutrient enrichment through other metrics including opportunistic algae, microphytobenthos or bio-turbation (eg Fitzgerald Bay south, m0217).

An nMDS of the habitat data, epiphyte load and opportunistic macroalgae was used to assess similarities or patterns in site composition. Figure 10 shows the majority of sites in Winninowie are clustered towards the right of the nMDS plot along with the Very Poor reference sites. Habitat data in Figure 8 shows that these sites were largely devoid of seagrass aligning with the Very Poor conceptual models. Miranda (m0211) and Mount Gullet (m0214) lie in the centre of the plot consistent with the Moderate reference sites, and Ward Point (m0218) was the only site to be close to the Excellent reference sites. Those three sites were typified by more than 80% cover of moderate and dense seagrass meadows but also had moderate cover of epiphytes and/or opportunistic algae. These results may reflect the shallower slope being more suitable for seagrass growth compared to those in the channels. However these shallower gradients may also result in warmer waters providing ideal growing conditions for seagrass epiphytes and algae.

Median nutrient concentrations were either similar to or less than those for the reference locations with the exception of turbidity (Table 4). The relatively low levels of dissolved inorganic nutrients (DIN) suggests that any excess nutrients in the system are likely to have been taken up by seagrass, epiphytes and opportunistic algae (Table 4). The PCA shows patterns in the data with a broad north-south gradient evident, with both turbidity and chlorophyll *a* being key drivers, particularly in the northernmost sites (Port Paterson and Blanche Harbour) in Figure 11.

The two northernmost sites, Port Paterson (m0210) and Blanche Harbour (m0212) recorded autumn averages of 1.57 NTU and 1.45 NTU respectively. This increased chlorophyll is likely to reflect the lower flushing rates in the northernmost parts of the gulf (Nunes and Lennon 1986). These sites are also located closest to Port Augusta, and Port Paterson is located within an embayment where the shallow, warm, well-lit waters provide excellent growing conditions for phytoplankton. This higher turbidity may be related to the increased chlorophyll in the same areas, with phytoplankton contributing to the scattering of light through the water column.

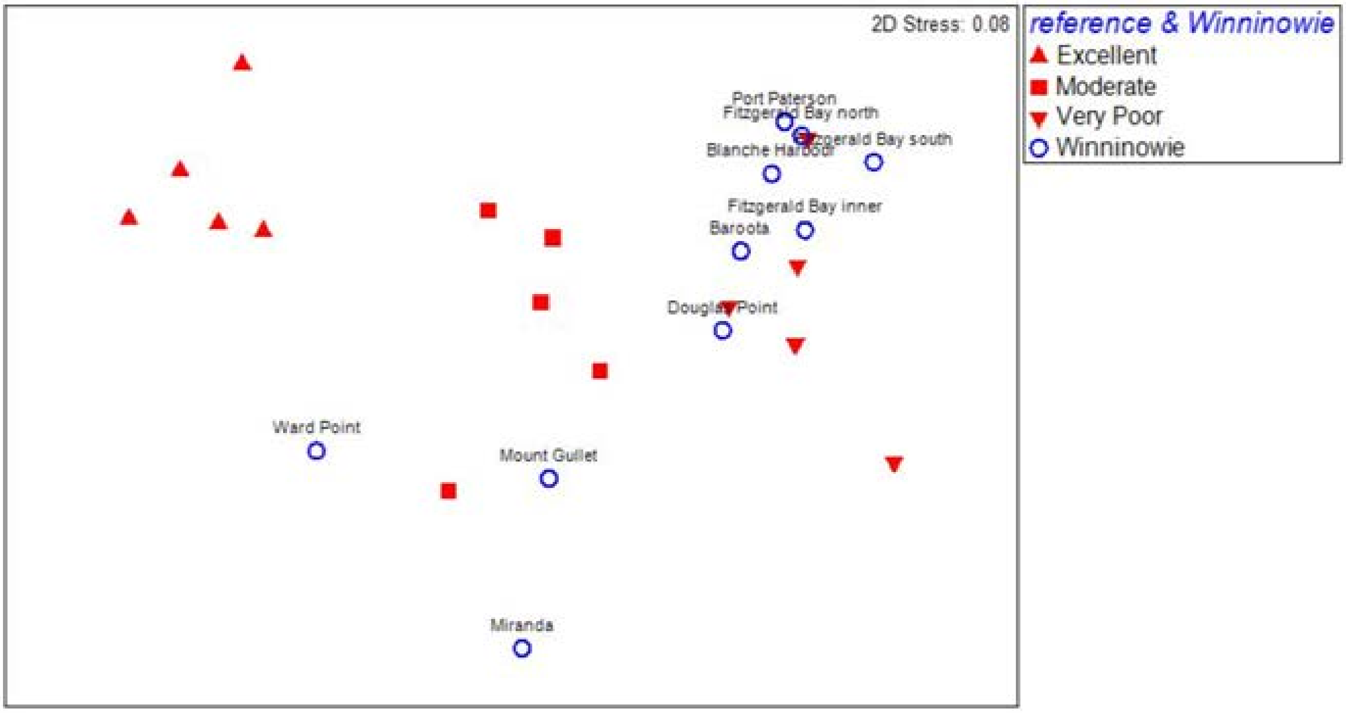


Figure 10 **nMDS plot of site averaged seagrass density and cover, epiphyte load and opportunistic algae for sites in Winninowie. Hypothetical reference sites included for comparison.**

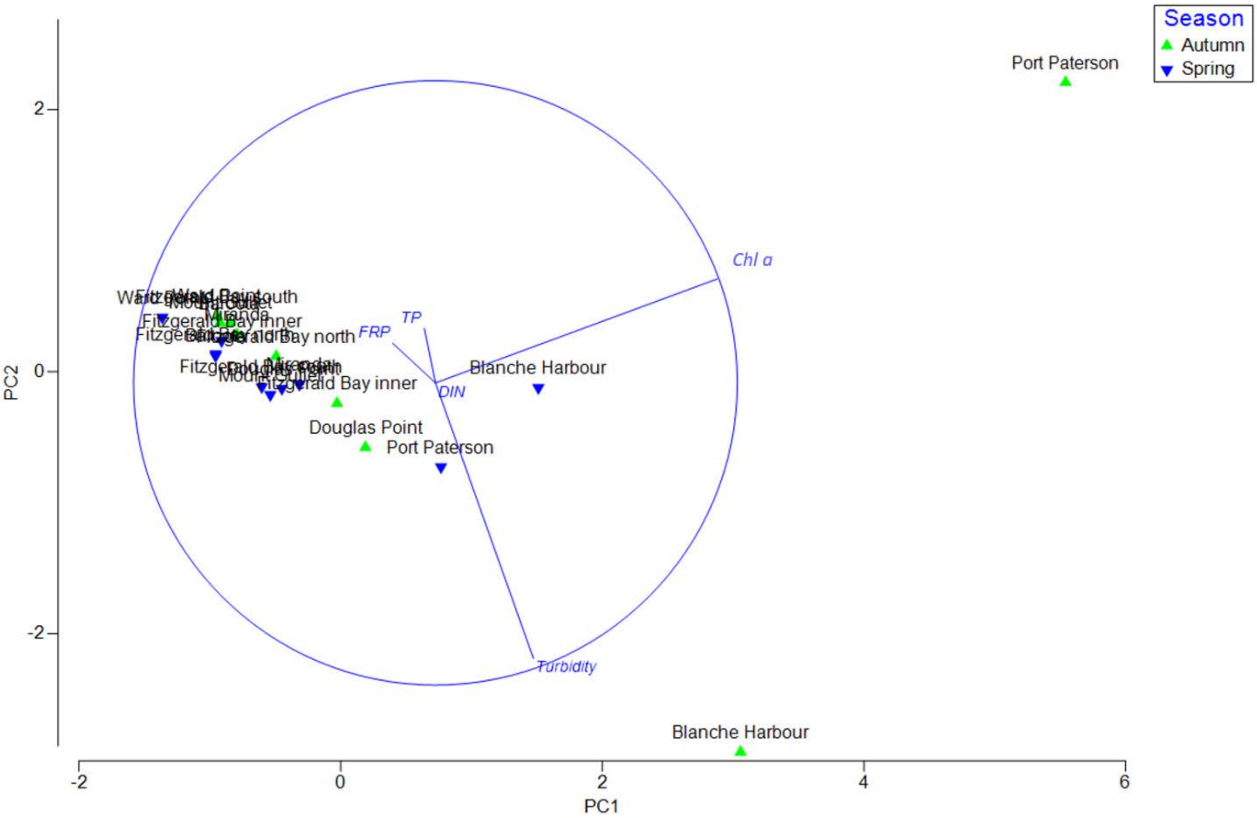


Figure 11 **PCA of seasonal average water chemistry results for the Winninowie biounit. The first two principal components (PC1 and PC2) account for 90.6% of the variability. Turbidity, Chl *a* = chlorophyll *a*, TP = total phosphorus, DIN = dissolved inorganic nitrogen, FRP = filtered reactive phosphorus.**

Table 4 Annual median water chemistry and chlorophyll *a* values for Winninowie biounit in 2012. Bold values indicate results significantly higher than reference.

	Dissolved inorganic nitrogen (DIN) (mg/L)	Total nitrogen (mg/L)	Filtered reactive phosphorus (mg/L)	Total phosphorus (mg/L)	Turbidity (NTU)	Chlorophyll *a* (µg/L)
Median	0.004	0.152	0.002	0.015	0.315	0.435
Standard deviation	0.002	0.062	0.002	0.006	0.280	0.757
n	60	60	60	60	60	42
Reference median	0.018	0.150	0.005	0.015	0.190	0.627
Mann-Whitney significance at $p < 0.05$	0.000	0.768	0.000	0.409	0.000	0.195

5.2 Conclusion

The aquatic ecosystem condition report (AECR) for Winninowie concluded the biounit was in Fair condition, which was consistent with the desktop threat assessment (Section 2.1). The results from the sampling showed highly variable habitat conditions, with some areas having largely intact seagrass meadows eg Miranda (m0211), Mount Gullet (m0214) and Ward Point (m0218), while other locations were totally devoid of seagrass where it would normally be expected to grow eg Fitzgerald Bay north (m0216) and south (m0217). Widespread and sometimes elevated seagrass epiphytes and opportunistic algae at sites in Winninowie indicates that the area can be subject to localised nutrient enrichment. The high variation is likely to be affected by strong tidal regimes, heat and salinity stress and current and/or historical nutrient and thermal inputs.

There was a north-south gradient in the water chemistry with chlorophyll and turbidity being key drivers in the northernmost sites, which may reflect lower flushing. Potential transport of nutrients from other biounits through gulf-wide circulation (Nunes Vaz 2014), restricted water exchange in small bays, and shallow, well lit, warm waters may make Winninowie especially susceptible to the impacts of nutrient enrichment.

Despite the long historical use of Winninowie, this is the first program to assess the condition of nearshore marine habitats between 2–15 m deep. As such, it is not possible to know whether these habitats have changed over time or are currently in transition to more degraded state, are stable or improving. This report does provide an important baseline against which future monitoring or investigation may be compared.

5.3 Pressures and management responses

This section highlights the current or historical pressures on the nearshore marine ecosystem and the management responses that are attempting to address these issues.

The Port Augusta East WWTP (Figure 2, Table 1) discharged 20 tonnes of nitrogen in 2010–11 into a small tidal creek which enters the marine environment near Hospital Creek (NPI 2012). In an attempt to address the wastewater volumes discharges to the sea, the Port Augusta City Council operates a sewer mining wastewater reclamation plant (WWRP) which is used for irrigation of council open spaces. SA Water supplies the WWRP with low salinity wastewater and in 2012 assisted in doubling capacity of the facility by constructing a new pump station and connecting the Port Augusta Prison to the network. SA Water's aim is to assist council to maximise reuse, reduce effluent flowing to the Port Augusta East and reduce the concentration and load of nutrient discharges to the marine environment.

