Environment Protection Authority

The South Australian monitoring, evaluation and reporting program for aquatic ecosystems

Rationale and methods for the assessment of nearshore marine waters

EPA
South Australia

The South Australian monitoring, evaluation and reporting program for aquatic ecosystems: Rationale and methods for the assessment of nearshore marine waters

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Contents

List of figures

List of tables

Summary

The Environment Protection Authority (EPA) has managed a monitoring, evaluation and reporting (MER) program since 1995 which historically assessed mainly water chemistry parameters to quantify ’ambient water quality‘ using comparisons to the values in the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC/ARMCANZ 2000).

In 2009, a new approach was conceived. *The South Australian monitoring, evaluation and reporting program for aquatic ecosystems: context and overview* (Goonan et al 2012) describes the rationale behind redesigning the statewide MER program for all waters (inland surface waters and nearshore marine) in order to use multiple lines of evidence to investigate broad ecological condition rather than just water chemistry. The report details the program’s context in a worldwide and local perspective and the methods used for reporting.

South Australian nearshore marine waters are widely recognised as diverse, extensive, and highly productive, but subject to a broad range of external pressures. The development of a comprehensive framework for the assessment and monitoring of environmental threats is therefore challenging, particularly given the fact that many of our marine systems have suffered at least some degree of historical decline.

The current report summarises the methods used by the EPA to assess broad condition of the nearshore marine waters throughout South Australia (the Nearshore MER program) and the development of Aquatic Ecosystem Condition Reports (AECR) to convey this information to the general public and key stakeholders. The spatial framework used to classify condition throughout shallow nearshore waters within South Australia is based upon *bioregions* published by Integrated Marine and Coastal Regionalisation of Australia (IMCRA v4.0) and *biounits* published by Edyvane (1999). The MER program uses a risk-based, three-tiered structure for assessing condition according to an ecological condition gradient. This approach ensures that the Nearshore MER program is:

1 efficient with respect to the targeting of onground sampling resources

2 making more effective use of historical data

3 flexible in terms of the nature of the sampling with a capacity to incorporate existing monitoring frameworks

4 capable of identifying and responding to knowledge gaps and driving research priorities

5 supportive of higher level State of the Environment/State of the Region reporting

6 capable of promoting and prioritising longer-term monitoring.

The Nearshore MER program aims to identify priorities based on historical data and current knowledge rather than using benchmarked guidelines of water chemistry derived without consideration of potentially significant biological and chemical interactions. This approach will bring South Australia into line with most other major environmental agencies throughout the world, and will be seen as a fundamental element in determining the current state of the marine environment. The concept of ‘multiple lines of evidence’ is increasingly being used to ensure that monitoring takes into account the inherent variability and perturbations possible in the natural environment, and that conclusions about condition are well founded and reliable, and still sensitive to subtle changes.

The Nearshore MER is structured around three levels (or tiers) of investigation.

Tier 1

A desktop threat assessment which reviews the available literature detailing the current and past threats to water quality and any historical monitoring and related data that may be available. Where information is available, this process will attempt to reconstruct the historical baseline for the region to enable comparisons to present and future observations. This process will define the likely stressors on habitat condition which will comprise the targets for onground monitoring. Where data are not available, a suitable level of caution is applied to conclusions and a statement of confidence may be applied to some outputs.

Previous work has shown that in shallow nearshore marine waters, the stressors are predominantly increased nutrients and/or sediments loads and decreased water clarity, and as such the program is initially designed around these issues. Within the Tier 1 stage a 'predicted' condition will be defined by the threat assessment process in order to estimate what the current condition should be based on the current risk factors. Consideration will also be given to historical long-term impacts where the pressure may or may not have been subsequently removed (eg historical seagrass loss from wastewater treatment plant sludge pipelines along the Adelaide metropolitan coast).

Tier 2

The information developed in the Tier 1 threat assessment process is to inform a field monitoring program. The program comprises a hierarchical (nested) program of monitoring at three different spatial scales (site, biounit and bioregion) that is aimed at broadly assessing the current habitat condition. Observations are to be undertaken over two time periods (autumn and spring), with the observed condition being the average of these snapshots. The program is designed to allow for specific questions to be identified and potentially answered at multiple scales, and is aimed to facilitate integration with other stakeholders (eg NRM boards). For example at the regional scale, this monitoring could contribute to the reporting within the state or local NRM plans such as Goal 3 of the State NRM plan: Improved condition and resilience in natural systems: coastal and marine ecosystem extent and condition (Government of South Australia 2012).

The Tier 1 threat assessment and historical status conceptual modelling are then compared with the results of the Tier 2 field program to develop an **Aquatic Ecosystem Condition Report (AECR).** The AECR comprises indicators for both the predicted and the observed condition of the system. This reporting is to be based on the results at the scale of 10s to 100s of km (biounit), but can be rolled up to the bioregional scale for State of the Environment reporting.

Tier 3

Where the observed Tier 2 condition is different to the expected condition defined in Tier 1, further investigations may be undertaken to identify possible reasons/causes; if initiated this would form a Tier 3 project that would be mentioned on the AECR. The Tier 3 project would include finer spatial- and/or temporal-scale resolution, identification of key processes and/or pollutant source detection methods to isolate relative contributions of stress factors within the target biounit. This work would aim to directly feed into management of any identified significant anthropogenic pollutant source in order to help facilitate a change in condition.

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1 Introduction

South Australia has over 4,000 km of coastline including cliffs, rocky shores, sandy beaches, mud flats, mangroves and samphire habitats (EPA 2003). Southern Australian marine waters are biologically very diverse and have a very high number of endemic species, due largely to geographic isolation (Poore 1995). These nearshore marine ecosystems are in a fine balance with numerous biotic and abiotic variables that exist naturally, and disrupting this balance can drive changes to the ecology in an area (Waldichuk 1977). Nearshore waters are exposed to a range of external pressures which can adversely affect their condition. These pressures can be wide and varied including short-term pulsed inputs such as stormwater, through to constant press discharges including sewage treatment plants and industrial discharges. The impacts on ecosystem condition can be temporary or permanent as well as localised or wide ranging.

Studies have shown that for South Australian marine waters, even small increases in nutrient concentrations can have disproportionate degenerative effects on biotic environments including increased epiphyte loading on seagrass leading to seagrass loss (Bryars et al 2011), and shifts from canopy-macroalgal to turf-dominated reef systems (Gorgula and Connell 2004; Russell et al 2005). Shifts in habitat structure from complex communities comprised of largely perennial species to simple systems with less diverse and opportunistic species are generally seen as undesirable, or in poor condition (Tilman et al 2001; Hughes et al 2005). These changes can be initiated by natural as well as anthropogenic driven disturbances.

The consequences of ecological systems in poor condition can be wide ranging and include the loss of ecosystem services which for seagrass systems in the US have been valued at approximately US$19,000 per hectare of seagrass per year (Costanza et al 1997); a figure which has been applied to South Australian waters and valued between AUD$12,635–$25,270 per hectare per year (Lothain 1999). These values take into account the roles that seagrass habitats play including production of commercial and recreational fisheries, nutrient and carbon cycling, sand movement and wave attenuation.

Using the above figures, seagrass loss within Gulf St Vincent, estimated at 9,000 ha (Edyvane 1999) is costing the South Australian community between $113–227 million each year. The value of rocky reefs is largely unquantified but could be considerable based upon the roles that rocky reef systems play in maintaining biodiversity, contributing to fisheries (Airoldi et al 2008) and nutrient cycling (Holmlund and Hammer 1999).

The development of a comprehensive framework for the monitoring and assessment of environmental threats and condition is therefore challenging, particularly given the wholesale changes to the terrestrial landscape altering quality and quantity of flows to the sea (Goonan et al 2012). The causes have likely resulted in changes to many of our nearshore marine systems.

Traditional experimental ecology can provide information with great confidence and precision, although fragmentary (Connell and Irving 2009), time consuming and expensive to undertake. With this in mind, monitoring techniques that facilitate the rapid assessment of multiple lines of evidence can offer an improved early-warning indicator of impact over a wider spatial scale and ultimately give more time for scientists and managers to implement decisive actions to avert escalating problems, while concurrently building experimental knowledge to further refine management strategies and our understanding of the system and its responses to pressures (Littler and Littler 2007). Rapid assessment over large spatial scales can also highlight ‘hotspots’ in need of more detailed investigation, thereby prioritising future works based on real information.

The South Australian Environment Protection Authority (EPA) coordinates a number of monitoring, evaluation and reporting (MER) programs across a range of aquatic systems throughout the state. Within South Australian nearshore marine waters, this ‘Nearshore MER program’ is to be based around defining habitat condition using a three-tiered assessment process and incorporates the development of Aquatic Ecosystem Condition Reports (AECR) as the primary means of communication to the general public and key stakeholders. An assessment report for each bioregion will also be developed which will detail the scientific evaluation of the results aimed at a scientific audience rather than the general public. This approach ensures that the nearshore marine MER program can define broad habitat condition across the state but is also:

- efficient with respect to the targeting of on-ground sampling resources

- making more effective use of historical data
- flexible in terms of the nature of the sampling with a capacity to incorporate existing monitoring frameworks
- capable of identifying and responding to knowledge gaps and driving research priorities
- supportive of higher level State of the Environment/State of the Region reporting
- capable of promoting and prioritising longer term monitoring.

The nearshore MER program therefore aims to identify monitoring and management priorities based on historical data and current knowledge using 'multiple lines of evidence' within biological systems, rather than using benchmarks of water chemistry derived without consideration of potentially significant biological and chemical interactions. Worldwide, monitoring on ambient condition is moving away from solely chemical-based comparisons with guideline values towards using biological information supported by chemical data to derive conclusions about condition.

This EPA approach will bring South Australia into line with many leading environmental monitoring agencies including the Healthy Waterways program in South East Queensland, the Great Barrier Reef Marine Park Authority, the European Union Water Framework Directive and the US EPA.

The broad objectives of the Nearshore MER include:

- providing a statewide monitoring framework for nearshore marine waters with sufficient frequency to allow for State of the Environment reporting purposes
- describing aquatic ecosystem condition for broad public understanding
- identifying the key pressures and appropriate management responses
- providing a reporting format that can support environmental decision making within government, community and industry.

1.1 The purpose of this report

This report provides a summary of:

- The habitat types to be targeted by the Nearshore MER program as well as the threats to their condition and the potential for change in condition over time.
- A description of the methods used in the assessment of broad condition of South Australia's nearshore marine waters and the development of the AECR including a justification for these methods based on the given objectives and the current level of resources available.
- A summary of investigations undertaken to test the veracity of the field program as well as identification and application of various indicators.

2 South Australian nearshore marine ecosystems, threats and responses

The Nearshore MER program aims to broadly assess ecosystem condition based upon the characteristics of the dominant subtidal habitats in the nearshore marine waters across South Australia. The dominant habitats throughout South Australia's nearshore sub-tidal waters (<15 m depth) are seagrass, rocky reefs and unvegetated soft sediment (Edyvane 1999a). The condition of each habitat is defined by common models for degradation due to the dominant (large scale) stressors which are generically defined in this MER program as nutrient input and water clarity. These models are broadly described below and in Figures 6–9 and Tables 1–7.

It is accepted that there are other types of habitats in South Australian waters in addition to those discussed above (eg sponge gardens and mega ripples) and that while these habitats may cover relatively smaller areas, or waters outside the scope of this MER program, they may still be regionally significant. If localised 'atypical' habitats are encountered in the Nearshore MER program then the conceptual models may be reviewed. If condition cannot be adequately quantified then a baseline state will be measured and used to assess change over temporal scales based on the previous monitoring.

There may be other stressors that feature in a particular bioregion (eg metals in northern Spencer Gulf) but these specific risks will be addressed individually through targeted MER programs rather than in the state-wide MER program. Similarly localised issues such as impact assessment from complex effluents will continue to be assessed under industry monitoring requirements administered by the EPA or other regulatory agencies.

2.1 Seagrass

Seagrasses in South Australia dominate the sheltered nearshore habitats, particularly in Spencer Gulf and Gulf St Vincent. These meadows occur in areas where there is some protection from extreme water movement (Walker and McComb 1992). Seagrasses are important habitats as they provide extensive shelter for a wide variety of fish and invertebrates (Pollard 1984; Bell 1989; Orth et al 2006), enhance biodiversity (Duarte 2000) and provide extensive ecosystem services (Walker and McComb 1992; Costanza et al 1997; Duarte 2000).

The maintenance of seagrass habitats will ensure that they remain productive and provide long-term ecosystem services to the surrounding environment and the South Australian communities that rely on them. Measuring different characteristics of these habitats is vital in being able to assess their condition. Within this context, the indicators that are measured need to function as early warning indicators to highlight that impacts are starting to occur before the habitats are significantly degraded or even lost.

Recent and historical investigations into the shallow benthic habitats along the nearshore waters of the gulfs suggest that these waters are largely dominated by the angiosperm genera *Posidonia* and *Amphibolis*. These species are known to mostly form large mono-specific meadows due largely to their vegetative growth, in addition to some propagule growth (Kirkman 1985).

There are two primary mechanisms resulting in seagrass loss:

- physical loss or removal
- lack of light.

The causes of impacts can be either natural or induced by anthropogenic activities, either in the water or on adjacent catchments that discharge into nearshore waters during rain and storm events.

Causes of physical loss can include:

- loss from intense storm activity and wave action (Cambridge and McComb 1984)
- grazing (Rose et al 1999)
- dredging (Erftemeijer and Robin Lewis 2006) and/or smothering
- boat moorings (Walker et al 1989; Hastings et al 1995) [Figure 1] and propeller damage (Hammerstrom et al 2007)

Causes or contributing factors of seagrass loss from lack of light include:

- reduced light due to turbidity or shading by physical structures (Neverauskas 1988; Longstaff and Dennison 1999; Collings et al 2006b)
- increase in epiphytes on seagrass due to a shift in nutrient regime reducing the amount of light available to the seagrass leaf (Neverauskas 1987a; Shepherd et al 1989b; Bryars et al 2011)
- increase in phytoplankton due to changed nutrient availability resulting in decreased light penetration through the water column (Moore et al 1996; Duarte et al 2000; Morris et al 2007).