Other sources of nitrogen into the biounit are stormwater runoff from urban and agricultural areas. These are more difficult to quantify than from industry (Gaylard 2014), but due to the low annual rainfall of the area, this discharge is likely to be sporadic. Shack communities use septic tanks to treat sewage which contributes nutrients into shallow groundwater. This creates the potential for nutrient rich groundwater flow from septic systems into the marine environment from shacks along both sides of the gulf. Since 2003, the Port Augusta City Council has required all new developments and significant civil upgrades to include Stormceptor® technologies to treat urban runoff prior to marine discharge. This is an ongoing initiative that will help reduce the quantity of nutrients and other pollutants entering the marine environment.

The historic use of the Fitzgerald Bay region for extensive sea cage aquaculture discharged substantial loads of nutrients into the nearshore waters (~200 tonnes of nitrogen per year at full production). The YTK industry briefly thrived in Fitzgerald Bay until disease caused significant stock loss and fish were moved into waters further south. The accumulation of nutrients into embayments with restricted flushing has the potential to cause impacts on benthic habitats such as seagrass. PIRSA Fisheries and Aquaculture requires an annual environmental monitoring program (FEMP) involving sampling sediment adjacent to actively farmed sites and using DNA profiling to measure changes in the benthic community compared to established control sites. In the event of poor results, PIRSA, the finfish industry and SARDI has responsibility for follow up action through the 10-point FEMP plan of action.

Two power stations at Port Augusta use very large volumes of seawater to cool their plants. The discharge of large volumes of thermal effluent from has shown little effect on seagrass communities (Wiltshire & Tanner 2010). Consistent with literature on the effect of thermal effluent on seagrasses in Gulf St Vincent (Ainslie *et al* 1994), however the increased temperature may have indirect effects on water quality, nutrient enrichment and heat stress on an ecosystem already reaching its upper temperature tolerance (Darling & Cote 2008). The power stations have closed and it is likely that the ambient temperature regime will return to a more natural system, and there may be improvement in ecological condition.

The restricted water movement and resultant high residence times (Nunes Vaz 2014) in Winninowie, are likely to exacerbate the effect of any pollution input compared to well flushed sites, creating a disproportionate impact. Additionally, Winninowie is at the northern most extent of Spencer Gulf and waters within the gulf circulate in a clockwise direction (Kämpf 2014), therefore the biounit may be subject to nutrient transport from water along the western side of Spencer Gulf.

5.4 Yonga biounit

Yonga has been affected by a long legacy of metal contamination due to discharges from land-based sources at Port Pirie and Whyalla (Gaylard 2014). This has changed the habitats (Ward *et al* 1986, Ward and Hutchings 1996) and possibly may be preventing recovery in some areas, and influencing the interpretation of the results in this document.

Habitat

A total of 50 sites were monitored in Yonga in autumn and spring in 2012 (Figure 3). Seagrass was the dominant habitat in 86% of sites, while 12% comprised bare sand and 2% reef. These results are consistent with the state benthic habitat mapping of 82.3% seagrass, 16.9% sand and < 1% reef (DEWNR 2010) again suggesting that this assessment is a good representation of the known major nearshore habitats in Yonga.

Seagrass cover, density and species composition were highly variable between sites (Figure 12). Plank Point (m0201), Conifer south (m0205), Tickera Bay inner (m0227) and Western Shoal north (m0257) all had more than 80% cover of seagrass dominated by *Posidonia* or *Amphibolis* sp (Figure 12). Conversely, no seagrass was observed at Ward Spit (m0231), Cockle Spit (m0232) or Tickera Bay outer (m0228), and seagrass only covered 0.6% of Glensea (m0202).

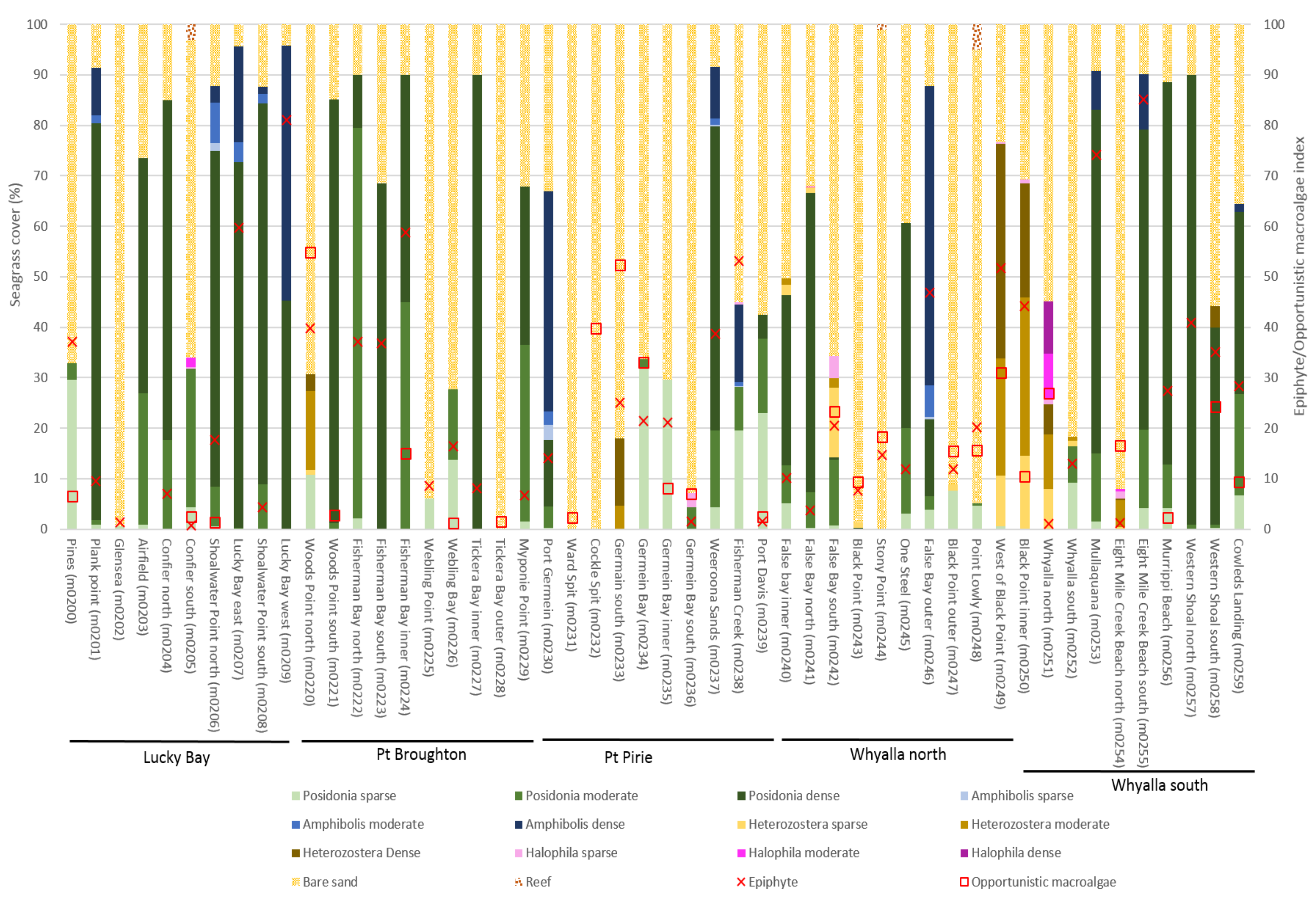


Figure 12 **Annual average benthic habitat composition for sites within the Yonga biounit**

As stated for the Winninowie biounit, a fundamental part of this MER program is a judgement of whether seagrass could grow at a particular location where it has been recorded as absent. Only after this can a measure of seagrass extent be used to define its condition. Four sites in Yonga recorded no or very little seagrass and a reconstruction of seagrass habitat suitability has been undertaken and outlined in Table 5.

Table 5 Seagrass habitat suitability assessment for sites in the Yonga biounit

Parameter	Glensea (m0202)	Tickera Bay outer (m0228)	Ward Spit (m0231)	Cockle Spit (m0232)
Depth	7–10 m	13 m	14–15 m	5–7 m
Particle size	Sand (63 µm–2 mm)	Sand (63 µm–2 mm) and pebble (4–64 mm)	Sand (63 µm–2 mm)	Sand (63 µm–2mm)
Profile	Flat (< 25 cm)	Flat (< 25 cm)	Flat (< 25 cm)	Flat (< 25 cm)
Wave energy	Low	Low	Low	Low
Water flow	High current flow	Possible high current flow	High current flow	High current flow
Adjacent seagrass	200 m	7,000 m	4,500 m	1,500 m
State benthic habitat mapping	Bare sand	Dense seagrass	Bare sand	Bare sand
Significant stressors	None	None	None	Possible influence from the Port Pirie heavy industry and shipping channel
Other evidence	None	None	None	High bio-turbation
Conclusion	High current speeds may prevent seagrass growth	High current speeds may prevent seagrass growth	High current speeds may prevent seagrass growth	Geomorphology may prevent seagrass growth (shipping channel)

The conceptual models used for this MER program (Appendix 2) state that without knowledge of historical (pre-1970s) benthic habitat composition and extent, it can be hard to determine whether habitats including seagrass may have existed in an area, have been lost due to disturbance or whether bare sand is a natural state. The small amount of *Posidonia* seagrass observed at Glensea was considered to be unusual for the area. The seagrass habitat suitability assessment indicated that fast currents may erode sediment, preventing seagrass persistence. The seagrass at Glensea could be opportunistic and unlikely to represent a persistent seagrass meadow. Table 5 suggests that Glensea, Ward Spit, Tickera Bay outer and Cockle Spit are likely to be unsuitable for seagrass growth and as such they have been assessed using the conceptual models and disturbance gradient appropriate for unvegetated sediments (Appendix 2).

The site at Conifer south (m0205) had areas of sand and sparse seagrass interspersed among the low, cobble reef (Figure 12). While some seagrass has been able to colonise areas of cobble where sand provides enough depth for root growth, much of the substrate is unlikely to be suitable for dense seagrass meadows. This is likely related to the faster bottom current speeds found in this region (O'Connell 2016). Areas of the site that captured cobble reef were excluded from the AECR assessment, but transects where reef was absent and appeared suitable for seagrass growth were included in the analysis of condition.

In general, the sites on the western side of Spencer Gulf in the Yonga biounit were dominated by dense and continuous seagrass suggesting that section of the biounit reflects the very low level of human activity and lack of any terrestrial runoff source in the adjacent land. Much of the seagrass in this area forms mixed meadows of moderate or dense *Posidonia* spp and *Amphibolis* spp However, sites adjacent Whyalla and False Bay showed high variability, ranging from bare sand with small sections of rocky reef at Stony Point (m0244) to over 87% seagrass cover at False Bay outer (m0246, Figure 12). False Bay inner (m0240), north (m0241) and south (m0242), and Onesteel (m0245) had between 35% and 78% seagrass.

The steelworks facility at Whyalla discharges significant loads of dissolved nutrients into False Bay (Gaylard 2014) resulting in the loss of an estimated 20 km^2 of seagrass in False Bay leading up to the 1990s (Harbison & Wiltshire 1993, Irving 2014). While nutrient discharges from the facility are still substantial, modifications to the discharge configuration beginning in the early 1990s has increased the retention time of the ammonia prior to reaching the nearshore marine environment, allowing greater biological assimilation. As a result of this and other improvements (eg large reed bed) at the facility, seagrass near the breakwater wall is regrowing (SEA 2014) and the habitat appears to be recovering, which is supported by the moderate to dense seagrass cover recorded in this survey.

On the eastern side of the gulf, habitats close to Port Pirie were variable with seagrass cover (*Zostera sp*) exceeding 90% at Weeroona Sands (m0237), but less than 10% at Germein Bay south (m0236). Throughout Germein Bay inner (m0235) *Posidonia* sp cover averaged 29% (Figure 12) and is closest to the town of Port Pirie and the WWTP. The nutrients discharged from the WWTP, turbidity caused by ships entering or leaving Port Pirie and the metals from the Port Pirie smelter are potential stressors to seagrass habitats.

Similarly, Germein south (m0233) is located close to the shipping channel and consisted of low cover of *Heterozostera* sp (Figure 12). It is unknown at this stage whether Germein south, Germein Bay inner and other sites close to Port Pirie are undergoing re-colonisation along the accepted species progression after disturbance as described by Clarke & Kirkman (1989), or whether the area may be unsuitable for other species. Comparatively, Port Davis (m0239) is in a similar depth of water to Germein south, Germein Bay inner and seagrass meadows consisting of various densities of the apex species *Posidonia* cover ~42%.

Further south towards Port Broughton, the habitat was generally in good condition with moderate to dense seagrass, which largely reflects the low permanent population of Port Broughton and correspondingly low anthropogenic inputs from the surrounding area. Two sites studied in the MER program for NSG; Woods Point north (m0220) and Woods Point south (m0221), were also in the area studied as part of an investigation into large-scale seagrass dieback due to high temperatures and extreme low tides between 1987 and 1994 (Seddon *et al* 2000).

Both m0220 and m0221 are in the region that experienced severe seagrass dieback. Currently, over 80% of Woods Point south was covered by dense *Posidonia* sp, whereas at Woods Point north, only 11% of the site was covered and was sparse in density. The remaining seagrass cover at m0220 (20%) was moderate or dense *Heterozostera* sp. It is unknown why the seagrass recovery is different between the sites, but may be influenced by hydrodynamics at the local scale which show an increase in bottom shear stress in this region (O'Connell 2016). These hydrodynamic conditions are likely to be responsible for the low (< 30%) cover of sparse and moderate seagrass at Webling Point (m0225) and Webling Bay (m0226) in Figure 12).

Only two sites in Yonga had reef habitat, Stony Point (m0244) and Point Lowly (m0248) in Figure 12. Across the sites, reef was only a small component of the total area at both sites, but despite this, the data provides further insight into nearshore marine ecosystem condition. The reef at Stony Point was dominated by red fine branching algae (50.7%) and red coarse branching algae (2.1%) in Figure 13. Turfing algal or bare substrate accounted for 33.3%, and the remaining reef area comprised ascidians or other organisms. The algal community at Point Lowly was different from that at Stony Point dominated by red fine and coarse branching macroalgae (40.8%) in Figure 13. While brown macroalgae covered 37.8 %, comprised largely of brown coarse branching. There were relatively low areas of bare substrate or turfing algae (Figure 13).

The ecology of low energy reefs that occur in the sheltered waters of NSG are not well understood and for this assessment the reefs have been assumed to be similar in characteristic to reefs studied in the Reef Health program (Turner *et al* 2007). Using the Reef Health criteria, the reefs along this coast appear degraded. However, there is significant uncertainty about the response of the reef communities to stressors such as nutrients, and the condition is considered to be a coarse assessment. Presently there is not enough evidence to determine whether the composition of the reefs in NSG are natural or an artefact of human disturbance (Gaylard 2014).

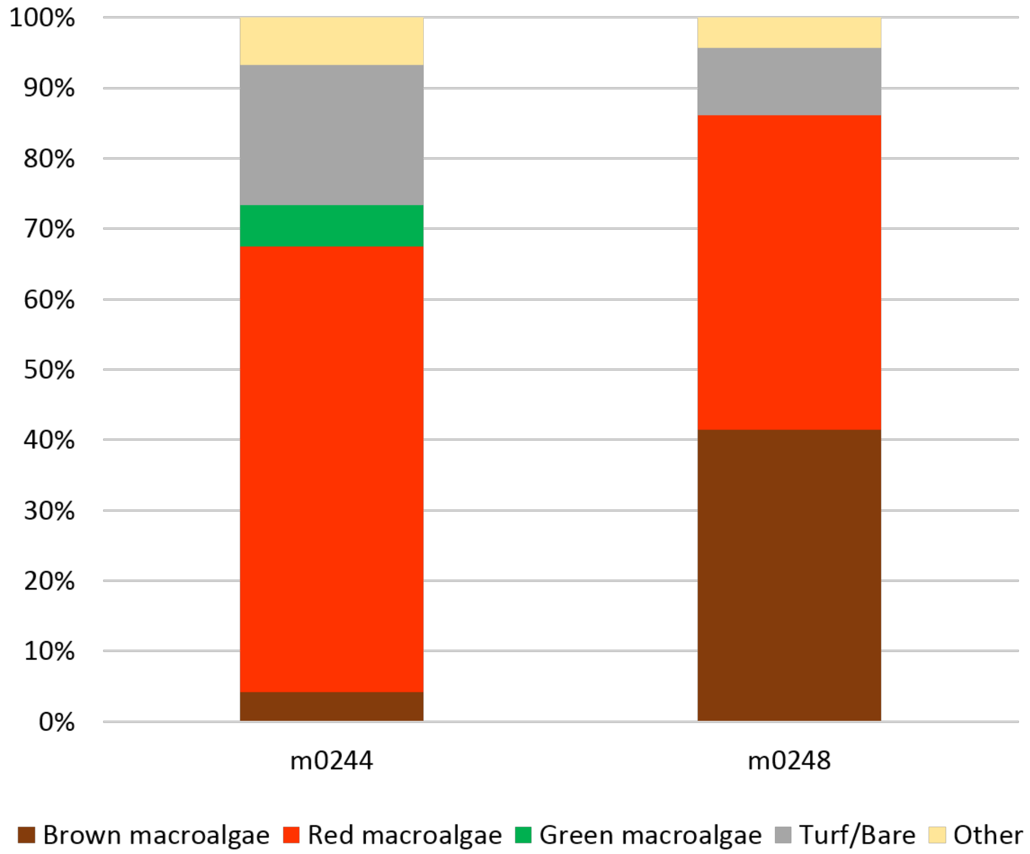


Figure 13 Spring average reef habitat composition at Stony Point (m0244 n = 9) and Point Lowly (m0248 n = 22)

5.4.1 Modifiers

The epiphyte load on seagrass in Yonga varied from no observable epiphytes at Airfield (m0203) through to high epiphyte load (> 70 out of 100) at Lucky Bay west (m0209), Mullaquana (m0253) and Eight Mile Creek Beach south (m0255) in Figure 12. Epiphytes were higher in spring than autumn for Yonga (Figure 6a) and opportunistic algal cover (eg *Ulva* sp.) followed the same seasonal pattern (Figure 6b). Sites that had no seagrass still showed substantial loads of opportunistic algae, indicating that nutrients were in excess eg Cockle Spit (m0232) in Figure 12.

Annual site habitat data, epiphyte load and opportunistic macroalgae were plotted using nMDS to allow visual assessment of patterns and hypothetical reference sites for comparison. Figure 14 shows a reasonably even spread of sites across the plot but there were observable patterns in the data. In general, sites closer to or down current from urban and industrial discharges from Whyalla (eg Whyalla north and south and False Bay south) and Port Pirie (eg Germein Bay inner, Germein south, Germein Bay south) were positioned towards the right of the plot, closer to the Very Poor hypothetical reference sites. Also within this cluster were sites that have been in the regions of extensive seagrass loss including Webling Bay and Wood Point north (Seddon 2000) or considered to be unsuitable for seagrass (eg Cockle Spit, Tickera Bay outer).

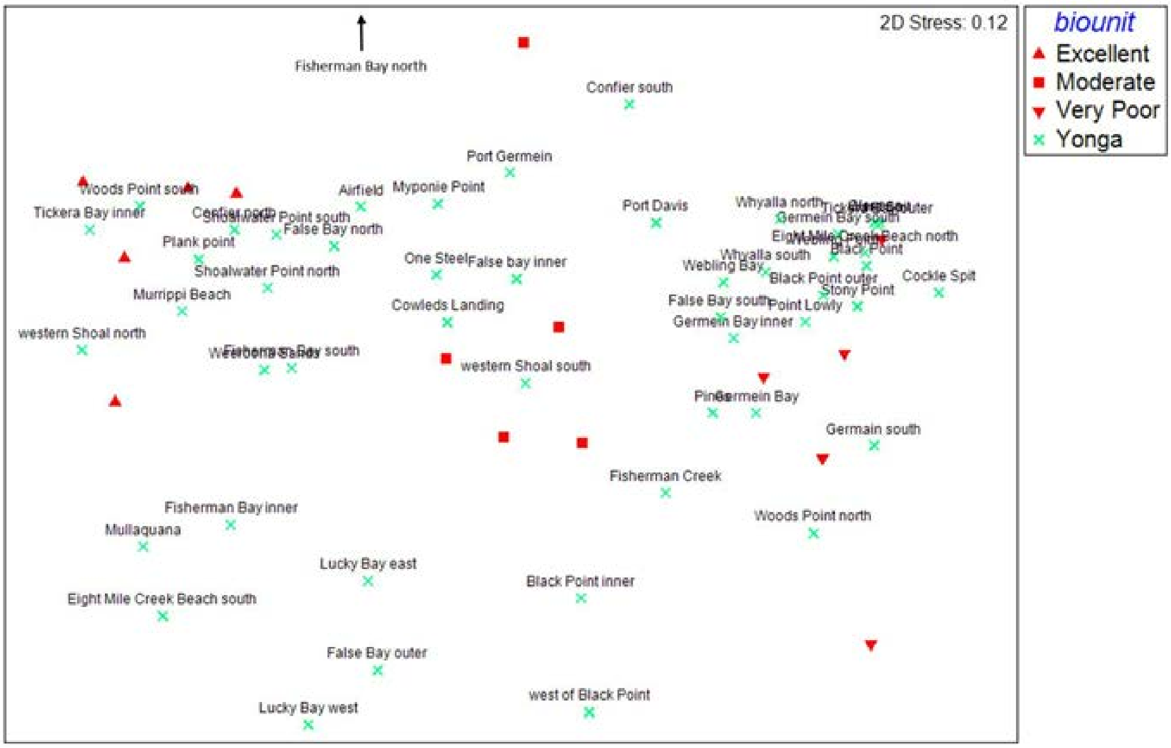


Figure 14 nMDS plot of site averaged seagrass density and cover, epiphyte load and opportunistic algae for sites in Yonga. Hypothetical reference sites included for comparison.

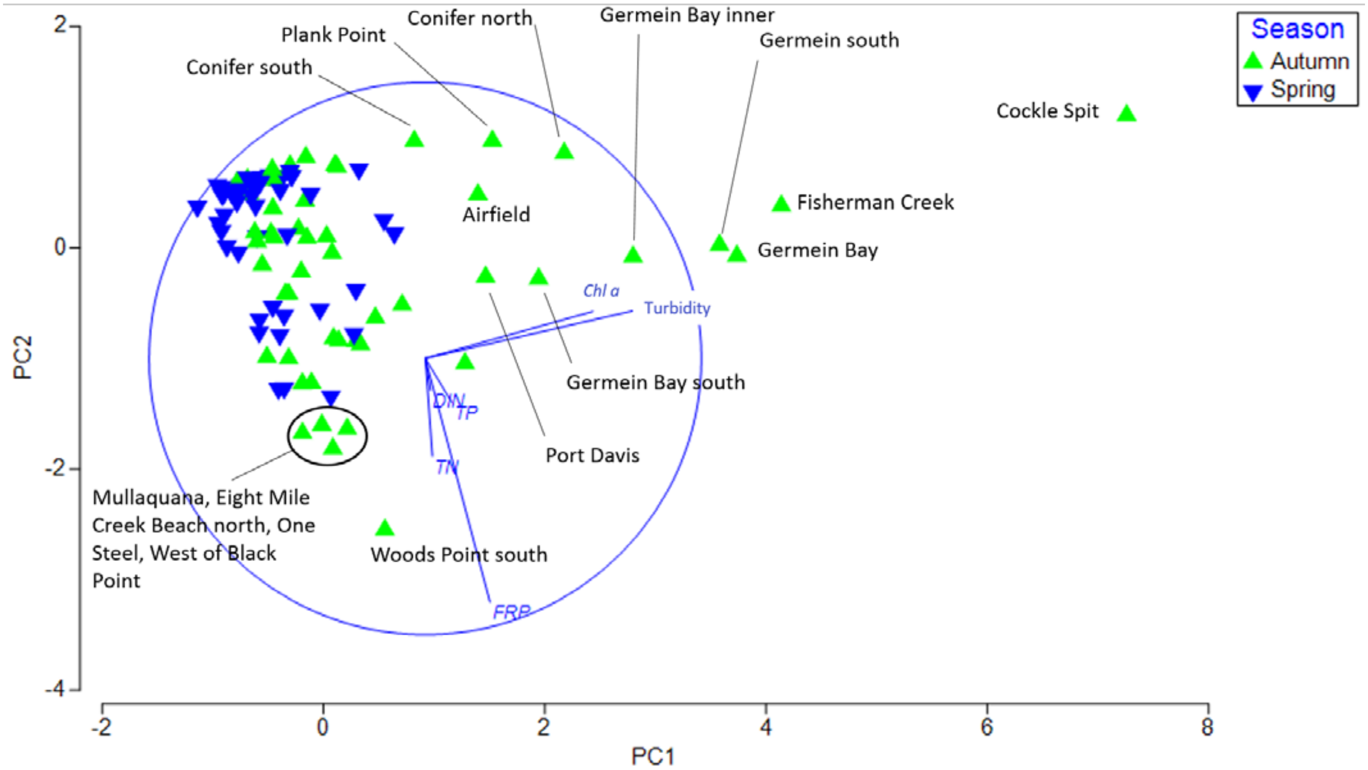


Figure 15 PCA of seasonal average water chemistry results for the Yonga biounit. The first two principal components (PC1 and PC2) account for 80.9% of the variability. Turbidity, Chl a = chlorophyll *a*, TN = total nitrogen, DIN = dissolved inorganic nitrogen, TP = total phosphorus, FRP = filtered reactive phosphorus

Water quality data suggests that nutrients have been taken up by algae or that high nutrient loads in the water column were not detectable due to the extremely sparse data collected (Table 6). Due to the sparse nature of water quality data, the data were pooled seasonally to increase the power of analysis. Principal component analysis (PCA) was performed for seasonal water quality data from the Yonga biounit to visually inspect patterns in the data (Figure 15).