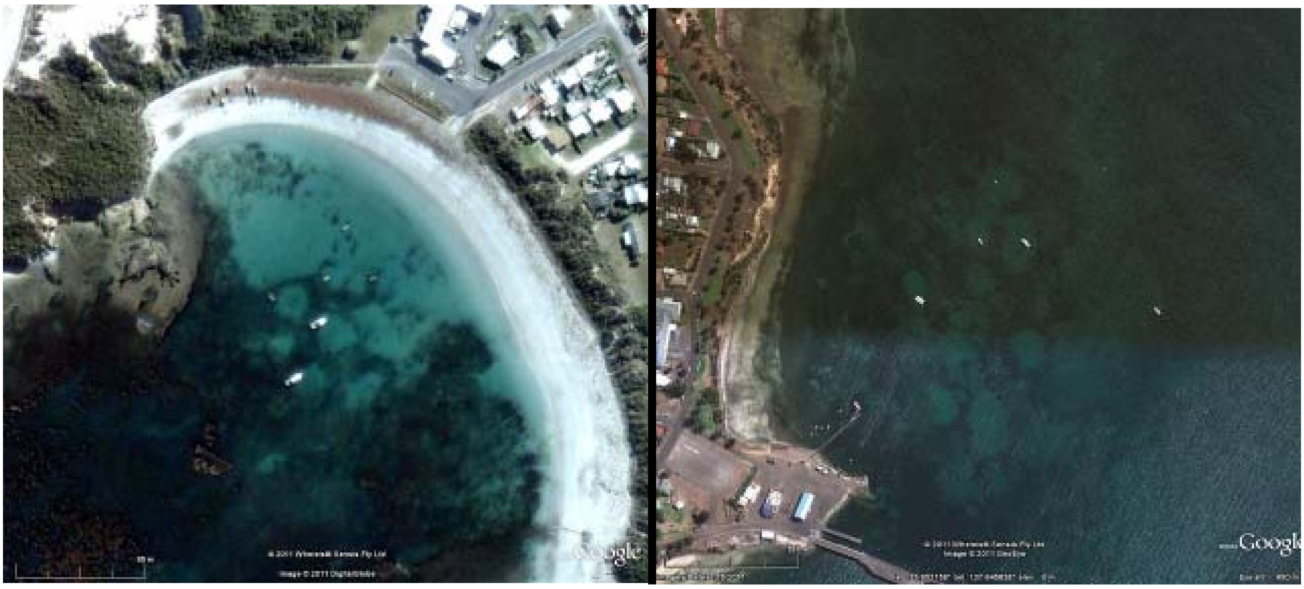

Figure 1 Circular scours in seagrass meadows in the South East and Kangaroo Island of South Australia. Circular scours particularly in close proximity to boat anchorages have been shown to be symptomatic of loss due to scour caused by swing boat moorings (Walker et al 1989; Hastings et al 1995) .

2.1.1 Seagrass loss due to lack of light

An increase in nutrient loads into the nearshore waters may result in a shift in the nutrient availability and potentially a change to biological limitation in the receiving waters (Howarth and Marino 2006). This altered regime may result in an increase in epiphytes on seagrass leaves (Bryars et al 2011) and/or an increase in phytoplankton or macroalgae, both reducing the amount of light available to seagrass potentially leading to the seagrass becoming starved of light (Shepherd et al 1989a). Light is a key determining factor in the maximum depth limit of seagrass, thus naturally limiting the depth to which seagrass can grow (Duarte 1991). It then follows that seagrass loss due to a lack of light would be more likely to occur at the outer depth limit where there is less available light, and there are numerous examples to support this including Botany Bay, NSW (Larkum 1976), Princess Royal Harbour, WA (Hillman 1991) and Chesapeake Bay, USA (Orth and Moore 1984). However, seagrass losses due to excess nutrients and epiphytes can still occur in shallow waters (Bryars et al 2011).

Poor water clarity has also been shown to reduce the available light to seagrass resulting in seagrass loss. This was seen as a significant contributing factor to seagrass loss in the nearshore waters of Adelaide's metropolitan coast (Collings et al 2006b; Fox et al 2007). Poor water clarity reduces the amount of light able to pass through the water column to the seagrass leaves. Poor water clarity can be caused by an increase in suspended material in the water column (eg sediment and phytoplankton) and from the presence of coloured dissolved organic matter (CDOM) which scatters or absorbs the light before it reaches the seagrass (Shepherd et al 1989b). This material can enter the water column from terrestrial sources, marine sources such as dredging or from the re-suspension of benthic sediment [see South Australian specific review in Gaylard (2009b)]. Additionally it is possible that smothering of seagrass by excessive sediment may result in direct loss by blocking light (Shepherd et al 1989b).

There are circumstances where seagrass may be lost from an environment but the original cause of the loss has been removed (eg dredging event or historical nutrient loads). An episodic event can cause a negative feedback cascade whereby seagrass continues to be lost due to the change in seagrass cover. This can include increased wave energy through loss of attenuation properties provided by seagrass resulting in expanding blowouts (Seddon et al 2003) and increased resuspension of sediments due to loss of seagrass resulting in increased turbidity and further seagrass loss (Duarte 1995).

2.2 Rocky reefs

Available rocky or consolidated substrate allows the settlement of algae and sessile invertebrates and, in southern latitudes where there is sufficient light, macroalgae tend to dominate these subtidal reefs (Turner et al 2007). Macroalgae are conspicuous parts of a reef which contribute significantly to the biodiversity and the ecosystem services of reefs (O'Hara 2001). Macroalgal communities tend to comprise a number of identifiable layers that can coexist in the same assemblage (Shepherd and Sprigg 1976) and some can be considered to be foundation species (sensu – (Dayton 1975) due to their large influence on the structure of the assemblages (Connell 2007).

Across much of South Australia the upper layer is made up of large brown canopy forming species (kelp), which are a conspicuous and extensive group of algae typically including the large brown algae in the orders Fucales (commonly *Cystophora* spp, *Sargassum* spp, *Seirococcus* spp and *Scytothalia* spp) or Laminariales (mainly *Ecklonia radiata* and possibly *Macrocystis* spp *and Durvillaea* spp in the South East of SA) (Connell 2007; Turner et al 2007). Rocky reefs create substantial habitat complexity and support some of the most productive and biologically diverse systems (Connell 2007), and the maintenance of these systems is vital for biodiversity, ecosystem services and fisheries.

Large brown macroalgae have been shown to be susceptible to poor water quality (Turner et al 2007; Collings et al 2008; Connell et al 2008) and the removal of these foundation taxa facilitates a shift to a less complex and less productive system dominated by small turfing species (Airoldi and Cinelli 1997; Gorgula and Connell 2004; Connell et al 2008), which results in a loss of diversity (Airoldi and Cinelli 1997). Due to their central role in a range of ecological processes, the loss of canopy forming algae is likely to lead to the significant loss of associated species and ecological function (Steneck et al 2002; Ling 2008).

Changes to rocky reef systems can be driven by a large number of factors, similar to seagrass systems that can be natural or anthropogenic in origin. The list below is taken largely from Turner et al (2007) but it highlights the most likely potential stressors to subtidal reef systems; that this list is by no means comprehensive. It should also be noted that many of these stressors may act in combination, particularly where an anthropogenic disturbance may drive changes in natural processes. Thereby the differentiation between natural and anthropogenic stressors is blurred. Potential stressors to subtidal reef systems include:

- herbivory (Steneck et al 2002)
- turbidity and sedimentation (Airoldi 2003; Turner 2004)
- climate and geological changes (Ling 2008)
- eutrophication (Gorgula and Connell 2004; Russell et al 2005)
- extractive resource use (Shears and Babcock 2003)
- toxicity due to a range of potential compounds (Ross and Bidwell 1999)
- invasive species (Grosholz 2002).

While the amount of research looking at cause and effect relationships on macroalgal reefs in South Australia is limited, the available work suggests that, consistent with the recognised mechanisms for seagrass loss in South Australia, the main drivers are likely to be increases in nutrient and sediment loads (Turner et al 2007; Connell et al 2008).

Connell (2007) highlights that the reasons for decline in kelp species on South Australian rocky reefs is not well understood, but the loss of canopy forming kelp species could be related more to declining water quality, particularly in sheltered areas (eg gulfs) lacking a functional group of herbivores, eg South Australia (Connell (2007) (Airoldi and Cinelli 1997; Benedetti-Cecchi 2000). While declining water quality may not directly cause the loss of the kelp species it is recognised that the addition of nutrients and sediments into reef environments can result in an increase in turfing algae (Gorgula and Connell 2004) and that the turfing algae not only benefit from the nutrient additions but their ability to trap sediment allows these species to thrive in areas of heavy sedimentation and exclude the natural recolonisation of the kelp species (Airoldi and Cinelli 1997).

2.3 Unvegetated sediments

The role of unvegetated soft sediments in the marine environment is often understated, but these habitats should not be considered as barren or unimportant. Soft sediments are home to a vast number of species (with many yet to be described), that provide multiple ecological roles including nutrient and carbon cycling, bioturbation, as well as providing food for higher trophic levels (Hutchings 1998). Threats to unvegetated sediments are similar to vegetated sediments in that excess nutrients, including carbon in terms of organic loading, can lead to a reduction in biodiversity of organisms living on and in the sediments and also disruption to gas exchange between the sediment and overlying water (Christensen et al 2003). The decomposition of organic matter by bacteria can result in bacteria utilising much of, if not all of the oxygen in the overlying water, which can lead to impacts on marine organisms, particularly sessile organisms that cannot move away from the deoxygenated water (Kamykowski and Zentara 1990). In many cases hypoxia can result from eutrophication of the water body and poor water exchange (Paerl et al 1998).

Cheshire et al (1996) found significant differences in the epi-benthic macrofauna at varying distances from southern bluefin tuna aquaculture installations in Boston Bay (SA) which were linked to the degree of organic loading. Conclusions of their study were similar to other published literature eg Brown et al (1987) which described zones of impact linked to the epi-benthic communities including an area of high impact which was dominated by polychaetes, nebalids, brachyurans and anthozoans with intermediate numbers of holothurians ascidians and sea urchins in very close proximity to the source (Cheshire 1996). Radiating out from the highly polluted zones the abundances of organic deposit feeders such as ascidians and holothurians were increased compared to control sites suggesting increased organic loads compared to control sites (Cheshire 1996).

In addition to changes in epi-fauna from organic loading there may also be changes to the community structure within the sediment (infauna). Organic loading has been shown to cause three main changes in infaunal communities:

- an increase in abundance of species tolerant to organic enrichment
- a decrease in taxonomic diversity
- a change in community structure (Madigan et al 2001).

In many cases, utilising these indicators allows changes to benthic condition above natural variability to be detected.

3 The Nearshore MER Framework

South Australia's 4,000 km of coastline diverse biologically and geomorphologically. It is expansive and in some places difficult to access. Monitoring all facets of all ecosystems across the state with high spatial and temporal replication is neither logistically practical nor necessary from a statistical or broadscale condition assessment viewpoint. Alternatively, focused monitoring on parameters that are likely to show changes linked to human disturbance from the threats in each region is considered to be a more efficient and cost effective approach (Gibson and Brown 2000; Ralph and Poole 2003). In order to ensure that potential major threats are considered and that the monitoring is appropriate for each region, a desktop threat assessment will be undertaken (section 4.1).

3.1 Bioregions and biounits

In order to undertake a Nearshore MER program across the state with the current level of resources, monitoring and reporting on state's waters must be broken down into smaller units in order to use the available resources efficiently. The Integrated Marine and Coastal Regionalisation of Australia (IMCRA v4.0) has used a spatial framework that delineates the marine environment throughout Australia's coastal waters into 'bioregions'. These bioregions are smaller spatial units based on collated biological data and inferred ecosystem patterns. Bioregions are an accepted tool in the description of ecosystem boundaries and are considered to be a useful spatial scale for regional planning and also supply a framework for smaller-scale ecologically relevant planning and management (DEH 2006).

At a finer spatial scale within each bioregion are the biounits; the boundaries of which delineate a finer resolution of inferred ecological boundaries and which can be measured in the 10s to 100s of km inside each IMCRA bioregion (Edyvane 1999a; Edyvane 1999b). The EPA's Nearshore MER program has been designed to report on a biounit scale, but has the ability to be combined to inform at the bioregional scale for statewide reports such as the State of the Environment report.

Habitat mapping undertaken by the Department of Environment, Water and Natural Resources (DEWNR) has further developed habitat boundaries throughout nearshore waters of South Australia, although at this stage the coverage is incomplete. Habitat maps generated by DEWNR will be used extensively to inform the Nearshore MER program and preference will be given to the DEWNR mapping if it conflicts with the older Edyvane (1999b) [see section 3.2.2). Maps of the biounits within each bioregion will be included in the publications detailing results for each bioregion.

3.2 Tiered monitoring

Tiered, adaptive monitoring programs where the spatial and/or temporal replication can be increased or additional parameters added (based on new findings), have been shown to be the most efficient monitoring approach (Ralph and Poole 2003). Generally, as the risk of significant environmental impact increases, the level or complexity of monitoring increases also. In this manner, sampling is targeted and results in increased efficiency in resource allocation. There are many examples of tiered monitoring programs for assessing environmental condition in the aquatic environment, including the US EPA's Integrated Water Resources Monitoring (Copeland et al 1999), sea cage aquaculture in New Brunswick (Department of Environment 2006) and the United States Geological Service (USGS) in their statewide water quality network (DeSimone et al 2001).

The Nearshore MER program has been designed using a three-tier framework. The tiers outline the process to assess the condition of the marine environment within this MER program.