The first two components account for 80.9% of the variability of the data, and indicates a good representation of the data. The main cloud of data comprise all sites in spring, and the majority of sites in autumn, and is characterised by low nutrients and turbidity (Figure 15). Extending away from the main cloud of points at the top of the plot are autumn sites from the southwest of Yonga: Conifer north (m0204) and south (m0205), Airfield (m0203) and Plank Point (m0201) that differ from their spring counterparts by having higher chlorophyll and turbidity. The result could be influenced by activities such as aquaculture further south in Spencer Gulf and high current speeds, possibly increasing turbidity.

Similarly, the sites: Cockle Spit (m0232), Fisherman Creek (m0238), Germein Bay (m0234), Germein south (m0233), Germein Bay inner (m0235) and south (m0239), and Port Davis (m0239) are clustered around Port Pirie, on the eastern side of Yonga. The chlorophyll may be contributing to high turbidity at these sites along with suspension of sediments that can often be observed after ships enter and leave Port Pirie through a narrow, relatively shallow passage. The chlorophyll results may also be driven by shallow waters of Germein Bay providing excellent growing conditions for phytoplankton (Figure 3).

Table 6 Annual median water chemistry and chlorophyll *a* values for Yonga biounit 2012. Bold values indicate results significantly higher than reference.

	Dissolved inorganic nitrogen (mg/L)	**Total nitrogen (mg/L)**	**Filtered reactive phosphorus (mg/L)**	**Total phosphorus (mg/L)**	**Turbidity (NTU)**	**Chlorophyll *a* (µg/L)**
Median	0.004	0.122	0.002	0.015	0.195	0.320
Standard deviation	0.003	0.188	0.002	0.010	0.271	0.295
N	300	300	300	300	300	195
Reference median	0.018	0.150	0.005	0.015	0.190	0.627
Mann-Whitney significance at p < 0.05	0.000	0.000	0.000	0.324	0.260	0.000

5.5 Conclusion

The aquatic ecosystem condition of Yonga was in Good condition and is consistent with the prediction of the Tier 1 assessment (section 2.3). Broadly, the southernmost sections of Yonga comprised dense meadows of long-lived *Posidonia* and *Amphibolis* seagrass species. Generally, sites located closer to urban centres (ie Whyalla and Port Pirie), where discharges are more prevalent, often had less seagrass cover, showed higher variability and in some locations were dominated by the colonising species *Heterozostera* sp. The presence of colonising species may indicate that seagrass could be returning to areas where it has previously been lost, particularly in areas where there has been improvement in discharge quality. Similarly, seagrass epiphytes and opportunistic algae were generally more abundant closer to urban centres, especially in sheltered waters of bays where water residence times are longer.

Historical impacts, such as seagrass dieback from extreme weather events (Seddon 2000), nutrient rich discharges (Irving 2014) or metal contamination within the region (Ward 1984, Gaylard 2014) may still be evident and possibly hindering seagrass regrowth. Slow rates of colonisation by many seagrasses, especially *Posidonia* spp and *Amphibolis* spp mean that loss is likely to be a long-term problem, even if current conditions are suitable for regrowth (Irving 2013).

Despite the long historical terrestrial use of Yonga, this is the first program to assess the condition of nearshore marine habitats between 2–15 m deep, and as such provides an important baseline. A substantial part of the nearshore MER program is investigating change in habitat condition or the level of stress over time. As such, subsequent monitoring periods will add information and allow for temporal conclusions on ecosystem condition.

5.6 Pressures and management actions

Prior to 2004, the WWTP at Whyalla discharged approximately 45 tonnes of nitrogen into a small tidal creek near Mullaquana (NPI 2012). In 2004, SA Water undertook an environmental improvement program which included construction of the Whyalla Reclamation Plant (WRP) capable of producing treated wastewater suitable for irrigation. This was completed in 2007 and recycled water is used for local council parks resulting in reduced nutrient loads from the WWTP to approximately 25 tonnes, which is likely to reduce any impacts to local marine ecosystems.

The Port Pirie WWTP discharged about 40 tonnes of nitrogen to Second Creek, a small tidal creek west of Port Pirie until in 2004, SA Water undertook an environmental improvement program. The program included an upgrade of the plant to a sequencing batch bioreactor reducing the load of nitrogen being discharged to approximately 15–25 tonnes (NPI 2012).

The discharge of ammonia from the steelworks at Whyalla has been shown to have significantly contributed to the loss of over 20 km^2 of seagrass throughout the False Bay area up until the early 1990s (Harbison & Wiltshire 1993, Irving 2014). The Whyalla Steelworks has monitored seagrass health and extent in False Bay periodically since 1990 and observed a gradual but sustained increase in seagrass extent adjacent to the works. Extension of the seawalls has resulted in higher retention time allowing greater biological assimilation resulting in lower dissolved nitrogen loads reaching the nearshore habitats. Additionally, process improvement and the construction of an engineered reed bed treatment system have contributed to the improvement (Gaylard 2014).

Stormwater runoff from urban catchments can discharge nutrients and sediments into nearshore waters contributing to the pressure on the nearshore environment. The Whyalla City Council has been undertaking stormwater upgrade works at key locations including improvement to stormwater drainage at the Whyalla foreshore. Stormwater improvement surrounding Ferry Street will direct stormwater into nearby wetland ponds enhancing the wetlands and reducing discharges to coastal waters.

5.7 Conclusions

This MER program has identified a range of nearshore habitats throughout the bioregion. Yonga had more extensive seagrass compared to Winninowie, which may be related to the geomorphology, Yonga has wide expanses of shallower water suitable for seagrass growth, while in contrast Winninowie is narrower with relatively deep channels where strong tidal currents can scour the seafloor, potentially preventing substantial seagrass habitat from developing (Easton 1978). Additionally, the higher natural salinity and water temperatures of Winninowie may be approaching the tolerances of these seagrasses species limiting their extent and increasing their susceptibility to other stressors (Wilson & Dunton 2017).

Habitat condition in Winninowie was variable, with some areas having largely intact seagrass meadows, while other locations were totally devoid of seagrass where it would normally be expected to grow. Seagrass epiphytes and opportunistic algae in Winninowie suggests the area is subject to localised nutrient enrichment, while strong tidal regimes in channels, heat and salinity stress and reduced flushing with open waters are also key factors in this region.

The southernmost sections of Yonga comprised dense meadows of long-lived *Posidonia* and *Amphibolis* seagrass. Sites located closer to urban centres, often have less seagrass higher epiphytic load and opportunistic algae, especially when in sheltered waters of bays where water residence times are longer.

This is the first broad-scale program to assess the condition of nearshore marine habitats throughout the entire Northern Spencer Gulf. The results of this program demonstrate that despite the unique natural conditions of this inverse estuary and the long industrial and agricultural use in the region, the habitat condition is still largely intact. Into the future, further monitoring will allow a more comprehensive evaluation of changes in condition over time which will enable management to be targeted to areas that are changing, and will also highlight areas in need of further investigation.

6 References

ABS 2012, *QuickStats: A simple at-a-glance summary of Census statistics for your selected area*, Commonwealth of Australia, viewed 12 December 2017, www.abs.gov.au/websitedbs/censushome.nsf/home/quickstats.

Ainslie RC, DA Johnstone and EW Offler 1994, 'Growth of the seagrass *Posidonia sinuosa* Cambridge and J Kou at locations near to, and remote from, a power stations thermal outfall in Northern Spencer Gulf, South Australia', *Transactions of the Royal Society of South Australia,* **118:** 197–206.

Anderson MJ, RN Gorley and KR Clarke 2008, *PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods*, PRIMER-E Ltd.

Anderson RJ 1988, *Solid Town: The history of Port Augusta,* Oxley, Australia.

Baker JL, SA Shepherd, N Barrett, and CFD Gurgle 2014, 'Subtidal macro-algae of Spencer Gulf', in Shepherd SA, SM Madigan, BM Gillanders, S Murray-Jones and DJ Wiltshire (eds), *Natural History of Spencer Gulf,* Adelaide, Royal Society of South Australia Inc, pp 136–152.

BHP 2009, *Olympic Dam expansion draft environmental impact statement: marine environment*, viewed 12 December 2017, www.bhpbilliton.com/home/society/regulatory/Documents/odxEisChapter16MarineEnvironment.pdf.

Bierman P, M Lewis, B Ostendorf and J Tanner 2011, 'A review of methods for analysing spatial and temporal patterns in coastal water quality', *Ecological Indicators,* **11:** 103–114.

Bureau of Meteorology 2012, *Climate data online,* viewed 12 December 2017, www.bom.gov.au/jsp/ncc/cdio/weatherData/av?p_nccObsCode=136&p_display_type=dailyDataFile&p_startYear=2012&p_c=-66419527&p_stn_num=018224.

Bureau of Meteorology 2013, *Tide predictions for Australia, South Pacific and Antarctica,* viewed 12 December 2017, www.bom.gov.au/oceanography/tides/.

Bureau of Meteorology 2014, *Average annual, seasonal and monthly rainfall,* viewed 2 August 2014, http://www.bom.gov.au/jsp/ncc/climate_averages/rainfall/index.jsp?period=an&area=sa.

Borowitzka MA, PS Lavery and M Keulen 2006, 'Epiphytes of seagrasses', in Larkum AWD, RJ Orth and C Duarte (eds), *Seagrasses: Biology, Ecology and Conservation***,** Springer Netherlands, pp 441–461.

Brown J, R Gowen and D Mclusky 1987, 'The effect of salmon farming on the benthos of a Scottish sea loch', *Journal of Experimental Marine Biology and Ecology,* **109:** 39–51.

Bryars S 2003, *An Inventory of Important Coastal Fisheries Habitats in South Australia*, Fish Habitat Program, Primary Industries and Resources South Australia, 909 pp.

Bryars S, G Collings and D Miller 2011, 'Nutrient exposure causes epiphytic changes and coincident declines in two temperate Australian seagrasses', *Marine Ecology Progress Series,* **441:** 89–103.

Bryars S and K Rowling 2008, 'Benthic habitats of eastern Gulf St Vincent: Major changes in seagrass distribution and composition since European settlement of Adelaide', in Bryars S (ed), *Restoration for coastal seagrass ecosytems: Amphibolis antartica in Gulf St Vincent, South Australia,* A report for the Natural Heritage Trust, PIRSA Marine Biosecurity, Department of Environment and Heritage and Environment Protection Authority, South Australian Research and Development Institute (Aquatic Sciences), Adelaide.