- **Tier 1** – A literature review and desktop threat assessment are developed to assess threats to ecological condition in each bioregion. This information is used to review and update conceptual models and tailor monitoring to address identifiable risks specific to each bioregion, if required. A *predicted* condition for each biounit is also developed using the threat assessment and available published literature, particularly historical monitoring information.
- **Tier 2** – Using the information from Tier 1 a rapid field assessment program is developed that reflects the level of risk (eg the number of sites selected). The monitoring for the Nearshore MER program is undertaken throughout each biounit within a bioregion using two monitoring periods to assess condition. These monitoring periods will be autumn and spring (section 3.2.3) and the results will be used to develop the *observed* condition.

The information collected from Tier 1 and Tier 2 is presented in an online Aquatic Ecosystem Condition Report (AECR) communication tool for each biounit. The AECR provides:

1 A broad assessment of ecological condition.

2 A list of pressures (ie significant threats) to the biounit ecosystem.

3 A list of management responses that are in place to address those pressures (with responsibilities shared by government, industry and/or community)

- **Tier 3** – Where there are noteworthy differences between the *predicted* and *observed* conditions for a biounit, suggesting deficiencies in our understanding of risks and biological responses, the biounit will be highlighted as in need of further research. The publication of the AECR will still proceed with caution but a statement of limitations may be made on the document stating that there may be facets of the system that we do not fully understand and the area is highlighted to be in need of further research. This further work may be undertaken by the EPA or another institution and the program may include source detection methods to investigate the origin of specific pollutants driving the observed condition.

3.2.1 Bioregions

The Nearshore MER program has a nested design that operates across different spatial scales for monitoring and reporting (Figure 2). These scales include bioregion ➔ biounit ➔ site. The IMCRA bioregionalisation system provides a good starting point to delineate the EPA's monitoring effort over time. The EPA program has combined a number of the IMCRA bioregions for the reasons of logistics, risk and access for sampling. The modified bioregions can be seen in Figure 3.

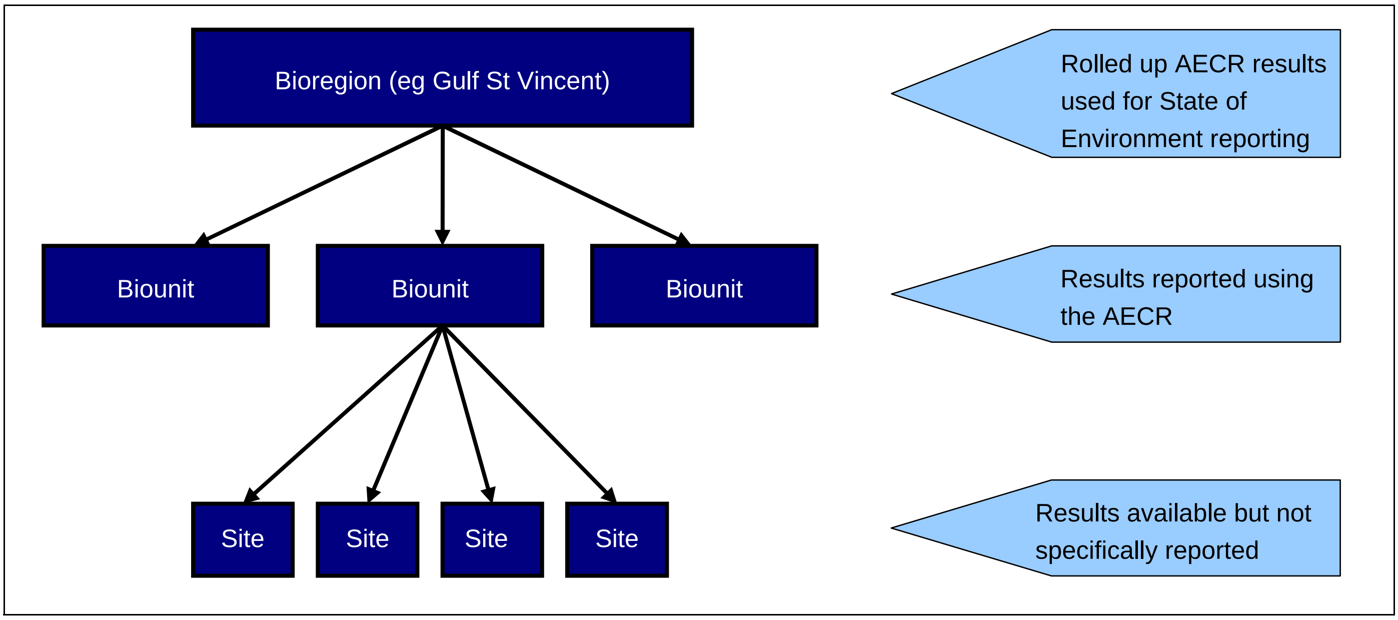

Figure 2 Nested design of nearshore marine monitoring and evaluation program

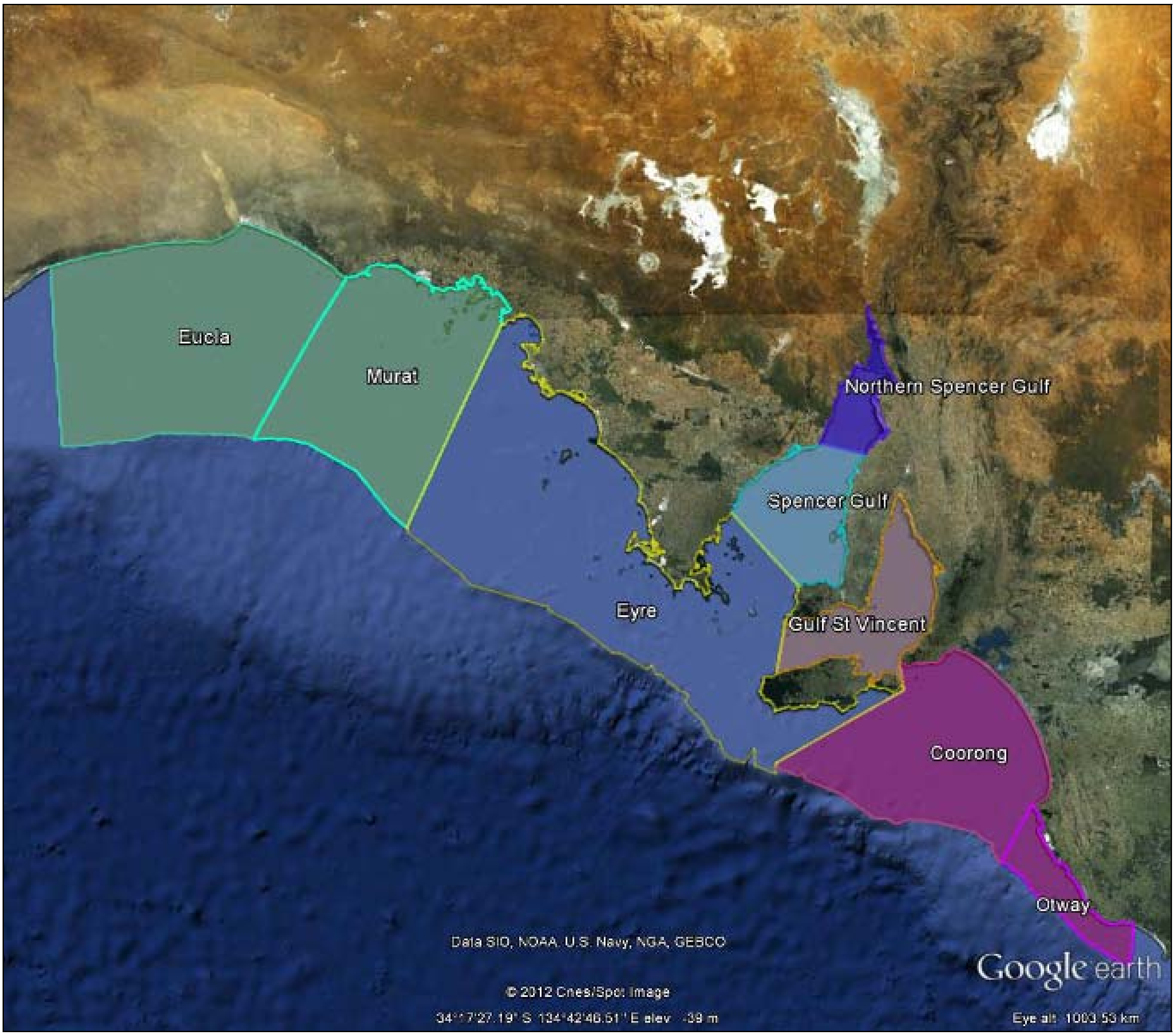

Figure 3 **IMCRA Bioregions. The MER program has combined the IMCRA bioregions of Eucla and Murat into one region, and Coorong and Otway into another region for logistical reasons.**

3.2.2 Biounits

For investigating at a finer level of detail than the bioregion, the biounits delineated by Edyvane (1999c) will be used as the reporting units for the Nearshore MER program using the AECR reporting tool. It is accepted that the Edyvane (1999c) mapping has its limitations, and that DEWNR has mapped in finer resolution many of the benthic habitats throughout South Australia which would more precisely delineate habitat boundaries. However the DEWNR dataset is incomplete; therefore the EPA Nearshore MER program will use the Edyvane (1999c) biounit delineations, but also refer to the DEWNR work where applicable.

There may be instances where a biounit is modified from Edyvane (1999c) for logistical reasons or to enable finer or clearer resolution of information. If this occurs the delineation will be outlined in the detailed publication of results for each bioregion (the *Assessment Report*).

In order to monitor each biounit, a number of sites will be sampled to gain an understanding of the condition of the area (Figure 4). Each of the sites will be comprised of replicates to ensure that there is increased confidence in results at the site, biounit and bioregion levels. This hierarchical or nested framework allows for reporting at multiple spatial scales for a number of different potential purposes. The Nearshore MER program reports at the biounit level.

Figure 4 **Example biounits within the Gulf St Vincent bioregion. Maps delineating biounits in each bioregion will be available in the regional assessment reports.**

3.2.3 Sites

As stated above, each biounit will be sampled incorporating multiple sites to ensure that results are as representative of the broadscale condition as practical. Sites within each biounit will be determined by using a random sampling design using a 500 m x 500 m grid overlayed on a bathymetric map of the biounit. Each site will be located between 2.0–15 metres deep (Lowest Astronomical Tide – LAT). The number of sites will be based on the expected condition determined by the Tier 1 assessment taking into account logistical and financial constraints. Typically, as the level of disturbance increases, the variability in ecological responses also increases (Odum et al 1979; Warwick and Clarke 1993) and as such the number of sites within the biounit will be modified to match the expected condition. In subsequent years of sampling, the number of sites will be refined based on the power afforded by previous results, within financial and practical constraints. For full details of the number and location of sites within each biounit refer to the assessment reports for each bioregion which will be published on the EPA website.

Sampling of each bioregion will be undertaken during autumn and spring in that year. At the current level of resourcing and site allocations in the trial bioregions, sampling for each season takes approximately four weeks. This lessens any influence of short-term storms or runoff events over that period. This sampling regime is aimed to capture the peak of biological activity (autumn) where indications of excess algal and epiphytic algal growth would presumably be at their peak (King and Schramm 1976; Silberstein et al 1986). The second sampling period in spring is intended to capture ecological condition following the wet period of winter runoff into nearshore coastal waters (Pattiaratchi et al 2007) BoM 2012) and during the time when biological activity increases with warming water temperatures.

The results will be assessed as two individual snapshots of ecological condition rather than averaging the data across each sampling period. This method does not infer connectivity between data points over the sampling period, which is likely to be questionable given the infrequent temporal replication. Overall condition of the biounit is determined by the average of the condition of the two sampling periods. Using subsequent years of sampling, longer term conclusions can be made by comparing change in a biounit over different years being mindful that any temporal assessment will take into account seasonality by comparing similar seasons.

3.2.4 Replication within a site

Many monitoring programs use a measure of a particular variable or even replicated measures of an unspecified parcel of water or habitat with the assumption that it is representative of the area or site, eg EPA Victoria (2000). The size of a site that this sample actually represents is rarely defined outside of the size of a quadrat or transect length but the conclusions are often far ranging in terms of the spatial scale. The Nearshore MER program has defined the size of a site and also undertaken detailed analysis to attempt to define how many samples should be taken within a site to ensure that the measurements are representative. Within the Nearshore MER program, a site is specified as being an area of almost 20 ha (200,000 m^2). This area is largely an arbitrary assignment without any reason behind its allocation other than the acceptance that assessing habitat at very small scales (ie the scale of a single water sample or underwater video transect) can be quite variable. As such increasing the replication within the spatial scale of a site may allow for less variability (Wiens 1989; Moore and Fairweather 2006) and conclusions will be more meaningful for larger scale resource management.

The EPA undertook a fieldwork program in order to test monitoring approaches, including the habitat structure index (Irving et al 2013), using the methods described in Section 6 and Appendix 1. The habitat structure index is a method of assessing the condition of seagrass habitats using multiple parameters (area, density, species, patchiness) to determine the condition of the area (Irving et al 2013). A preliminary set of data taken from spring 2009 was used as a pilot data set to determine a statistically robust number of underwater video transects within a site for a set effect size to measure change in seagrass condition (% change). Using this information, paired with knowledge of what is logistically feasible and financially possible, a determination of the amount of replication within a site can be made.

Appendix 3 outlines the post hoc power analysis undertaken to determine the ideal and/or feasible number of random transects within a 20-ha site that can be applied for future sampling. The power analysis was undertaken using the parameters of seagrass condition, however other parameters will be assessed in the future once sufficient data on other habitats have been collected (eg rocky reefs). The sites analysed were representative of zones with known impacts from significant nutrient enrichment through to areas that are considered to be in relatively good condition (Gaylard 2004; Fox et al 2007). The power analysis revealed that 10 replicates across a 20-ha site (Figure 5) provided the ability to detect a 12% change in epiphyte density with a power of 0.95 (Appendix 3 Power analysis for seagrass condition). While increasing the replication would facilitate the detection of a smaller percentage of change, 10 replicates was also considered to be logistically feasible with the current resources and a 12% change in epiphyte density was considered to be sufficiently significant.

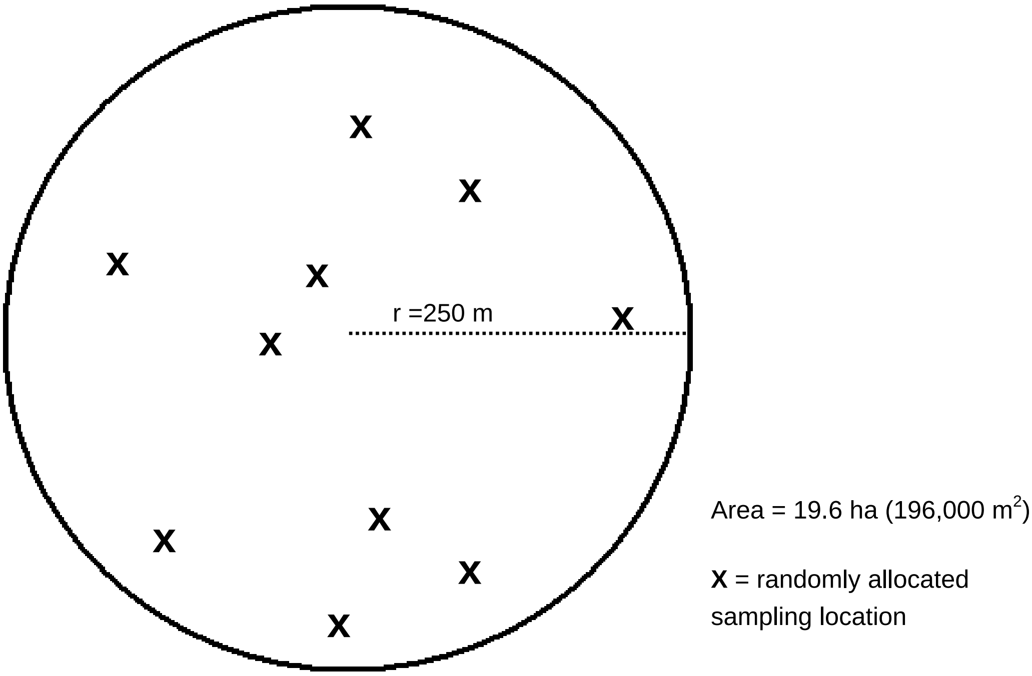

Figure 5 Example of 10 random sampling locations within a site

4 Conceptual models and ecological condition

Conceptual models of ecosystems and ecological processes are key tools used to describe and formalise the current state of knowledge of a biological system or process, and used to guide monitoring and research in order to encompass all significant factors. Conceptual models that describe the response of a system to stress have been used in developing strategies for natural resource management that put emphasis on the maintenance of important ecological characteristics. For example, the Adelaide Coastal Waters Study developed a conceptual model of inputs and processes underway in the nearshore waters of Adelaide, and throughout the five years of this study the conceptual model was constantly refined to inform the study design and aid in the interpretation of results (Fox et al 2007).

The biological condition gradient is a type of conceptual model that relates an observed ecological response to increasing levels of human disturbance (Davies and Jackson 2006). This gradient assumes that biological 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 of the environment. The model, as employed by Davies and Jackson (2006), can be divided into six levels of condition along a disturbance–response curve (Figure 6). The curve ranges from no observable disturbance (natural or unimpacted condition) through to conditions found at high levels of disturbance (severely degraded). As this model does not infer causality, natural degradation will still be captured and assessed as a disturbance, for example a large storm prior to sampling. However, these events are still valid in the assessment of condition of a biounit, consideration only needs to be given if the process is assigning the cause of the impact and under the Nearshore MER program this would require a Tier 3 assessment. If a ‘natural’ disturbance is considered likely, this will be explained in both the AECR and the assessment report for that location.

The original biological condition gradient model was developed for freshwater streams for the US EPA (Davies and Jackson 2006; Stoddard et al 2006). However, it provides a useful generalised framework for summarising and communicating the broad ecological condition across a range of systems, including marine environments (Bradley et al 2010). In the Nearshore MER program, a generic conceptual biological condition gradient has been developed detailing the dominant biological responses to nutrient enrichment and decreased water clarity (Tables 1–7). These models will guide the monitoring and assist in the interpretation of results. Each time a bioregion is assessed, the conceptual models will be reviewed to ensure that all relevant and new information are incorporated.

Throughout much of South Australia there is only sparse information, if any, relating to the historical distribution and composition of shallow benthic habitats. Virtually all habitat mapping of benthic systems has been undertaken well after substantial changes (ie pre-1930s) had already occurred in the terrestrial landscape (Edyvane 1999b; DEH 2008; DEH 2009b; DEH 2009a). Changes in terrigenous habitats are known to have implications for shallow nearshore systems mostly through alteration of water runoff regimes and transport of pollutants into coastal waters, eg SE drains developments (Wear et al 2006), construction of breakout creek in Adelaide (Pattiaratchi et al 2007)). These modifications have already driven changes in the shallow benthic habitats and the opportunity to establish an accurate and well defined baseline has long since passed. Using the current state as a baseline will not accurately represent the true condition (Dayton et al 1998).

Determination of the current status of any one location is predicated upon an understanding of its pre-disturbance state. For example, investigations into the status of reefs on the Adelaide metropolitan coast since the mid-1990s (Cheshire et al 1998; Cheshire and Westphalen 2000) identified that systems immediately adjacent to urban development were likely to be degraded, although this interpretation was based on the status of less urbanised reefs to the north and south.

A coarse baseline status reconstruction will be undertaken for each location, including each of the major systems (reef, seagrass and bare sand). This information will parameterise the conceptual models developed for each bioregion to define change over time and current condition. Methods to be utilised in the reconstruction process will largely follow those used by (Bryars and Rowling 2008) which took into account the available literature for each sampling area and determined whether there was any available information on historical benthic habitats for that location. For example, if *Posidonia* or *Amphibolis* seagrass were present in the sampling area then it is likely that these were historically (ie pre-European settlement) present. As used by Bryars and Rowling, 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. The variables that will be considered will include (but are not limited to): wave energy, sediment particle size, mean and maximum current speed, light availability and the presence of stressors that may have contributed to loss. This evaluation will be described in the assessment reports for each bioregion and is likely to be flagged for future research (Tier 3).

Connell et al (2008) used reconstruction methods to investigate historical composition of rocky reefs along the Adelaide metropolitan coast which demonstrated degradation over long time periods prior to structured reef condition monitoring.

4.1.1 Developing a conceptual condition gradient

Ecological condition is not easily measured, and there are numerous factors that could correlate to condition; these assessments can vary depending on the spatial and temporal scales that are investigated. Cheshire et al (1998) suggest that one method for explaining ecological condition is by defining the attributes of a 'baseline' or unimpacted state, and that the level or severity of disturbance can be judged against deviation from that baseline state. This approach has been adopted here where the program was trialled at a number of locations that were considered far removed from current nutrient and sediment risk factors.

To determine a condition gradient for South Australia's nearshore marine waters an assessment of reference condition was undertaken (Appendix 1 Assessment of reference condition). The reference condition assessment used underwater video and water samples to survey condition of habitats and take a snapshot of water chemistry parameters in areas with very few known current anthropogenic inputs (Appendix 1 has a full descriptions of methods). The first survey was undertaken in 2009 at Flinders and Pearson Islands, approximately 17 and 35 nautical miles from Elliston off the Eyre Peninsula. The second survey was undertaken in 2010 at the Sir Joseph Banks Group of Islands within Spencer Gulf. All of these locations are removed from anthropogenic inputs and have significant shallow nearshore habitats including seagrass, rocky reef and unvegetated sediments. The aims of these surveys were to:

1 Observe and document habitats in a natural or unimpacted state within the bounds of currently available sites South Australian waters where unimpacted habitats are still thought to occur. This information can then further contribute to the conceptual models and our understanding of what constitutes a 'reference condition' in shallow nearshore coastal waters.

2 Contribute to a range of indicators or condition gradients, which relate to unimpacted shallow nearshore marine environments in South Australia. These data will be supplemented by historical EPA data from sites with known impacts of nutrients and/or poor water clarity to define a range of parameters used in the MER program to describe ecological condition.

Appendix 1 details the methods undertaken to survey the sites considered to be in a reference condition and to investigate a range of ecological conditions.

Processes describing the degradation of the common South Australian nearshore marine habitats are well documented (Section 2) including, among others:

- Adelaide Coastal Waters Study (Westphalen et al 2004; Bryars et al 2006a; Collings et al 2006b; Fox et al 2007) for seagrass systems
- Reef Health monitoring program (Cheshire et al 1998; Cheshire and Westphalen 2000; Turner et al 2007; Collings et al 2008) for rocky reef systems. See also Connell et al (2008)

Aquafin CRC Risk and Response (Fernandes et al 2007a; Fernandes et al 2007b; Tanner and Volkman 2009; Tanner and Fernandes 2010)

- for unvegetated sediment habitats.

In addition to the published data, historical EPA data collected between 1997 and 2008 were interrogated to fill data gaps not addressed by the published data. Appendix 2 summarises long-term water chemistry data in areas with established seagrass loss to add further lines of evidence to describe degraded habitats.

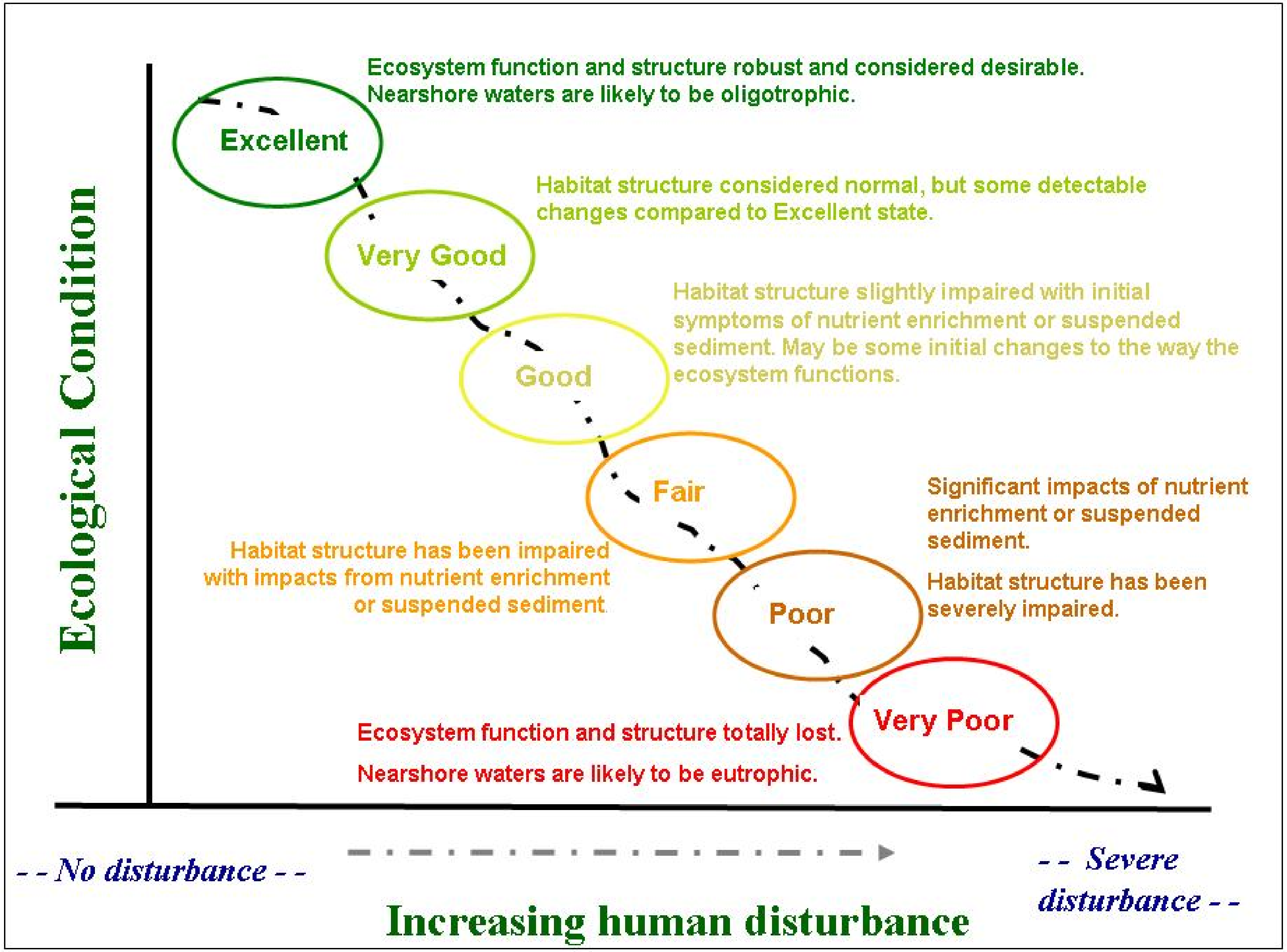

Figure 6 Ecological condition gradient for nearshore marine environments

While the documented condition of reference and degraded systems is possibly only limited to the spatial extent of those programs, generic conceptual models can be developed to allow a starting point for describing condition in each region. These generic conceptual models are outlined in Tables 1–7. As a part of this Nearshore MER program in each region, these conceptual models will be reviewed and updated for each region using information from the threat assessments and other published literature where available. This will ensure that regional scale differences are taken into account. Changes to the models will be published in the assessment report for that region which will accompany the Aquatic Ecosystem Condition Reports. The Nearshore MER program thus entrains a process of self-improvement of the conceptual models at increasingly finer scales, depending on the level of risk.