Butterfield M and S Gaylard 2005, *The heavy metal status of South Australian dolphins*, Environment Protection Authority,
23 pp.

Bye JAT and JJA Whitehead 1975, 'A theoretical model of the flow in the mouth of Spencer Gulf, South Australia', *Estuarine and Coastal Marine Science,* **3:** 477–481.

Chavez FP, PG Strutton, GE Friedrich, RA Feely, GC Feldman, DG Foley and MJ McPhaden 1999, 'Biological and chemical response of the equatorial Pacific Ocean to the 1997–98 El Niño', *Science,* **286:** 2126–2131.

Cheshire AC 1996, *Investigating the environmental effects of sea-cage tuna farming: methodology for investigating seafloor scouring*, Department of Botany, University of Adelaide, Adelaide, 43 pp.

Cheshire AC, G Westphalen, A Smart and S Clarke 1996, *Investigating the environmental effects of sea-cage tuna farming II. The effects of sea-cages*, Department of Botany, University of Adelaide, Adelaide, 72 pp.

Clarke K and R Warwick 2001, Changes in marine communities: An approach to statistical analysis and interpretation, PRIMER-E, Plymouth, 172 pp.

Clarke SM and H Kirkman 1989, 'Seagrass dynamics', in Larkum AWD, AJ McComb and SA Shepherd (eds), *Biology of the seagrasses: A treatise on the biology of seagrasses with special reference to the Australian region*, Elsevier, North Holland, Amsterdam, pp 304–345**.**

Collings G, D Miller, E O'Loughlin, A Cheshire and S Bryars 2006, *Turbidity and reduced light responses of the meadow forming seagrasses Amphibolis and Posidonia, from the Adelaide metropolitan coastline*, ACWS Technical report no 12, South Australian Research and Development Institute (Aquatic Sciences), Adelaide, 52 pp.

Commonwealth of Australia 2006, *A Guide to the Integrated Marine and Coastal Regionalisation of Australia version 4*, Department of the Environment and Heritage, Canberra, 16 pp.

Connell SDF, AD Irving, K Griffin, K Owen and BD Russell 2012, *Nutrients as drivers of seagrass loss: tests of threshold effects,* Prepared for SA Water, School of Earth and Environmental Science, University of Adelaide, 66 pp.

Corbin T and S Wade 2004, *Heavy metal concentrations in razorfish (Pinna bicolor) and sediments across Northern Spencer Gulf*, Environment Protection Authority, Adelaide, 21 pp.

Costanza R, R d'Arge, R de Groot, S Farber, M Grasso, B Hannon, K Limburg, S Naeem, RO Neill, J Paruelo, R Raskin, P Sutton and M van den Belt 1997, 'The value of the world's ecosystem services and nature capital', *Nature*, **387:** 253–260.

Darling ES and IM Côté 2008, 'Quantifying the evidence for ecological synergies', *Ecology Letters*, **11** (12): 1278–1286.

Davies CM, Altavilla N, Krogh M, Ferguson CM, Deree DA and NJ Ashbolt 2005, 'Environmental inactivation of *Cryptosporidium* oocysts in catchment soils', *Journal of Applied Microbiology,* **98:** 308–317.

Davies SP and SK Jackson 2006, 'The biological condition gradient: A descriptive model for interpreting change in aquatic ecosystems', *Ecological Applications*, **16(4):** 1251–1266.

Deeley DM and E Paling 1999, *Assessing the ecological health of estuaries in the southwest of Australia*, Land and Water Resources Research and Development Corporation, Canberra, 132 pp.

Delgado O, J Ruiz, M Perez, J Romero and E Ballesteros 1999, 'Effects of fish farming on seagrass *(Posidonia oceanica)* in a Mediterranean bay: seagrass decline after organic loading cessation', *Oceanologica Acta,* **22:** 109–117.

Department of Environment, Water and Natural Resources 2010, State Marine Benthic Habitats, DEWNR, viewed 12 December 2017, https://data.environment.sa.gov.au/NatureMaps/Pages/default.aspx.

Duarte CM 2002, 'The future of seagrass meadows', *Environmental Conservation,* **29:** 192–206.

ECOS 1983, 'How marine life copes with a lead smelter', *ECOS Magazine,* CSIRO Publishing.

Easton AK 1978, 'Reappraisal of the tides in Spencer Gulf, South Australia'. *Australian Journal of Marine and Freshwater Research*, **29:** 467–477.

Edyvane KS 1999a, *Conserving Marine Biodiversity in South Australia Part 2: Identification of areas of high conservation value in South Australia*, South Australian Research and Development Institute (Aquatic Sciences), Primary Industries and Resources SA, Adelaide, 328 pp.

Edyvane KS 1999b, *Conserving Marine Biodiversity in South Australia Part 1: Background, Status and Review of Approach to Marine Biodiversity Conservation in South Australia*, South Australian Research and Development Institute (Aquatic Sciences), Primary Industries and Resources SA, Adelaide, 182 pp.

Ellis JC and CF Gilbert 1980, *How to handle 'less-than' data when forming summaries*, Water Research Centre Enquiry Report ER 764, Water Research Centre, Medmenham, England.

Favier D, Scholz G, Vanlaarhoven J, Bradley J and L Phipps 2004, *A management plan for the Broughton Catchment, South Australia*, Department of Water, Land and Biodiversity Conservation, pp 2004–16.

Fletcher W, J Chesson, K Sainsbury, T Hundloe and M Fisher 2005, 'A flexible and practical framework for reporting on ecologically sustainable development for wild capture fisheries', *Fisheries Research,* **71:** 175–183.

Fourqurean JW, CM Duarte, H Kennedy, N Marbà, M Holmer, MA Mateo, ET Apostolaki, GA Kendrick, D Krause-Jensen and KJ McGlathery 2012, 'Seagrass ecosystems as a globally significant carbon stock', *Nature Geoscience,* **5:** 505–509.

Gaylard S 2014, 'Marine Pollution within Spencer Gulf', in Shepherd SA, SM Madigan, BM Gillanders, S Murray-Jones, and DJ Wiltshire (eds), *Natural History of Spencer Gulf,* Royal Society of South Australia Inc, Adelaide, pp 378–391.

Gaylard S, Nelson M and W Noble 2013a, *Nearshore Marine Aquatic Ecosystem Condition Reports: Lower Spencer Gulf Assessment Report 2010*, Environment Protection Authority, 95 pp.

Gaylard S, M Nelson and W Noble, 2013b, *The South Australian monitoring, reporting and evaluation program for aquatic ecosystems: Rationale and methods for the assessment of nearshore marine waters*, Environment Protection Authority, 70 pp.

Gaylard S, S Thomas and M Nelson 2011, 'An Assessment of the Current Status of Bioavailable Metal Contamination Across South Australia Using Translocated Mussels *Mytilus galloprovincialis*', *Transactions of the Royal Society of South Australia,* **135:** 39–45.

Gillanders BM and NL Payne 2014, 'Giant Australian Cuttlefish', in Shepherd SA, SM Madigan, BM Gillanders, S Murray-Jones, and DJ Wiltshire (eds), *Natural History of Spencer Gulf*, Royal Society of South Australia Inc, Adelaide, pp 288–301.

Gostin V and SM Hill 2014, 'Spencer Gulf: Geological setting and evolution', in Shepherd SA, SM Madigan, BM Gillanders,
S Murray-Jones, and DJ Wiltshire (eds), *Natural History of Spencer Gulf*, Royal Society of South Australia Inc, Adelaide, pp 21–35.

Guilford SJ and RE Hecky 2000, 'Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: Is there a common relationship?', *Limnology and Oceanography,* **45:** 1213–1223.

Harbison P 1984, 'Regional variation in the distribution of trace metals in modern intertidal sediments of Northern Spencer Gulf, South Australia', *Marine Geology,* **61:** 221–247.

Harbison P and D Wiltshire 1993, *BHP Marine Environment Studies 1992*, Environmental Consulting Australia.

Harbison P 1986, 'Mangrove muds—a sink and a source for trace metals', *Marine Pollution Bulletin,* **17:** 246–250.

Helsel DR 1990, 'Less than obvious; statistical treatment of data below the detection limit', *Environmental Science & Technology,* **24:** 1766–74.

Helsel DR 1987, 'Advantages of nonparametric procedures for analysis of water quality data', *Hydrological Sciences Journal,* **32:** 179–190.

Helsel DR and RM Hirsch 2002, *Techniques of Water-Resources Investigations of the United States Geological Survey: Hydrologic Analysis and Interpretation*, US Geological Survey, Virginia USA.

Holmer M, M Argyrou, T Dalsgaard, R Danovaro, E Diaz-Almela, CM Duarte, M Frederiksen, A Grau, I Karakassis, N Marbà, S Mirto, M Pérez, A Pusceddu and M Tsapakis 2008, 'Effects of fish farm waste on *Posidonia oceanica* meadows: Synthesis and provision of monitoring and management tools', *Marine Pollution Bulletin,* **56:** 1618–1629.

Irving AD 2014, 'Seagrasses of Spencer Gulf', in, SA Shepherd, SM Madigan, BM Gillanders, S Murray-Jones, and DJ Wiltshire (eds), *Natural History of Spencer Gulf*, Royal Society of South Australia Inc, Adelaide, pp 121–135.

Irving AD 2013, 'A century of failure for habitat recovery', *Ecography,* **36(4)**: 414–416.

Irving AD, JE Tanner and S Gaylard 2013, 'An integrative method for the evaluation, monitoring, and comparison of seagrass habitat structure', *Marine Pollution Bulletin,* **66:** 176–184.

Islam MS 2005, 'Nitrogen and phosphorus budget in coastal and marine cage aquaculture and impacts of effluent loading on ecosystem: review and analysis towards model development', *Marine Pollution Bulletin,* **50:** 48–61.

Jones G, J Baker, K Edyvane and G Wright 1996, 'Nearshore fish community of the Port River-Barker Inlet Estuary, South Australia. I. Effect of thermal effluent on the fish community structure, and distribution and growth of economically important fish species', *Marine and freshwater Research,* **47:** 785–799.

Kämpf J 2014, 'System models for Spencer Gulf', in Shepherd SA, SM Madigan, BM Gillanders, S Murray-Jones, and DJ Wiltshire (eds), *Natural History of Spencer Gulf,* Royal Society of South Australia Inc, Adelaide, pp 69–78.

Larkum AWD, EA Drew and PJ Ralph 2006, 'Photosynthesis and metabolism in seagrasses at the cellular level', in Larkum AWD, RJ Orth, and CM Duarte (eds), *Seagrasses: Biology, ecology and conservation,* Springer, Dordrecht, pp 324–345.

Lauer PR, M Fernandes, PG Fairweather, J Tanner and A Cheshire 2009, 'Benthic fluxes of nitrogen and phosphorus at southern bluefin tuna *Thunnus maccoyii* sea-cages', *Marine Ecology Progress Series,* **390:** 251–263.

Leujak W and RFG Ormond 2007, 'Comparative accuracy and efficiency of six coral community survey methods', *Journal of Experimental Marine Biology and Ecology,* **351:** 168–187.

Martinez-Porchas M and LR Martinez-Cordova 2012, 'World aquaculture: environmental impacts and troubleshooting alternatives', *Scientific World Journal,* 389623.

McArthur LC and JW Boland 2006, 'The economic contribution of seagrass to secondary production in South Australia', *Ecological Modelling,* **196:** 163–172.

McDowell LM and P Pfennig 2013, *Adelaide Coastal Water Quality Improvement Plan (ACWQIP) – A plan that covers the issues, challenges and a way forward for water quality improvement for Adelaide's coastal waters*, Environment Protection Authority, Adelaide, 162 pp.

Miller D, ARM Wright and PG Fairweather 2014, 'Benthic habitat mapping in Spencer Gulf', in, Shepherd SA, SM Madigan, BM Gillanders, S Murray-Jones, and DJ Wiltshire (eds), *Natural History of Spencer Gulf*, Royal Society of South Australia Inc, Adelaide, 80–91 pp.

Naylor RL, RJ Goldburg, JH Primavera, N Kautsky, MCM Beverage, J Clay, C Folke, J Lubchenco, H Mooney and M Troell 2000, 'Effects of aquaculture on world fish supplies', *Nature,* **405:** 1017–1024.