The conceptual diagrams described in this section are considered generic and apply to the way that the information is collected to generate an aquatic ecosystem condition report (AECR). The description of seagrass systems apply in areas that are considered to be suitable for seagrass growth and it is accepted that they may be areas that are not suitable for dense continuous meadows of *Posidonia* sp or *Amphibolis* sp seagrass. This will be taken into account when evaluating the condition of that site (see section 6.1.2). Similarly the conceptual model for rocky reefs is also generic and applies to rocky reefs that are similar to the model proposed by Shepherd and Sprigg (1976) and in Turner et al (2007). It is accepted that these models are conceptual only, will not apply in all circumstances, but be used as a guide and assessed for each biounit that is evaluated to determine their suitability.

Table 1 Generic conceptual model of disturbance gradient – seagrass habitats

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. Seagrass condition may start to become more variable.	Moderate density or dense patches of seagrass.	Moderate density of seagrass in frequent patches or uniform sparse seagrass coverage.	Seagrass would typically be sparse and patchy.	Seagrass totally lost where previously inhabited.
Seagrass condition (habitat structure index or HSI)	>90	Habitat structure reducing and/or becoming more variable 70–89	From 50–69	From 30–49	From 10–29	Below 10

Table 2 Generic conceptual model of disturbance gradient – rocky reef habitats

Component	Excellent	Very Good	Good	Fair	Poor	Very Poor
Robust brown macroalgal cover (Fucales & Laminariales)	Greater than 40%			Below 40%		
Turfing algae	Less than 25%			Greater than 25%		
Bare substrate	Less than 20%			Greater than 20%		

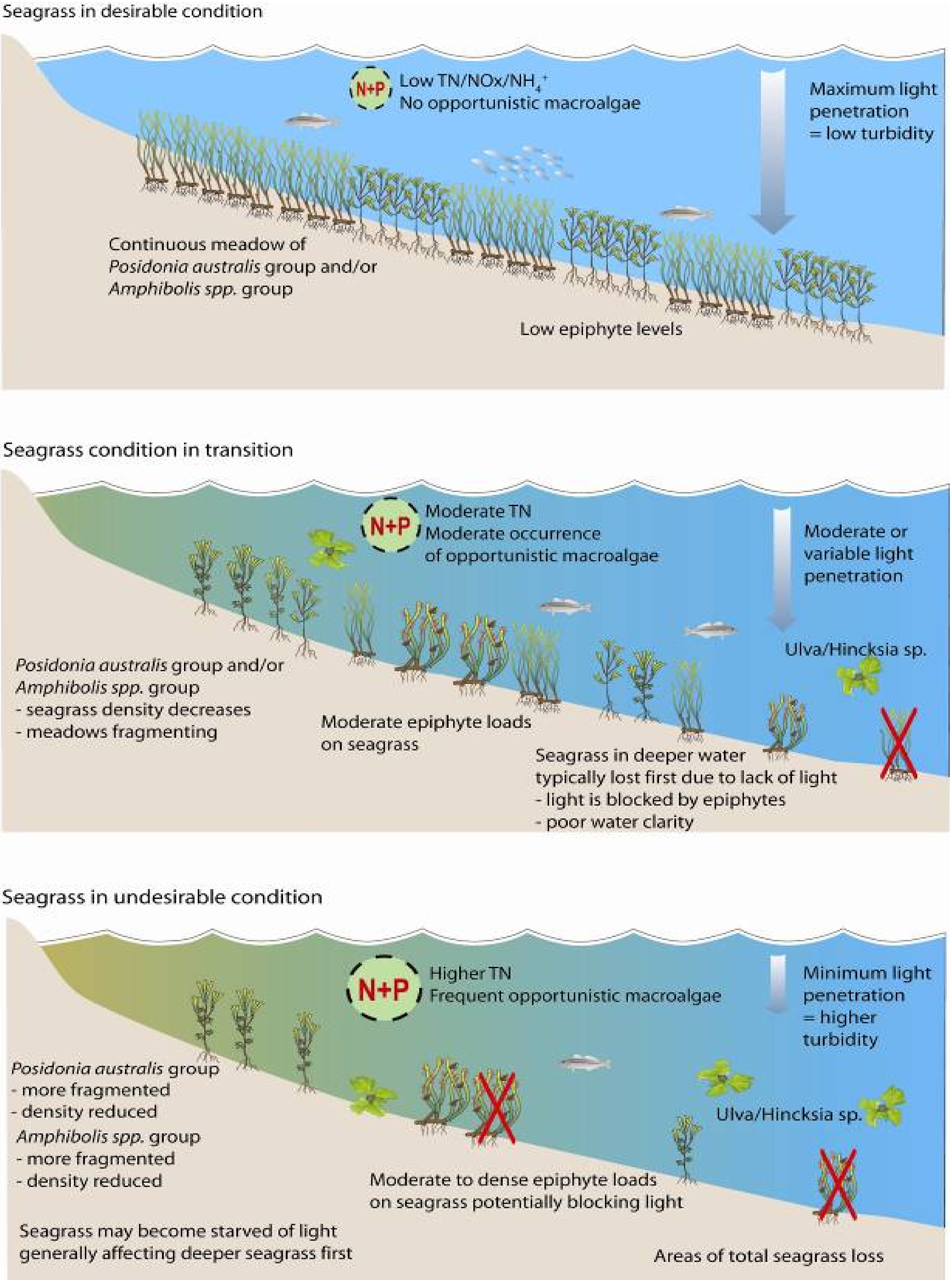

Figure 7 **Conceptual diagram for increasing nutrients and/or decreased light penetration and its effects on seagrass meadows in South Australia within this MER program. Please refer to section 4.5 Assumptions of conceptual models.**

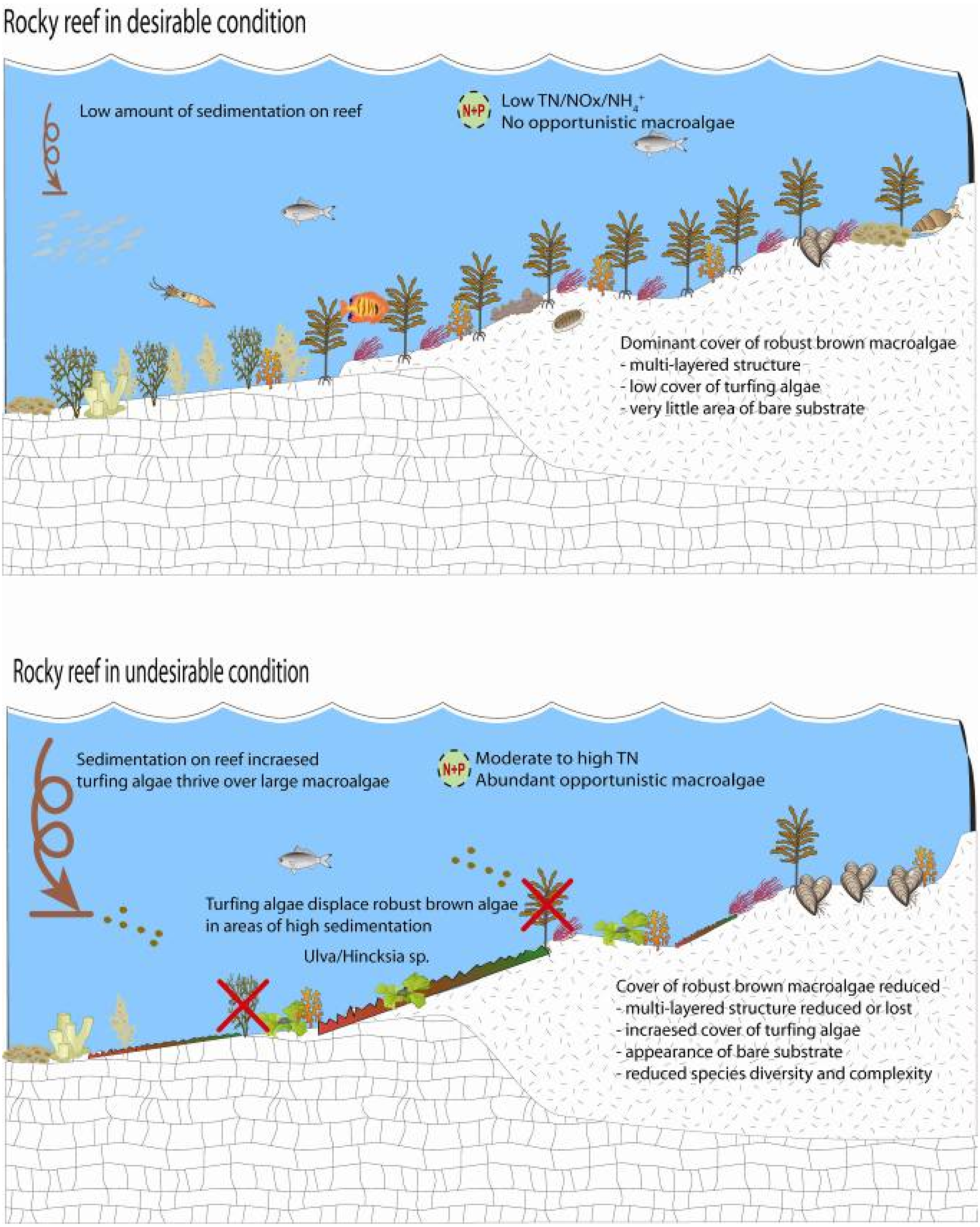

Figure 8 **Conceptual diagram for increasing nutrients and/or decreased light penetration and its effects on shallow rocky reefs in South Australia within this MER program. Please refer to section 4.5 Assumptions of conceptual models.**

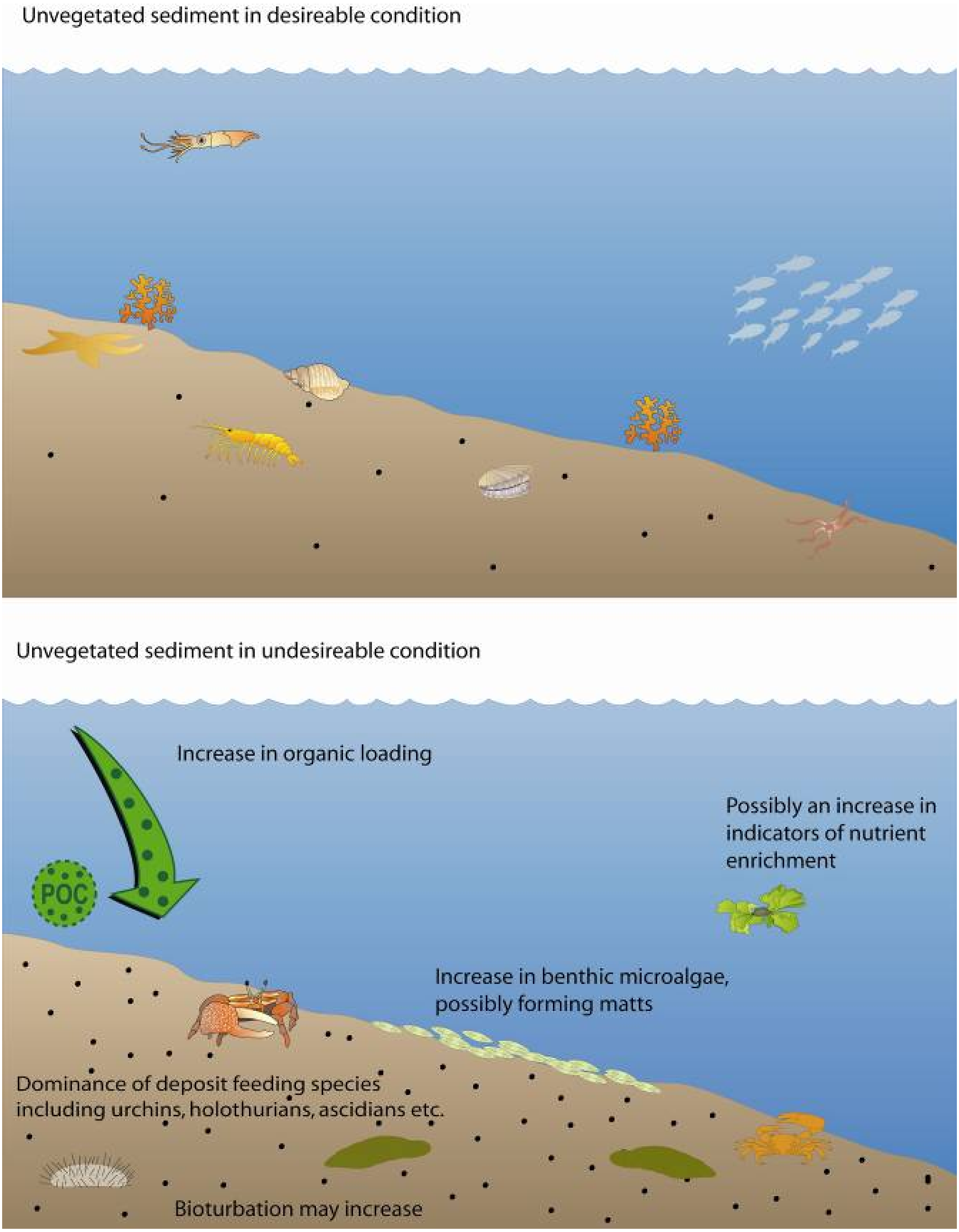

Figure 9 **Conceptual diagram for increasing organic loading and/or nutrient enrichment of unvegetated sediments in shallow nearshore waters in South Australia within this MER program. Please refer to section 4.5 Assumptions of conceptual models.**

The conceptual model for unvegetated sediments could be significantly more complex by including the well-established changes that occur in the infaunal communities in response to increases in organic loading (Brown et al 1987; Cheshire 1996). However, this type of sampling is both highly variable as well as expensive and time consuming so it does not fit within the scope of a rapid assessment MER program. If the need for this work is established, sampling may be incorporated into a Tier 3 assessment (section 3.2). The indicators available for video analysis of unvegetated sediments along a disturbance gradient are generally established for heavily impacted areas where there is a steep organic enrichment gradient (Brown et al 1987; Cheshire 1996). At this point the Nearshore MER program is limited to video analysis and the findings may be limited to gross indicators of organic loading in unvegetated sediments. If more detailed methods for rapid assessment of unvegetated sediments become available and a clear disturbance gradient can be defined, they may be trialed and could be included in the future (eg sediment redox and overlying water oxygen saturation).