Nelson M, S Gaylard and W Noble 2013, *Nearshore Marine Aquatic Ecosystem Condition Reports: Gulf St Vincent Assessment Report 2010 and 2011*, Environment Protection Authority, Adelaide.

Neverauskaus VP 1989, *Impact of Land Based Discharges on Seagrass Beds Near Adelaide South Australia*, 9th Australasian Conference on Coastal and Ocean Engineering, Institution of Engineers, Adelaide.

National Pollutant Inventory 2012, *2011/2012 report for South Australia Water Corporation Port Augusta East Waste Water Treatment Plant*, Viewed 12 December 2017, www.npi.gov.au/npidata/action/load/facility-source-result;jsessionid=E40ED9956C3864DFB7C01C5C259C587A/criteria/destination/ALL/source-type/ALL/substance-name/All/subthreshold-data/Yes/year/2013?pageIndex=35&sort=regBusinessName&dir=asc&pageSize=100

Nunes Vaz RA and GW Lennon 1986, 'Physical Property Distributions and Seasonal Trends in Spencer Gulf, South Australia: an inverse estuary', *Australian Journal of Marine & Freshwater Research,* **37:** 39-53.

Nunes Vaz RA 2014, 'Physical characteristics of Spencer Gulf'. in, Shepherd SA, SM Madigan, BM Gillanders, S Murray-Jones, and DJ Wiltshire (eds), *Natural History of Spencer Gulf*, Royal Society of South Australia Inc, Adelaide, pp 44–68.

Nunes Vaz RA, GW Lennon and DG Bowers 1990, 'Physical behaviour of a large, negative or inverse estuary', *Continental Shelf Research,* **10:** 277–304.

OneSteel 2014, *Onesteel Whyalla steelworks,* viewed 12 December 2017, www.libertyonesteel.com/about-us/our-businesses/whyalla-steelworks/

Paravicini V, C Morri, G Cirbilli, M Montefalcone, G Albertellis and CN Bianchi 2009, 'Size matters more than method: Visual quadrats *vs* photography in measuring human impact on Mediterranean rocky reef communities', *Estuarine, Coastal and Shelf Science,* **81:** 359–367.

Parsons-Brinkerhoff and SARDI 2003, *Technical review for aquaculture management plans, Phase 2 – Volume A Upper Spencer Gulf*, Adelaide.

Patil GP 1995, ‘Editorial: composite sampling’, *Environmental and Ecological Statistics,* **2:** 169–179.

Ralph PJ 1998, ‘Photosynthetic response of laboratory-cultured *Halophila ovalis* to thermal stress’, *Marine Ecology Progress Series,* **171:** 123–130.

Reay WG 2004, ‘Septic tank impacts on ground water quality and nearshore sediment flux’, *Ground Water,* **42:** 1079–1089.

RESIC 2013, *Working toward integrated, cost competitive infrastructure that meets the future requirements of the resources and energy sectors. A report on the achievements and deliverables of RESIC – a partnership between government and industry providing advice to the South Australian Government on the resource sector’s infrastructure requirements*, Department for Manufacturing, Innovation, Trade, Resources and Energy, Adelaide

Rountos KJ, BJ Peterson and I Karakassis 2012, ‘Indirect effects of fish cage aquaculture on shallow *Posidonia oceania* seagrass patches in coastal Greek waters’, *Aquaculture Environment Interactions,* **2:** 105–115.

Ruiz JM, C Marco-Méndez and JL Sánchez-Lizaso 2010, ‘Remote influence of off-shore fish farm waste on Mediterranean seagrass (*Posidonia oceanica*) meadows’, *Marine and Environmental Research,* **69:** 118–26.

Seddon S, RM Connolly and KS Edyvane 2000, ‘Large-scale seagrass dieback in Northern Spencer Gulf, South Australia’, *Aquatic Botany,* **66:** 297–310.

Shepherd SA 1983, ‘Benthic communities of upper Spencer Gulf, South Australia’, *Transactions of the Royal Society of South Australia,* **107:** 69–85.

Shepherd SA and RC Sprigg 1976, ‘Substrate, Sediments and Subtidal Ecology of Gulf of St Vincent and Investigator Strait’, in Twidale CR, MJ Tyler and BPE Webb (eds), *Natural History of the Adelaide Region*, Royal Society of South Australia, Adelaide, pp 161–174.

Smith VH 2006, ‘Responses of estuarine and coastal marine phytoplankton to nitrogen and phosphorus enrichment’, *Limnology and Oceanography,* **51:** 377–384.

Steer MA 2015, *Surveying, searching and promoting giant Australian cuttlefish spawning activity in Northern Spencer Gulf*, South Australian Research and Development Institute (Aquatic Sciences), Adelaide.

Steer MA, S Gaylard and M Loo 2013, *Monitoring the relative abundance and biomass of South Australia's Giant Cuttlefish breeding population, Final report for the Fisheries Research and Development Corporation*, South Australian Research and Development Institute (Aquatic Sciences), Adelaide.

Tanner JE and M Fernandes 2010, ‘Environmental effects of Yellowtail Kingfish aquaculture in South Australia’, *Aquaculture Environment Interactions,* **1:** 155–165.

Thomas IM, RC Ainslie, DA Johnston, EW Offler, and PA Zed 1986, ‘The effects of cooling water discharge on the intertidal fauna in the Port River estuary, South Australia’, *Transactions of the Royal Society of South Australia,* **109:** 159–172.

Turner DJ and AC Cheshire 2002, *Effect of Dispersed Sediment Plumes from Beach Sand Replenishment Dredging on Recruitment of Phaeophycean Algae to Rocky Reefs in Gulf St Vincent, South Australia: Final Report: Incorporating Surveys from 1998–2001*, Department of Environmental Biology, University of Adelaide, 52 pp.

Turner DJ, TN Kildea and G Westphalen 2007, *Examining the health of subtidal reef environments in South Australia Part 2: Status of selected South Australian reefs based on the results of the 2005 surveys*, South Australian Research and Development Institute (Aquatic Sciences), Adelaide, 97 pp.

Ward TJ 1987, ‘Temporal variation of metals in the seagrass *Posidonia australis* and its potential as a sentinel accumulator near a lead smelter’, *Marine Biology,* **95:** 315–321.

Ward, TJ and PA Hutchings, 1996, 'Effects of trace metals on in faunal species composition in polluted intertidal and subtidal marine sediments near a lead smelter, Spencer Gulf, South Australia', *Marine Ecology Progress Series*, **135**: 123–135.

Ward TJ, RL Correll and RB Anderson 1986, ‘Distribution of cadmium, lead and zinc amongst the marine sediments, seagrasses and fauna, and the selection of sentinel accumulators, near a lead smelter in South Australia’, *Australian Journal of Marine & Freshwater Research,* **37(5),** 567–585.

Waycott M, CM Duarte, TJ Carruthers, RJ Orth, WC Denninson, S Olyarnik, A Calladine, JW Fourqurean, KL Heck, AR Hughes, GA Kendrick, WJ Kenworthy, FT Short and SL Williams 2009, *Accelerating loss of seagrasses across the globe threatens coastal ecosystems*, Proceedings of the National Academy of Sciences of the United States of America*,* **106:** 12377–81.

Wilson SS and KH Dunton 2017, 'Hypersalinity during regional drought drives mass mortality of the seagrass Syringodium filiforme in a subtropical lagoon', *Estuaries and Coasts,* https://doi.org/10.1007/s12237-017-0319-x.

Wiltshire KH and J Tanner 2010, *Assessment of potential impacts of Alinta Energy discharges into Hospital Creek, upper Spencer Gulf, South Australia*, South Australian Research and Development Institute (Aquatic Sciences), Adelaide.

7 Glossary and abbreviations

AECR	aquatic ecosystem condition reports
AHD	Australian height datum
CWMS	community wastewater management system
DEWNR	Department of Environment, Water and Natural Resources
DIN	dissolved inorganic nitrogen
EMS	environmental management system
EPA	South Australian Environment Protection Authority
FRP	filtered reactive phosphorus
ICRA	integrated marine and coastal regionalisation of Australia
PIRSA	Department of Primary Industries and Resources South Australia
LOR	limit of reporting
MER	monitoring, evaluation and reporting
nMDS	non-metric multi-dimensional scaling
NSG	Northern Spencer Gulf
PCA	principal components analysis
PIRSA	Primary Industries and Regions, South Australia
SARDI	South Australian Research and Development institute
SIMPROF	Similarity Profile Analysis
WWRP	wastewater reclamation plant
WWTP	wastewater treatment plant
TN	total nitrogen
TP	total phosphorus
YTK	Yellowtail Kingfish

Appendix 1 Methods

This section provides a more detailed explanation of the methods used in the NSG MER program. Further details can be found in Gaylard *et al* (2013a).

Site selection

The number of sites sampled within each biounit was initially based on the Tier 1 assessment taking into account the number and type of threats and the area of the biounit with a higher number of sites to address higher variability in larger or more disturbed biounits. However, this number was also limited by available resources and logistics. Sites were allocated using a stratified random design, which overlaid a numbered 500 m x 500 m grid over the waters less than 15 m deep throughout the biounit. A random number generator was used to select grid locations typically allocating double the number of required sites. Final site positions were selected from the random positions by excluding positions in close proximity to hazards such as breaking or potentially dangerous waters, known industrial discharges to ensure an ambient perspective was maintained and clumping of sites were also reduced. Final site positions are included in the attached maps for each biounit and in Appendix 4.

Methods at each site

Benthic habitats within each site were characterised by recording 10 x 50 m underwater video belt transects from random start points within the site (Figure 6). Video transects were recorded using a geo-referenced camera (Scielex/Kongsberg) angled at 90° to the seafloor. A live video feed ran directly from the camera to an audio and video encoding system (Geostamp) which overlaid a GPS location, direction, speed, date and time strings to the video and recorded to a hard drive. A surface screen allowed the operator to position the camera approximately 1 m from the benthos in order to maximise image quality and resolution. This set up provided a field of view of approximately 1 m^2, such that each belt transect equates to approximately 50 m^2. A full high definition (HD) video camera (GoPro Hero 2) was attached to the analogue video housing and the two units synchronised. This footage was used to provide confirmation on taxonomic identification where possible.

Quantifying water chemistry was undertaken by sampling two replicate 2.5-L water samples at each transect into a pre-rinsed 25-L container. After three transects the water in the container was mixed thoroughly, sub-sampled and then discarded. This process is repeated across the site for all 10 transects to provide snapshots (n = 3 per site) of water nutrient concentration (total nitrogen, total ammonia, total kjeldahl nitrogen, total oxidised nitrogen, total phosphorus and filtered reactive phosphorus) and turbidity. Samples for soluble nutrients were immediately filtered using a 0.45-µm filter and frozen as soon as practical prior to analysis. At each site, 2 x 1-L samples were taken from the water column for chlorophyll analysis, and immediately iced and placed in darkness. The samples were filtered using a 0.45-µm filter at the end of each day and the filter paper frozen prior to analysis. All samples were sent to the Australian Water Quality Centre for analysis within the laboratory holding times.

Compositing water samples in the above manner is commonly used to reduce analytical costs of environmental sampling, and with careful planning may reveal the same information as analysing many samples while still retaining, if not increasing, the precision of sample-based interferences (Patil 1995). A full description of the method for compositing water samples obtained under the nearshore MER program is described in Gaylard *et al* (2013a).

In the event of water chemistry analyses being below the detection limit (LOR), ie below the detection limit of the analytical equipment), a method of substituting the censored value with half the reporting limit has been adopted. For example the total ammonia LOR is 0.005 mg/L; if a result is recorded as < 0.005 mg/L then the result used is 0.0025 mg/L (Ellis & Gilbert 1980). This arbitrary approach has limitations (Helsel 1990) but in the nearshore MER program it was considered appropriate due to the amount of data generated, and the unbiased nature of using half the reporting limit compared to other methods that substitute for the reporting limit or allocation to a value of zero (Helsel 1990).

A multi-parameter sonde (YSI 6920 v2) was used to log physical water quality parameters including electrical conductivity, pH, dissolved oxygen and chlorophyll *a* at 10-second intervals for approximately 2.5 mins at 0.5 m from the surface at each location (n = ~15 per transect and ~150 per site).

Conceptual models and condition gradient

As well as detailing the methods undertaken to broadly assess ecological condition for the nearshore MER program, Gaylard *et al* (2013a) describes the development of generic conceptual models that have been used to suggest processes of degradation based on established literature and a condition gradient (Appendices 2 and 3). Conceptual models that describe the response of an ecosystem to stress have been used in developing strategies for natural resource management that put emphasis on the maintenance of important ecological characteristics and by extension system processes/services. The condition gradient is a type of conceptual model that relates an observed ecological response to increasing levels of human disturbance (Davies *et al* 2005). This gradient assumes that habitat condition deteriorates as the degree of human disturbance in the surrounding and adjacent environment increases and conversely, the best condition occurs where there is little to no human disturbance (Appendix 3).

The conceptual models use existing knowledge linked to data collection in the development of this program (Gaylard *et al* 2013a). They establish a biological condition gradient in response to nutrient enrichment and reduction in water clarity for seagrass, rocky reef and unvegetated sediment habitats in shallow (2–15 m) nearshore waters in South Australia. A description of these models is provided in Appendix 2 and their development is described by Gaylard *et al* (2013a). Results of the Tier 2 assessment in this program are compared to these conceptual models to describe condition.

As our understanding of the nearshore marine environment increases these conceptual models will be refined.

Data analysis

The biological and water chemistry data were analysed against a number of questions. Each question was assessed by using either univariate statistics to test whether a population is different to another (eg reference population), or by using multivariate statistics, which assess a combination of many parameters to compare the similarity (or dissimilarity) of populations to each other, or a combination of both test types. The data can be considered at a number of different spatial scales including the entire Northern Spencer Gulf bioregion. At the bioregion scale, results are considered at a very high level and considering broad-scale gradients. Key outcomes at the bioregion scale include:

- Has data acquisition been representative of the known broad habitat types?
- How do various metrics change across the bioregion – are there large-scale biological gradients present?
- What are the major determinants in any differences between biounits within the bioregion?

Biounits are a smaller spatial unit than bioregion and typically extend between 10–100 km. Biounits are the unit of assessment used for the AECR score, and allow finer spatial assessment with respect to the location of known pollution sources and smaller scale perturbations. They also relate closer with scales of management.

The fundamental outcomes of the biounit level assessment include:

- Has data acquisition been representative of the known broad habitat types?
- How does the biounit compare to the reference condition?
- Are there any biological gradients within the biounit?
- Is there any relationship between habitat scores and physical water quality observations within the biounit?
- How reliable are the indicators to show differences?

Multivariate statistics were used to explore similarities within the biological data, and used to infer biological gradients within a biounit, or within the bioregion to show how similar in multivariate space each level of data (site, biounit, bioregion) were to the reference condition or each other. Biological data was normalised and analysed using Bray-Curtis dissimilarity and then displayed using non-metric multidimensional scaling (nMDS) ordination plots (Clarke & Warwick 2001) to assess trends. Differences between specific variables (eg seagrass cover, macroalgae or seagrass epiphyte load) at a number of levels including between seasons, biounits and the reference condition were tested. A resemblance matrix was created using Bray–Curtis similarity for the data and then analysed using univariate permutational analysis for nonparametric data (Primer v6 + PERMANOVA, Anderson *et al* 2008).

Similarity profile (SIMPROF) test was undertaken on the bioregion data to explore whether there were any significant groups in the multivariate biological data at a 5% significance level (Clarke *et al* 2008). Principal components analysis (PCA) was undertaken on water chemistry data after the Euclidian distance was normalised and transformed using either the square root or the log (x+1) transformation (Clarke & Warwick 2001). All multivariate statistics were undertaken using Primer v6.0.

The environmental data (ie water chemistry) was highly skewed, so the non-parametric Mann Whitney U test was used to test for equality of two test population medians (Helsel 1987, Helsel & Hirsch 2002). All Mann-Whitney U tests were undertaken using Minitab 14 with $\alpha = 0.05$. Multivariate patterns in the water chemistry were investigated using principal component analysis (PCA) after the Euclidean distance was normalised and transformed (Clarke & Warwick 2001).

Statement of limitations

Gaylard *et al* (2013a) details the rationale and methods used in the nearshore MER program to define and assess ecological condition. It is again stressed that the ecological condition rating developed for this monitoring program is designed to be a broad overview using rapid assessment techniques, which are reviewed by a panel of experts to ensure consistency with the conceptual models and with the current level of understanding. The MER program is thus designed to be iterative and may change as increased understanding of disturbance gradients within seagrass, temperate reef and sandy bottom systems in South Australia is developed.

The broad regional focus of the Tier 1 and Tier 2 aspects of the MER program are not designed to provide scientific certainty or causal relationships between specific potential pollution sources and observed environmental degradation. Rather, site-specific uncertainty and causal relationships may be further investigated by specific projects under Tier 3 (section 1.1).

A key overlying premise is that the program assumes clear gradients and the vast diversity of southern Australian marine systems, coupled with any number of response gradients makes categorisation of habitats based on broad index scores difficult. Blind adherence to index results is not encouraged if other supporting evidence suggests otherwise. All technical reports are peer reviewed by experienced independent South Australian marine scientists as to whether the results align with our understanding of marine systems as well as common sense. If discrepancies are highlighted then further work will be needed to determine if the conceptual models for that biounit need revision, or our understanding on the marine environment is accurate.

It is accepted that even though there are challenges with the concept of ecological ‘health or condition’ and ‘report cards’ due to the over-simplification of inherently complex multi-dimensional systems, the potential benefits arising from the increased accessibility of the biological information to the wider community makes it a useful approach. Diagnosis of Poor condition should raise community and political concern, and result in action to manage the relevant issues (Deeley & Paling 1999). It should also be noted that an assessment of Poor condition does not necessarily mean that a particular location is degraded due to anthropogenic activity and where possible this will be conveyed through the AECR format.

Appendix 2 Conceptual models – Northern Spencer Gulf

Gaylard *et al* (2013a) detailed the monitoring methods undertaken to broadly assess ecological condition for the development of the AECR. This report detailed generic conceptual models that have been used to suggest processes of degradation based on established literature and develop a condition gradient (Appendix 3). The overarching assumption is that habitat condition is directly correlated to ecological condition, which holds true for many other temperate locations around the world (Duarte 2002, Waycott *et al* 2009). Conceptual models that describe the response of an ecosystem to stress have been used in developing strategies for natural resource management emphasising on the maintenance of important ecological characteristics. The condition gradient is a type of conceptual model that relates an observed ecological response to increasing levels of human disturbance (Davies & Jackson 2006). This gradient assumes that habitat condition deteriorates as the degree of human disturbance in the surrounding and adjacent environment increases, and conversely, the best condition occurs where there is little to no human disturbance.

This section addresses whether the major habitats in the Northern Spencer Gulf allow the use of the generic conceptual models provided in Gaylard *et al* (2013a) in order to ensure that the assumptions of those models are still reasonably applicable for this bioregion.

Seagrass

The ecological information summarised for each biounit in NSG (Section 2) indicates that the bioregion is dominated by seagrass meadows comprising mainly *Amphibolis antarctica* and/or species from the *Posidonia australis* group (*P sinuosa*, *P australis* & *P angustifolia*) with little to no recorded occurrences of *P ostenfeldii* group. This suggests that the conceptual model of seagrass degradation along a gradient of decreasing light outlined in Gaylard *et al* (2013a) would be applicable in this bioregion.

A fundamental aspect of this MER program is the assessment of whether the benthic habitats have changed over time. Humans have altered the landscape around NSG for over 150 years which is likely to have influenced, at least in some part, the nearshore benthic habitats. Without knowledge of historical (pre-1970s) benthic habitat composition and extent, it can be hard to determine whether habitats including seagrass may have existed in an area, or have been lost due to disturbance, or whether bare sand is a natural state.

In order to establish whether seagrass may have been present naturally at a site, this program has used methods adapted from Bryars and Rowling (2008), who established a coarse habitat reconstruction for each sampling area to determine if seagrass has been present prior to monitoring. If seagrass is currently present in the sampling area then it is likely that it was historically (pre-European settlement) present. As used by Bryars and Rowling (2008), this assumption is robust in gulfs and sheltered bays due to the slow growing and colonising ability of the dominant *Amphibolis* and *Posidonia* genera. In areas where seagrass is absent and there is no historical evidence of seagrass presence, an assessment will be made based on a range of known variables to attempt to determine whether seagrass was likely to have been present historically to determine the likelihood of change over time due to current activities (Gaylard *et al* 2013a).

Seagrass meadows in the sheltered waters of NSG tend to be extensive and continuous (Edyvane 1999b) and within the depth range of the AECR MER, comprised *P australis* complex and/or *Amphibolis* spp genera (Irving *et al* 2013, Irving 2014). Seagrass habitat is quantified for areas that are considered to be suitable for seagrass growth. Seagrass genus, density and area at a site are used to describe the habitat and compare it to a conceptual condition gradient (Table 7). Further detail of the conceptual seagrass gradient can be found in Gaylard *et al* (2013a).

Table 7 Conceptual seagrass condition gradient

Component	Excellent	Very Good	Good	Fair	Poor	Very Poor
Seagrass *Posidonia australis* complex seagrass cover	Dense meadows of seagrass typically *Posidonia australis* group and/or *Amphibolis* spp	Generally dense seagrass meadows with areas of moderate density not making up the majority of the seagrass area	Moderate density and decreasing seagrass area due to increasing bare sand patches or bare sand patch size	Moderate density of seagrass with frequent bare sand patches or uniform sparse seagrass coverage	Seagrass would typically be sparse and patchy	Seagrass only remains in isolated small patches where it had previously existed
Seagrass condition using the AECR MER program	>80% cover of dense meadows of seagrass typically *Posidonia australis* group and/or *Amphibolis* spp	70–79% cover of generally dense seagrass meadows. Seagrass density may start to become more variable	60–69% cover of moderate density seagrass. Bare sand patches increasing in number or area	40–59% cover of moderate density of seagrass with frequent bare sand patches or uniform sparse seagrass coverage	20–39% cover of sparse and patchy seagrass	<20% cover of sparse and patchy seagrass

Rocky reefs in Northern Spencer Gulf

With respect to rocky reef habitats, Baker *et al* (2014) outlines the key species that dominate both shallow and deeper reefs in the NSG bioregion. Macroalgal species and cover are influenced by a range of factors including oceanography, lower wave energy, extreme tidal ranges, high salinities and large seasonal variation in sea surface temperatures which have led to lower species richness in NSG than in the lower Spencer Gulf (Baker *et al* 2014).

However, the canopy species are largely consistent with the broad premise that, particularly in waters less than 15 m deep, large canopy forming brown algae dominate reefs throughout southern Australia (Shepherd & Sprigg 1976, Turner *et al* 2007). Rocky reef systems are complex multi-layered communities and a rapid assessment based on remote observations cannot fully capture the level of complexity compared to measurements obtained using SCUBA.

The use of photo quadrats and video has become increasingly popular to assess reef canopy assemblages (Leujak & Ormond 2007, Paravicini *et al* 2009), and for the broad-scale assessment used in this MER program is deemed sufficient. Condition of rocky reefs was described using the conceptual rocky reef condition gradient in Table 8 and described in detail including the numerous assumptions and limitations within (Gaylard *et al* 2013a), based on the Turner *et al* (2007) reef status indices.