Table 3 Generic conceptual model of disturbance gradient – unvegetated sediment habitats

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.	

Table 4 Conclusions based on the conceptual model of disturbance gradient

Component	Excellent	Very Good	Good	Fair	Poor	Very Poor
Habitat	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 areas and are	Habitat structure slightly impaired with initial symptoms of nutrient enrichment and/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 which has inferred impacts on resilience biodiversity, productivity and sediment stability.	Habitat structure has been severely impaired suggesting significant changes to ecosystem function including resilience, biodiversity, productivity and sediment stability. Significant impacts of nutrient enrichment or suspended sediment.	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.

Component	Excellent	Very Good	Good	Fair	Poor	Very Poor
		likely to be quickly reversible.		Detrimental effects may extend to numerous sites or small areas where longer term recovery is required.	Detrimental effects may extend to numerous sites and possibly long term recovery.	

The above conceptual models detail the habitat and the processes that change with increasing disturbance from common stressors defined in Section 4. This habitat degradation has been shown to have significant impact on ecosystem services (Walker and McComb 1992; Costanza et al 1997; O'Hara 2001; Tilman et al 2001; Waycott et al 2009) which may affect all South Australians.

The following conceptual models look at the aspects or processes that can contribute to the degradation of the seagrass, reef and unvegetated sediment habitats but could be seen as seasonal or episodic but contribute to the stress on the habitat (Kendrick and Burt 1997). The duration of their influence is critical in many circumstances (eg seagrass epiphytes) and this may have a critical effect on the condition of the habitat over time (Neverauskas 1988; Shepherd et al 1989b). These habitat 'modifiers' can be transient or the habitat may be resilient to tolerate the presence of small amounts or short duration of their effects. These conceptual models describe the change in the modifier with increasing disturbance. As the MER program is only able to look at two snapshots throughout the year, there is uncertainty as to whether these modifiers are actually degrading the habitat condition or are transient. These parameters are important to understand, however will not be included in the calculation of the AECR rating as the scale of observation is not necessarily commensurate with their influence on the condition of the habitat. As such, results will be included in the detailed discussion for each site and biounit within the assessment report. They can also be used to suggest whether the habitat is currently under stress and whether the habitat condition may change over time if that stress continues.

Table 5 Conceptual model – biological modifiers along a nutrient enrichment gradient

	Excellent	Very Good	Good	Fair	Poor	Very Poor
Seagrass Epiphytes	Very low or dominance of calcareous epiphytes on seagrass. Natural epiphyte load.	Epiphyte loads increasing and/or becoming variable Sparse classification of epiphytes.	Emergence of filamentous epiphytes on seagrass Epiphytes measured as moderate at some sites.	Epiphytes potentially contributing to reduction in light availability to seagrasses near sources or at outer depth limit. Epiphyte growth considered moderate across the region.	Seagrass likely to become impacted by reductions in light Seagrass covered in dense epiphyte loads throughout the biounit.	High potential for reductions in seagrass area due to continual reductions in light Dense seagrass epiphyte loads likely to continue throughout the year

	Excellent	Very Good	Good	Fair	Poor	Very Poor
Opportunistic macroalgae	Rare occurrences of opportunistic macroalgae.		Sparse amount of opportunistic algae (Ulva, Hincksia, etc) at some sites in the biounit.	Sparse amount of opportunistic algae (Ulva, Hincksia, etc) throughout the biounit.	Frequency of opportunistic algae considered moderate.	Dominance of opportunistic algae potentially smothering any remaining habitats.
Bacterial mat	No visible algal or bacterial mat on soft unvegetated substrates.			Sparse occurrence of algal or bacterial mats on soft unvegetated substrates.	Moderate amount of algal or bacterial mats on soft unvegetated substrates.	Dominance of algal or bacterial mats on soft unvegetated substrates.

In addition to biological indicators of nutrient enrichment, the Nearshore MER program will also undertake sampling to determine water chemistry parameters at the time of sampling. Assessing instantaneous nutrient concentrations for most forms of soluble and total nutrient (nitrogen and phosphorus) as well as a measure of phytoplankton (via chlorophyll) can provide insight into the nutrient dynamics at the site and provide information into biological activity and how the nutrients may be taken up at the site. Admittedly water nutrient concentrations are typically variable and not an integrated measure over time. Therefore their value is not as high as the measures that biological indicators provide.

However inorganic nutrients provide a snapshot of the nutrient immediately available for biological uptake and in conjunction with biological information a clearer picture of the nutrient enrichment status can be defined. In oligotrophic waters such as South Australia, inorganic nutrients (especially nitrogen) are typically very low and in many cases below the detection limit of analytical laboratories (Collings et al 2006a; Burkholder et al 2007; Pattiaratchi 2007). In these cases the use of total nutrients, and in particular the total nitrogen to total phosphorus (TN:TP) ratio can provide details about the nutrient dynamics at the site, such as whether the system is nitrogen limited at the time of sampling, and has been shown in many cases can be a better indicator of nutrient limitation than inorganic nutrients especially in oligotrophic waters (Downing 1997; Guildford and Hecky 2000; Smith 2006).

Table 6 Conceptual model – water chemistry modifiers along a nutrient enrichment gradient

<table>
<tr><th>Component</th><th>Excellent</th><th>Very Good</th><th>Good</th><th>Fair</th><th>Poor</th><th>Very Poor</th></tr>
<tr><td>Chlorophyll</td><td colspan="2">Very low chlorophyll a, typically lower than 0.59 ug/L</td><td colspan="2" rowspan="2">Chlorophyll a and total nitrogen are likely first become more variable than unimpacted sites and will increase above natural variation with the level of disturbance. Sites will be assessed using multivariate and univariate statistics to investigate similarities between sites and biounits.</td><td colspan="2">Chlorophyll a may be in the order of 2.10 µg/L or higher</td></tr>
<tr><td>Total nitrogen</td><td colspan="2">Very low total nitrogen, typically lower than 0.15 mg/L</td><td colspan="2">Total nitrogen may be in the order of 0.280 mg/L or higher</td></tr>
</table>

Table 7 Conceptual model water chemistry modifiers along a water clarity gradient

Component	Excellent	Very Good	Good	Fair	Poor	Very Poor
Water clarity	Very high water clarity typical turbidity <0.16 NTU. There may still be occasional turbidity events but the duration is likely to be very short (ie days)		Turbidity will first become more variable and is likely to increase as the level of disturbance increases. Sites will be assessed using multivariate and univariate statistics to investigate similarities between sites and biounits.		Turbidity measurements in locations impacted by poor water clarity may be in the order of 1.57 NTU or higher. The duration of poor clarity is a key factor in environmental impact. In poor and very poor areas the duration of poor water clarity is likely to be in excess of days to weeks.	

4.2 Multiple lines of evidence

Fairweather 1999 stated that no single measure or group of measures will provide the perfect view of the stability and resilience of an ecosystem, and inconsistencies may regularly confound condition or 'health' assessments. Therefore it has been recommended that a range of indicators (ie multiple lines of evidence) at different trophic levels and physico-chemical interactions should be evaluated simultaneously, and possibly iteratively refined for particular situations or locations (Amir and Hyman 1993; Cairns et al 1993). It is envisaged that by utilising a monitoring program that investigates multiple lines of evidence it may allow the differentiation between natural variability, natural disturbance and human-driven changes to habitats.

Cloern 2001 describes how the recognition of coastal eutrophication is still relatively recent, and how the early work (circa 1960–90) was influenced by limnologists' understanding of the early nutrient enrichment model in freshwater systems. The freshwater model was a simple signal–response function whereby some signal or measure of nutrient availability, typically phosphorus in freshwater systems, was correlated with a response such as a change in the biomass of phytoplankton (Cloern 2001).

As work on marine eutrophication progressed throughout the world during the 1990–2000s, the weight of evidence suggested that this simple signal–response model did not apply in coastal systems anywhere near as well as it does in freshwater systems. The Cloern (2001) review indicates that the most important direct response to nutrient enrichment in coastal systems is probably changes in the balance of selective forces that dictate the plant communities. Fast growing algae that are better adapted to high nutrient environments will increase at the expense of slow growing perennial species that are best adapted to oligotrophic environments (Cloern 2001). The conceptual model for eutrophication in marine systems therefore needs to be expanded to incorporate not just phytoplankton but multiple responses to the input of nutrients including, among other things, changes in water transparency, distribution of vascular plants and macroalgae, sediment biogeochemistry and nutrient cycling.

In Section 2 some of the natural and anthropogenic mechanisms where biological disturbance of habitats can occur were discussed. A key part in any monitoring program is to differentiate between natural variation and disturbance from a myriad of different input sources (Underwood 1992). Nearshore marine environments can be exposed to multiple stressors can affect the condition of habitats. Potential stressors can include nutrients, turbidity, toxicants, physical disturbance, temperature changes, altered hydrologic regimes and hypoxia, which can all cause changes to ecological condition as single, cumulative or synergistic processes. Responses of an ecosystem will be the integrated result of the direct and indirect exposure of all of the stressors which will be manifested in changes to abundance, diversity and resilience of organisms, communities and ecosystems (Adams 2005).

The conceptual models presented in Tables 1–7 outline the processes or stages in a phase shift that are most likely to indicate change in condition for two of the most common habitats found in the nearshore waters of South Australia. The parameters selected to be monitored as a part of the Tier 2 program will be designed to detect the different stages in the conceptual models and assess the condition using the ecological condition gradient (Figure 6).

4.3 Frequency of monitoring

The frequency of monitoring throughout the Nearshore MER program is based on the risk to the receiving environment as well as the resources available. Any monitoring program will try to balance the number of sites sampled and the number of times that the same site will be monitored over time and this is often dictated by the resources available; the Nearshore MER program is no different. The onus of the Nearshore MER program is orientated towards high spatial replication over a bioregion rather than a lower number of sites monitored at a higher frequency.

The sampling frequency may be increased if the level of risk significantly changes, but it is likely that additional monitoring in a bioregion would be investigatory in nature (Tier 3) with specific questions arising from the Tier 2 survey.

4.4 Assumptions of the conceptual models

Significant population growth in coastal regions and the associated intensification of land- and marine-based activities is estimated to heavily impact 91% of all inhabited coasts around the world in the next 40 years, and will contribute more than 80% of the world's pollution into the marine environment (Nellemann et al 2008). This pollution will cause changes to the water chemistry, which in turn will alter habitats and their ecological function (Vitousek et al 1997). With this in mind, the Nearshore MER program considers that ecological condition in the nearshore marine environment deteriorates as the degree of human disturbance in the adjacent environment increases, and conversely, the best condition occurs where there is little to no human disturbance of the environment (Figure 6).

Monitoring will focus on nearshore environments in a zone of 2–15 m depth, which has been identified as a balance between achieving sensible coverage relative to logistic constraints:

- This zone includes ecosystems in likely proximity to land-based pollution sources (Connell 2007b), a feature firmly established in both the Adelaide Coastal Waters Study into seagrass decline and the Reef Health investigation (Westphalen et al 2004; Turner et al 2007).
- The ecosystems within this zone generally comprise rocky reef, seagrass and/or bare sandy bottom for which there is a considerable and growing body of research related to their responses to disturbance, in particular declines in water quality.
- The 2-m limit allows ready (and safe) access for boats and camera systems without the need to radically adjust the sampling methods, the simplicity and consistency of which is essential in establishing a long-term, spatially varied dataset.
- Although light attenuation from shallow to deep water and distance from shore can vary substantially (see (Duarte 1991; Masini et al 1995; Collings et al 2006b), the 15-m lower limit is less likely to be light limited particularly in offshore areas of the gulfs and oceanic regions. The 15-m profile is considered to be the lower depth limit for many *Posidonia* species but certainly encompasses most of the perennial taxa for which seagrass loss is of primary concern (see summary in Westphalen et al 2004). Seagrass decline is often considered to occur in deeper areas first (Westphalen et al 2004, Collings et al 2006). A loss of condition or cover of seagrasses at depth may be the first sign of decline, although it is worth noting that on the Adelaide metropolitan coast, seagrass loss was largely in shallower nearshore areas.
- The 15-m depth places a logistic limit on sampling across seabed with a low level of slope. Conversely, this depth also offers some capacity for spatial coverage in steeper, higher relief systems.
- Assessment of seagrass condition assumes that the seagrass in a pristine state would grow in continuous meadows. This is considered reasonable for *Amphibolis* spp and the *Posidonia australis* group (*Posidonia australis*, *P sinuosa* and *P angustifolia*), however there is insufficient evidence to suggest that seagrasses from the group *Posidonia ostenfeldii* (*P coriacea, P kirkmanii* and *P dehartogii*[1]), live in similarly continuous meadows (Kuo and Cambridge 1984).

[1] The *Posidonia ostenfeldii* group also comprises of an additional two species; *P ostenfeldii* and *P robertsoniae* however these species have not been recorded in South Australia.

The Nearshore MER program will use the available information to determine the risk factors in the Tier 1 assessment and attempt reconstruction of a baseline condition for each Bioregion. All available scientific work will be taken into account in order to aid in the interpretation of the results and the application of the conceptual models in each Bioregion. Interpretation of results will pay particular attention to hydrodynamic modelling, work detailing nutrient sources such as industry monitoring programs, as well as historical monitoring data. If available, previous years' AECR monitoring will be utilised to ensure that the most up to date information is used.