Table 8 Conceptual rocky reef condition gradient

Component	Excellent	Very Good	Good	Fair	Poor	Very Poor
Robust brown macroalgal cover (*Ecklonia Sargassum*, *Cystophora* and *Scaberia*)	Reef dominated by members of Alariaceae, Cystoseiraceae and Sargassaceae families with areal cover of robust brown macroalgae > 40%			Cover of robust brown macroalgae below 40%		
Turfing algae	Areal cover of turfing algae < 25%			Areal cover of turfing algae > 25%		
Bare substrate	Bare rock substrate on reef < 20%			Bare rock substrate > 20%		

Unvegetated sediments

Unvegetated sediments dominate areas that are unsuitable for seagrass or macroalgal growth for a range of factors including water current speeds/wave energy, light availability (depth) and/or unsuitable/unstable substrate. Our knowledge of ecological processes and responses of bare sandy substrate to disturbance in southern Australia is limited, and often based on very costly sampling and analysis of infauna. Establishing a condition gradient is difficult when using remote video assessment methods. Brown *et al* (1987) and Cheshire (1996) have described a range of epi-faunal indicators of organic enrichment radiating from sea cage tuna farms, which suggest a shift in species composition towards an increase in deposit feeding organisms compared to control locations. These species may be amenable to video monitoring techniques depending on image quality (Cheshire 1996). The condition of unvegetated sediments will be described using the simplistic conceptual condition gradient described in Table 20, but note the limitations and assumptions outlined in Gaylard *et al* (2013a).

Table 9 Conceptual condition gradient for unvegetated sediments

Component	Excellent	Very Good	Good	Fair	Poor	Very Poor
Unvegetated sediments				Prevalence of deposit feeding epi-benthic animals	Dominance in deposit feeding epi-benthic animals compared to reference condition	

Within NSG the dominant habitat on soft sediment is seagrass. As a seagrass habitat degrades for whatever reason the habitat becomes bare sand. The delineation between seagrass habitats, severely degraded seagrass habitats where the seagrass has been lost due to disturbance and naturally unvegetated sediments is very difficult especially when reaching deeper depths where light may naturally limit the extent of seagrass. The nearshore MER program has a cut off of 15 m depth which is designed to help overcome this delineation where natural light attenuation through water may be a key factor in seagrass survival. Gaylard *et al* 2013a) describes a process of assessment where unvegetated sediment habitats are encountered in the NSG. This assessment looks at a number of key factors that may contribute to the likelihood of seagrass being able to survive or have previously survived at that location. Where seagrass is determined not to be able to survive a coarse conceptual model of the condition of unvegetated sediment (Table 9) will be used to describe condition.

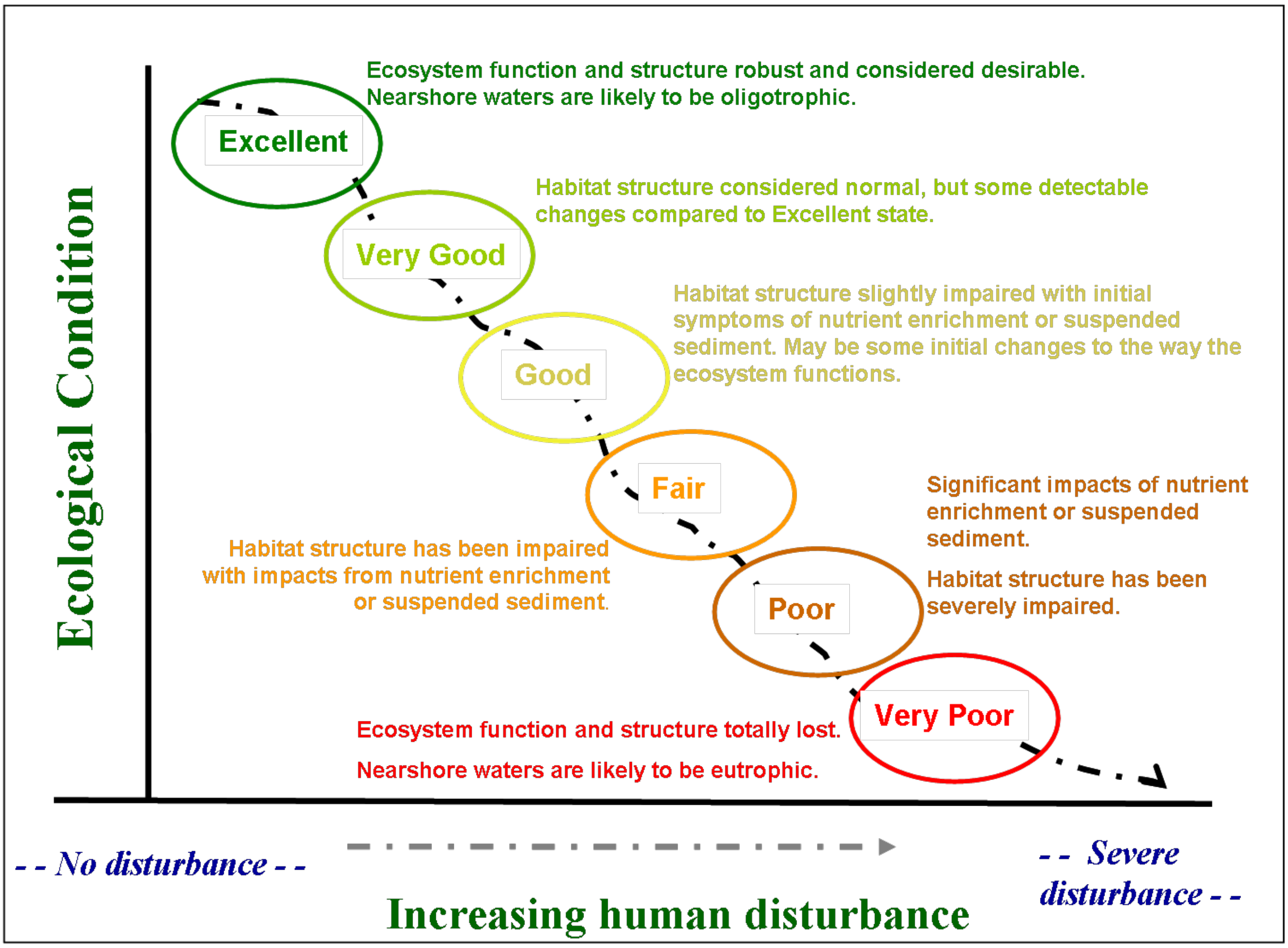
Multivariate comparisons

The conceptual models and the data compiled in Gaylard *et al* (2013a) reflect the physical, biological and chemical attributes that may be typical in South Australian nearshore habitats in in waters between 2–15 m deep when in Excellent condition. This work also shows how attributes may change along a disturbance gradient relating to declining condition (Appendix 3). In order to demonstrate how sites assessed in this MER program fit within the conceptual models, reference points were created using the information within the models to show how these attributes changed in comparison to data collected in this MER program and displayed using a multivariate (nMDS) plot. Overlaying the sites with the reference points will show where the composition of the sites fit with the conceptual models and whether there are biological gradients present in the data or other patterns that may help interpret the data in relation to pollution sources, natural variation or other factors.

Table 10 Conclusions for the broad scale condition of the biounit based on the conceptual models

Component	Excellent	Very Good	Good	Fair	Poor	Very Poor
Ecosystem	Structure and function of habitats considered in natural or unimpacted condition. Nearshore waters are likely to be oligotrophic. Adequate light for a maximum photic zone.	Habitat structure considered natural, but some detectable changes compared to Excellent state. Habitat changes are unlikely to be leading to changes in ecosystem function. Any detrimental effects are limited to small pockets and quickly reversible.	Habitat structure slightly impaired with initial symptoms of nutrient enrichment or suspended sediment. May be some initial changes to ecosystem function. Detrimental effects limited to site level changes but limited to short-term recovery.	Habitat structure has been impaired with impacts from nutrient enrichment and/or suspended sediment. These habitat changes are likely to be changing ecosystem function including resilience, biodiversity, productivity, and sediment stability. Detrimental effects may extend to numerous sites or small areas where longer-term recovery is required.	Habitat structure has been severely impaired leading to significant changes to ecosystem function including resilience, biodiversity, productivity, and sediment stability. Significant impacts of nutrient enrichment and/or suspended sediment. Detrimental effects may extend to numerous sites and possibly long-term recovery.	Ecosystem function and structure totally lost. Nearshore waters are likely to be eutrophic. Detrimental effects at a regional scale and recovery may not be possible.

Appendix 3 Ecological condition gradient



Appendix 4 List of site numbers, biounits and locations

Biounit	Site	Latitude	Longitude	Description
Winninowie	m0210	–32.5801	137.7976	Port Paterson
Winninowie	m0211	–32.7402	137.8822	Miranda
Winninowie	m0212	–32.6881	137.7884	Blanche Harbour
Winninowie	m0213	–32.8469	137.8307	Douglas Point
Winninowie	m0214	–32.8255	137.874	Mount Gullet
Winninowie	m0215	–32.9333	137.8597	Baroota
Winninowie	m0216	–32.9534	137.7682	Fitzgerald Bay north
Winninowie	m0217	–32.9675	137.789	Fitzgerald Bay south
Winninowie	m0218	–32.9754	137.9224	Ward Point
Winninowie	m0219	–32.9219	137.7636	Fitzgerald Bay inner
Yonga	m0200	–33.3175	137.4006	Pines
Yonga	m0201	–33.43	137.3958	Plank Point
Yonga	m0202	–33.4756	137.4154	Glensea
Yonga	m0203	–33.5371	137.3591	Airfield
Yonga	m0204	–33.6071	137.2915	Confier north
Yonga	m0205	–33.645	137.344	Confier south
Yonga	m0206	–33.6825	137.2505	Shoalwater Point north
Yonga	m0207	–33.716	137.1682	Lucky Bay east
Yonga	m0208	–33.7275	137.2486	Shoalwater Point south
Yonga	m0209	–33.7226	137.0874	Lucky Bay west
Yonga	m0220	–33.3372	137.7916	Woods Point north
Yonga	m0221	–33.3642	137.7906	Woods Point south
Yonga	m0222	–33.4667	137.7438	Fisherman Bay north
Yonga	m0223	–33.5329	137.6929	Fisherman Bay south
Yonga	m0224	–33.5278	137.8546	Fisherman Bay inner
Yonga	m0225	–33.6211	137.7919	Webling Point
Yonga	m0226	–33.6234	137.7056	Webling Bay
Yonga	m0227	–33.6828	137.7357	Tickera Bay inner
Yonga	m0228	–33.6896	137.6545	Tickera Bay outer

Biounit	Site	Latitude	Longitude	Description
Yonga	m0229	–33.7499	137.701	Myponie Point
Yonga	m0230	–33.0276	137.9631	Port Germein
Yonga	m0231	–33.0397	137.7865	Ward Spit
Yonga	m0232	–33.0481	137.9466	Cockle Spit
Yonga	m0233	–33.0494	138.0001	Germein south
Yonga	m0234	–33.0856	137.8222	Germein Bay
Yonga	m0235	–33.1113	137.9498	Germein Bay inner
Yonga	m0236	–33.1389	137.7935	Germein Bay south
Yonga	m0237	–33.1508	137.7287	Weeroona Sands
Yonga	m0238	–33.1537	137.8412	Fisherman Creek
Yonga	m0239	–33.1786	137.7599	Port Davis
Yonga	m0240	–32.9828	137.6815	False bay inner
Yonga	m0241	–32.987	137.6706	False Bay north
Yonga	m0242	–32.996	137.6703	False Bay south
Yonga	m0243	–32.9974	137.7237	Black Point
Yonga	m0244	–32.9983	137.7504	Stony Point
Yonga	m0245	–32.9999	137.6177	One Steel
Yonga	m0246	–33.0008	137.6808	False Bay outer
Yonga	m0247	–33.0063	137.718	Black Point outer
Yonga	m0248	–32.9987	137.7717	Point Lowly
Yonga	m0249	–32.9873	137.6917	west of Black Point
Yonga	m0250	–32.988	137.7132	Black Point inner
Yonga	m0251	–33.022	137.6262	Whyalla north
Yonga	m0252	–33.0441	137.615	Whyalla south
Yonga	m0253	–33.0672	137.5841	Mullaquana
Yonga	m0254	–33.0759	137.6245	Eight Mile Creek Beach north
Yonga	m0255	–33.0969	137.5701	Eight Mile Creek Beach south
Yonga	m0256	–33.1414	137.5418	Murrippi Beach
Yonga	m0257	–33.1766	137.5188	Western Shoal north
Yonga	m0258	–33.2029	137.5016	Western Shoal south

Biounit	Site	Latitude	Longitude	Description
Yonga	m0259	–33.2159	137.4743	Cowleds Landing