Each time a region is monitored the understanding of the environment will be enhanced, which in turn increases the available information to be considered for that region. In addition to the Nearshore MER work, it is likely that other institutions and research organisations will be undertaking similar or related work which may also add valuable information which can be considered. For these reasons the Nearshore MER program is iterative and increased understanding of the marine environment may result in changes to the program as it is undertaken. If changes do occur then these will be documented throughout the program.

5 Tier 1 desktop assessment and expected condition

5.1 Threat assessment

In order to tailor the monitoring program to cater for specific threats in a region, an assessment of threats to ecological condition within each bioregion will be carried out. Where possible the threat assessment will use or adapt existing risk assessments carried out by regional agencies (notably NRM Boards). The threat assessment for each bioregion will accompany the aquatic ecosystem condition reports (AECR) in the assessment report.

Numerous agencies already undertake risk and threat assessments and this MER program will try not to replicate existing work. However where suitable assessments are not available, a simple and coarse assessment of threats to the nearshore environment will be undertaken. This threat assessment enumerates potential threats by type (eg sea cage aquaculture) and broadly scales the consequence of this activity in the biounit based on the potential consequence after all control measures are in place (residual risk).

This assessment takes into account the presence of a pollution source (or transport of pollution) in the region (likelihood) and the size and scale of the possible impact (consequence) as well as the number of potential threats in each region (severity or scale). This process generates a coarse assessment of the number of threats and can be used to tailor the monitoring program and develop a prediction of the condition for each bioregion. This will be undertaken prior to the Tier 2 monitoring so that information gained in this process can feed into the planning stages of the field program. A key part of this process is liaison with local councils, NRM Boards and other agencies to gain the most information about key pressures in the area. This will better inform the threat assessment process using the most up to date and local information.

Within each biounit the existing information will be used to determine whether the application of the generic conceptual models is suitable. This will be a coarse assessment based on the types of habitats present and whether the assumptions of the generic conceptual models are still valid for each biounit. If there are atypical habitats that do not fit the conceptual models these will have to be considered on a site-by-site basis and the conceptual models refined for future use in that biounit. In the event that there is no previous biological information on the types of habitats are available prior to the Tier 2 program for that biounit, the generic conceptual models will be assumed to be reasonable for the purpose of site selection and developing a predicted condition. The validity of this assessment will be reviewed on analysis of the collected biological information in the Tier 2 assessment.

5.2 Predicted condition

The predicted condition is developed through the results of the threat assessment. This takes all the available information to predict the likely condition of the biounit on the condition gradient. The aim of this predicted (or expected) condition is to demonstrate that many of these areas have been exposed to disturbance for over 100 years and we may not predict (or expect) that the biounit will be in pristine condition.

This predicted condition sets a reference point to compare the observed condition based on the level of inputs, and while the biounit is still compared against what we consider to be reference condition we may not expect many biounits to be in reference condition. Disparity between the predicted condition and the observed condition may highlight deficiencies in our understanding of some of the ecological processes in that biounit, which may require further work (ie Tier 3) or closer investigation of the threat assessment.

6 Tier 2 monitoring and evaluation methods

6.1 Monitoring methods

6.1.1 Key measures

There are a number of fundamental aspects that drive key processes in the nearshore marine environment. A threat assessment process has been carried out which demonstrated that the main threats to nearshore habitats along Adelaide’s metropolitan coast (and wider Gulf St Vincent) were from nutrients and sediments causing either poor water clarity or sedimentation (Westphalen et al 2004; Turner et al 2007; Gaylard 2009b). With this in mind the three main aspects that the Nearshore MER program will monitor are:

1 Habitat condition incorporating ‘atypical’ habitats

2 Modifying variables which help inform the state of key processes:

   – water clarity

   – nutrient enrichment.

Seagrass have carbohydrate stores which can be used for energy reserves in periods where photosynthesis does not exceed energy requirements allowing them to survive in periods of low light (Burke et al 1996; Collings et al 2006b). Therefore the duration of poor water clarity or dense epiphyte load is of critical importance to the condition and survival of the habitat. Macroalgae do not have large carbohydrate reserves and are susceptible to small reductions in light levels (Markager and Sand-Jensen 1992). While biological indicators integrate the ambient conditions the habitats are exposed to over periods of days to weeks or even months, it is possible that site-specific factors contribute to the resilience of some systems, and that may exceed the timeframe of these integrated measures.

With this in mind the two snapshots of water chemistry in the Nearshore MER are insufficient to establish the duration of stress driven by the presence of these modifiers and longer term monitoring of each site is logistically unfeasible. Only the measure of habitat condition will be used to calculate the aquatic ecosystem condition report (AECR) and the information gathered on nutrient enrichment including epiphyte load, abundance of opportunistic macroalgae, water chemistry and water clarity will be used to inform the level of stress the habitat is currently under and can be used as an early warning system to signal the potential for the habitat to change over time.

While not used in the AECR score, these additional parameters will be included in the investigation of differences at the site level within the biounit to relate to the amount of stress the habitat was under at the time of sampling and also assessed in context with the proximity to nutrient discharges identified in the threat assessment.

6.1.2 Site assessment

Many monitoring programs focus on one habitat type (eg seagrass) and will search out this habitat type to monitor its condition (Neverauskas 1987b; Wood and Lavery 2000). The Nearshore MER will assess each randomly allocated site for its condition encompassing reef, seagrass and bare sand environments between 2–15 m depth. This approach will remove bias with respect to the location of monitoring sites and will assess the region from a truly ambient viewpoint. Most biounits have large areas that are potentially deeper than 15 m, which will not be included in the assessment of habitat condition in the MER. Therefore it is stressed that this program will describe condition representative of the nearshore (2–15 m) habitats monitored in the biounit, not the entire biounit.

Underwater photo or video has been used for decades to assess epi-benthic communities throughout the world. While originally used in demanding environments (eg water too deep for SCUBA >30 m), their use has increased significantly over the last 10 years with the increase in image resolution and the reduction in size and cost of video and photo systems. Underwater video recordings are becoming increasingly used to monitor benthic habitats. While at a smaller spatial scale than satellite or aerial photography, video allows for greater quantification of detail than methods such as remote sensing and aerial photography, and additionally can investigate locations which are unsuitable for satellite or aerial techniques (Roelfsema et al 2009).

In seagrass habitats, high resolution underwater video can quantify a variety of habitat measures in a permanent record that can be re-analysed in the future. Experiments to compare data quality and results have shown that photo and video methods are comparable to traditional SCUBA-based methods for quantifying epi-fauna and hard substrata macro-benthos (Roberts et al 1994), seagrass mapping (Norris et al 1997), seagrass change (McDonald et al 2006) and reef communities (Miller et al 2003; Ramos et al 2010). Video/photo-based methods can also be more accurate (Leujak and Ormond 2007), cost effective and safer than traditional SCUBA methods. While some alternative methods can provide accurate fine scale data that are suitable for detailed scientific investigations, less detailed information over larger spatial scales can be more appropriate for management purposes (Dumas et al 2009), (Hill and Wilkinson 2004), particularly given limitations in resources. Video methods also provide the ability to accurately sample habitats without destructive techniques which add to the disturbance burden (McDonald et al 2006).Video does have some limitations with moderate winds and small swell dictating suitable monitoring to ensure good quality footage.

In order to characterise habitats over a site, 10 x 50-m underwater video belt transects were undertaken at randomly chosen locations in water between 2–15 m deep. Transects were undertaken using a geo-referenced 450-line analogue video camera (Scielex/Kongsberg) angled at 90 degrees to the seafloor, in a custom made housing. A live video feed to a surface screen viewed by a trained operator ran directly from the camera into an audio and video encoding system (Geostamp) which overlays a GPS location, direction, speed, date and time strings to the video and on a hard drive.

The surface screen and trained operator allowed the camera to be positioned approximately 1 m from the substrate in order to maximise image quality and resolution. This set-up provided a field of view of approximately 1 m^2, whereby each belt transect equates to approximately 50 m^2. Videos were analysed upon return from the field using an inhouse video analysis software package. The operating procedure is outlined in Appendix 4. At times, the resolution of the analogue video camera can appear pixilated when finer detail may be warranted. Therefore a full high definition (HD) video camera (GoPro Hero 2) was synchronised with the analogue camera and is run simultaneously with the analogue video, and when analysis demands a higher resolution for taxonomic identification or finer detail (eg rocky reef assessment), the HD footage can be used.

Quantifying water chemistry at each site was undertaken by sampling three replicate 2.5-litre water grab samples at each transect location mixed into a 25-litre container. After 3 transects the water in the container was mixed thoroughly and sub-sampled. This process is repeated across the site for 9 of the 10 transects (n = 9) to provide a snapshot of water nutrient concentration (total nitrogen, total ammonia, total kjeldahl nitrogen, total oxidised nitrogen, total phosphorus and filtered reactive phosphorus) and turbidity. This number provides simplicity and equal measures in the water sampling process rather than using an uneven number of samples to cover 10 biological transects. This is believed to not affect the results of the program.

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 a 2-litre grab sample is taken 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 for storage prior to being sent for extraction and analysis. All samples were frozen and analysed within the laboratory holding times. Compositing water samples is a method that 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). The method for compositing water samples incorporated into the Nearshore MER program is outlined in Appendix 5.

All water samples were sent to the Australian Water Quality Centre for analysis.

In the event of values being below the reporting limit (ie: below the detection limit of the analytical equipment), a method of substituting the censored value with half the reporting limit has been adopted (Ellis and Gilbert 1980). This arbitrary method does have its limitations ((Helsel 1990) but in the Nearshore MER program it was considered appropriate due to the amount of data generated, the low number of 'non-detects' and the unbiased nature of using half the reporting limit compared to methods that substitute for the reporting limit or a zero value (Helsel 1990).

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

6.1.3 Habitat condition

Seagrass

Section 2 describes the dominance of seagrass habitat throughout South Australian nearshore waters as well as its value through its role in ecosystem services, nutrient cycling and fisheries production. (Irving et al 2013) describes a methodology to quantify seagrass habitat condition integrating multiple variables of seagrass habitat structure into a dimensionless number that is scaled out of 100. This analysis has been modified from a similar method for calculating fragmentation indices within terrestrial forests developed by (Bogaert et al 2000). The index uses the species, weighted in relation to its perceived value based on succession following disturbance (Clarke and Kirkman 1989), and the area of cover to determine seagrass structure across a transect. A seagrass condition value of 100 indicates seagrass with excellent structure (and therefore excellent condition). As structure deteriorates, the index approaches zero (Irving et al 2013). The seagrass condition measures include:

1. Seagrass area along the transect – based on the field of view of 1 m^2 for a transect of 50 m resulting in a maximum possible seagrass area of 50 m^2,
2. Seagrass density – measured semi-quantitatively as 1–30%, 30–60% and 60–100% cover along the 50-m transect,
3. Seagrass species identity – to the lowest taxonomic resolution possible,
4. Continuity of seagrass meadows (uniform or patchy), and
5. Proximity of patches – distance between patches as a measure of how patchy a meadow may be.

Using the habitat structure index, transects where seagrass is totally absent but adjacent similar transects include seagrass or the baseline reconstruction process (all available evidence) suggests that seagrass could be or has been present historically, the seagrass condition will be considered to be poor. The average HSI from all transects considered suitable for seagrass growth will be used to calculate the habitat condition for the biounit as a score out of 100.

As stated in section 4.4 the (Irving et al 2013) method assumes that seagrass meadows are present in continuous meadows, however Kuo and Cambridge (1986) highlight that the *Posidonia ostenfeldii* group of seagrasses may naturally grow in clumps rather than continuous meadows due to their presence in higher wave energy environments. Using the habitat structure index as it currently stands in these environments may underestimate condition. When *Posidonia ostenfeldii* group of seagrasses are encountered, an assessment of condition will be made without assuming a continuous meadow structure.

Rocky reefs

While accepting that towed video or photographic methods for assessing macroalgal composition has limitations (Alvaro et al 2008) for the purposes of a broad assessment, some key observations of reef composition can still be concluded, such as canopy algal cover (Miller et al 2003; Alvaro et al 2008; Ramos et al 2010). Given the resolution of underwater video it is unlikely that species level identification will be possible. Reef life-form codes for macroalgae will be used as outlined in (Cheshire and Westphalen 2000) and Turner et al (2007).

Where rocky reef habitats were encountered, broad characteristics were assessed using a point intersect method involving non-random points spread over a 1-m^2 quadrat where the habitat is assessed at each point intersection (Leujak and Ormond 2007; Oh 2009). Each quadrat was assessed using a 9 by 7 grid where the points intersected result in 56 data points to generate an area of cover for each parameter recorded over the 1-m^2 quadrat.

The Reef Health Program (Turner et al 2007) defined and employed a number of condition indices for reef status. These indicies were defined for South Australian temperate reefs in shallow waters, and with caution, may be applicable here. These indices show, among others:

1. Cover of robust canopy forming macroalgae (notably Fucales and Laminariales) above 40%
2. Cover of turfing algal assemblages (< 1 cm in height) below 25%
3. Cover of bare substrate below 20%

See (Turner et al 2007; Collings et al 2008) for a full description.

Where possible replicate quadrats of the reef habitat along the transect were assessed and the average percent cover for each life-form along the reef transects was assessed and compared to the conceptual models described in Table 2. It is important to realise that macroalgal cover can change substantially depending on the season, species composition and associated reproductive preferences with many species of mostly fucoids shedding their reproductive fronds periodically (Edgar et al 2004). For this reason the assessment of the reef habitats are undertaken across a site for the year rather than individual seasons. The large brown algae included in the assessment as a maximum rather than an average to ensure that if the reef is dominated by fucoids the condition is not underestimated due to seasonal shedding of fronds. All other parameters are averaged across the year to assess the composition of the reef. The approach employed in the Nearshore MER surveys did not use all of the specific indices as calculated by Turner et al (2007) but focussed on some of the key indicators of degradation along South Australia's coast that can be assessed using a towed video.

Table 6 shows a conservative method for defining broad condition of rocky reef habitats. As stated above rocky reefs are complex systems that frequently change through space and time, and a comprehensive condition assessment is not possible with the current resources. A coarse assessment will be undertaken using criteria described above. Due to the uncertainty a conclusion of good condition will only be made when all three criteria described above are met. Similarly only a conclusion of poor condition will reflect a state when all three criteria are not met. In circumstances when results of the criteria are mixed the index value will be the mid-point (Table 8).

It should be considered that the reef indices are at an experimental stage and will be trialled through the first few years of the MER program, and that the indices may be adjusted if further information from the published literature is presented. For this reason, the reef condition index does not cover all possible permutations of criteria and there is uncertainty in the reef assessment indices.

The EPA also undertakes a reef survey program across a number of locations using the Reef Life Survey method <www.reeflifesurvey.org> and the results of this finer-detail assessment will be considered when evaluating reef indices and their relevance in areas throughout South Australia's nearshore waters.

Table 8 Reef condition index

Reef condition index	Index result
All measures above the criteria	100
Some measures meeting criteria, some not meeting criteria	50
All measures below the criteria	0
No rocky substrate present	NULL

Unvegetated soft sediment

Work defining the condition of unvegetated sediments has largely centred on detecting impacts from organic pollution, typically as a result of sea cage aquaculture. The two main methodologies used have been an assessment of epi-benthic communities using video transects or assessment of infaunal communities using sediment cores, or a combination (Cheshire 1996; Madigan et al 2001; Loo 2007).

The studies which have utilised video recordings highlight its potential for rapid assessment of epi-benthic communities. Brown et al (1987) showed a gradient of epi-benthic communities radiating away from sea cage aquaculture in Scotland. Subsequently Cheshire et al (1996) used that work to develop South Australian-specific indicators of organic enrichment from Southern Bluefin Tuna (SBT) farms in Port Lincoln. These studies showed epi-benthic species indicative of areas impacted by organic loading were higher in abundances of filter feeding or deposit feeding taxa including ascidians, holothurians, sea urchins and spider crabs *Leptomithrax gaimardii* (Cheshire 1996). These studies have also shown an impact on the epi-benthic community but typically only at short distances from the cage indicating gross effects rather than subtle far-field effects (Cheshire et al 1996). This suggests that for ambient monitoring in the Nearshore MER program, the use of epi-benthic communities may be too coarse a measure to detect condition of unvegetated sediment in all except significantly polluted sites (Madigan et al 2001).

The use of infaunal communities in sediment has been shown to detect more subtle environmental impacts when used in SBT sea cage aquaculture compared to other methods (Cheshire et al 1996, Madigan et al 2001). As organic enrichment increases, three main changes may be observed in infaunal communities (Madigan et al 2001):

- an increase in abundance of species tolerant to organic enrichment
- a decrease in taxonomic diversity
- a change in community structure.

Using these indicators, changes to benthic condition can be detected above natural variability. However, benthic infauna sampling and analysis is time consuming and very expensive for a rapid condition assessment program. The use of infaunal sampling and analysis methods would be more suitable to Tier 3 investigations (See section 3.2).

Due to the current inability to assess the condition of unvegetated soft sediment habitats in areas other than highly-impacted ones, locations composed entirely of this habitat type are not routinely assessed as part of the Nearshore MER program. Section 4.2 outline the process that was undertaken to evaluate whether a site is likely to have seagrass or reef habitats present, or whether there are likely to be natural factors that have resulted in the area being unsuitable for their colonisation. When the Tier 1 and 2 assessments concluded that seagrass or reef habitats were unlikely to have been naturally present and a transect was composed entirely of unvegetated soft sediment, then the habitat will be scored according to the broad conceptual model defined in section 4. This is where unvegetated sediments that do not have indicators or organic enrichment (or sparse presence of indicative epifauna) will be scored as 100 and those with prolific indicators of enrichment will be scored at zero and moderate densities will be scored at 50. This system is coarse and is likely to only show degradation at grossly polluted sites, but fortunately these are likely to be rare.

Mixed habitats

In the event of mixed habitats occurring along a 50-m transect length, the assessment system habitat score is normalised adjusted based on the length of observed cover relative to the length of substrate considered available for that habitat type (Figure 10). The Tier 1 and 2 assessments suggest whether an area could be, or has been, suitable for seagrass or reef habitat (eg sediment type, presence of seagrass) which is described in Section 4. If there is sand between patches of seagrass, the default position will be that seagrass could grow between these patches (assuming similar substrate). When reef is dispersed within patches of sand, the default position will not be that seagrass could not grow between the reef habitats.

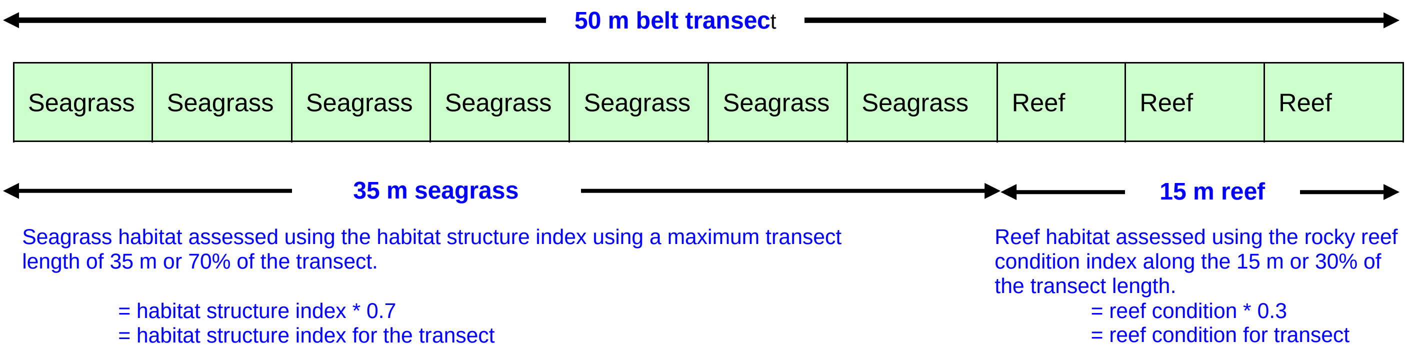

Figure 10 Schematic representation of a 50-m transect of mixed seagrass and reef habitat and the assessment methods

Atypical sites

The assessment of condition using the above systems has the potential to leave some habitats scoring poorly even though they could be perceived as being in good condition or particularly valued. An example of this issue would be a sponge garden. In order to account for this possibility, a subjective measure of value can be allocated to locations considered to be atypical but of significant value. It is likely that this will only apply to a site that has an 'atypical' habitat, and is likely to have only a small bearing on the overall habitat integrity score for that region. In this circumstance, the site (or perhaps just a replicate) can be excluded from the AECR overall assessment, but its presence as atypical will be recorded and recognised as being potentially important and in need of further research or detailed assessment.

Overall score for habitat condition

Habitat condition and therefore the AECR score is calculated as the mean score of each habitat type within the biounit which is adjusted based on the total area of each habitat type in the biounit that was monitored. This method is used to prevent bias in the scores based on a small amount of a particular habitat monitored having a large influence on the overall score.

For example if 20 sites are monitored (ie 200 transects) within the biounit and across all transects the composition is 81% seagrass, 10% reef and 9% unvegetated sediment which was considered to not be suitable for seagrass growth. The mean habitat condition scores for the biounit are HSI = 67 for seagrass, a score of 50 for rocky reef and 100 for unvegetated sediment. Without adjusting the scores based on the proportion of habitat monitored then the result would be the mean of these scores = 72.3 which according to the linear scale of the 6 categories is an AECR rating of Very Good. However the high score for unvegetated sediment is skewing the overall result as it only represents 9% of the monitored area but is included at 33% of the score.

If the habitat scores are adjusted based on the proportion of the habitat monitored then the adjusted scores would be:

- Seagrass: HSI score (67) x seagrass area monitored in biounit (0.81) = adjusted score 54.3
- Reef: Reef score (50) x reef area monitored in biounit (0.1) = adjusted score 5.0
- Unvegetated sediments: Unvegetated sediment score (100) x area of unvegetated sediment monitored in biounit (0.09) = adjusted score 9.0

Total habitat score is the mean of the three adjusted scores is 68.3 which equates to the AECR score of Good.

This example shows the potential for bias in the calculation based on equally weighting of the habitat scores rather than adjusting the scores based on the amount of that habitat type monitored in the MER program for that biounit.

The AECR communication tool score will be generated based on the above adjusted process for each biounit. Within the assessment report there will be discussion on how similar the areas monitored are compared to the area of that habitat type in the waters less than 15 m deep in the biounit as determined by habitat mapping. Included in the AECR communication tool will also be a discussion on the habitat ‘modifiers’ which may be influencing the habitat, and may also suggest that the habitat is under stress from nutrient enrichment or poor water clarity.

6.2 Habitat condition modifiers

6.2.1 Water chemistry parameters

The assessment of habitat condition largely describes the biounit according to the conceptual model of a ecological condition gradient. The nutrient variables, including chemical and biological parameters, as well as water clarity, build additional lines of evidence to highlight potential stressors on the system. The nutrient and water clarity variables are largely explanatory and help to describe the processes or stages in the conceptual model.

A system to scale the water chemistry variables based on where the data sits in relation to observations at reference areas and impacted areas will be used to determine the risk of habitat degradation. This system uses statistical end points based on the ANZECC Guidelines for Fresh and Marine Water Quality (2000) but does not use the default trigger values contained within ANZECC due to significant evidence that the trigger values for nutrients are too high to be protective of habitats throughout the oligotrophic waters of South Australia (Gaylard 2005; Gaylard 2009a; Gaylard 2009b).

The document describes developing site-specific trigger values to determine whether management goals (ie protection of aquatic ecosystems) are being met or whether further work is warranted. The ANZECC approach uses a comparison of the median of a population of data compared to the 80th percentile of a reference population. For the Nearshore MER program the data collected from the Sir Joseph Banks group of islands and offshore islands including Flinders and Pearson islands have been used as the reference population for this comparison. It is accepted that these locations are not ideal but this is considered a good starting point for further refinement when the program matures and more nearshore locations around the state are monitored. The 80th percentile of reference populations has been considered to represent slightly to moderately disturbed waters (ANZECC/ARMCANZ 2000).

To further inform the range of potential water chemistry, locations with historical water quality data related to ongoing impacts have been interrogated. These locations were the jetties at Largs Bay, Semaphore, Grange, Henley Beach, Glenelg and Brighton using historical EPA data set from 1998–2008. Multiple lines of evidence suggests that these locations have impacted ecological condition, with elevated nutrients and/or chlorophyll *a* (Gaylard 2004), seagrass loss (Fox et al 2007; Cameron 2008) and/or degraded rocky reefs (Turner et al 2007; Collings et al 2008).

The Nearshore MER data will be compared to the median value of the data from the above locations to determine whether they may be at risk of impacts from nutrient enrichment or poor water clarity (Table 9). In the event that anomalous or unexpected results are revealed, then further investigatory monitoring may be triggered (ie Tier 3 monitoring).

While not included in the AECR rating for the biounit the water chemistry provides valuable insight into the nutrient dynamics and water clarity at the time of sampling and will be discussed in detail, including using multivariate and univariate statistics for comparisons, within the assessment report for each biounit. Over time it is possible that the scale for each variable could be adapted for each specific biounit to develop site-specific trigger values according to the process outlined in the ANZECC Guidelines for Fresh and Marine Water Quality. This would provide a more realistic indication of water quality and nutrient enrichment as it is likely that changes in nutrient status of a region that may elucidate biological responses are likely to be relative changes over time and less importance would be placed on a comparison to a reference state. Any changes would be identified in the assessment report for each bioregion.

The method for interpreting each variable used in the Nearshore MER program is outlined in Table 9.

Table 9 Scaling system for water chemistry, chlorophyll *a* & turbidity variables

Median biounit variable	Description	Likely result
Less than the median of the reference sites	The parameter throughout the Biounit is similar to the reference sites.	Potentially few impacts from that parameter measurable in the biota throughout the biounit
Higher than the median but less than the 80th percentile of the reference sites	Recommended by ANZECC (2000) as slightly- to moderately-disturbed.	Likely to be measurable changes in the habitats due to prolonged elevated nutrient or turbidity concentrations changing ecological processes throughout the biounit
Greater than the median value of areas observed to result in habitat degradation	The parameter is similar to what has been described as impacted	Likely to be significant changes to the habitat throughout he biounit as a result of prolonged elevated nutrient or turbidity concentrations

Total nitrogen

Nitrogen is a key (and generally productivity limiting) nutrient for plants and algae in the marine environment (Section 2). Nitrogen comes in many forms but generally only the soluble states (ammonia and oxidised nitrogen) are readily available to biological systems. However, under certain conditions, bacteria can breakdown the organic forms of fixed nitrogen and produce ammonia which can be reintroduced into the water column in a bioavailable form. This cycling of nutrients is important in the overall budget for an ecosystem and can contribute significant loads into the water column, even if the nutrients enter the system as relatively 'unavailable' organic nitrogen.

The assessment of reference condition (Appendix 1) indicated a median total nitrogen concentration of 0.15 mg/L and an 80th percentile of 0.228 mg/L (Table 8). Historical monitoring by the EPA from 1998–2008 across sites with known impacts due to nutrient enrichment, resulted in a median of 0.285 mg/L (Appendix 2).

The Nearshore MER program will classify total nitrogen into one of four levels (Table 10).

Table 10 Total nitrogen scale

Median total nitrogen (mg/L)	Likely result
Less than 0.150	No different to reference locations
Between 0.151 and 0.228	Slightly to moderately disturbed
Above 0.286	No different to locations with impacts likely to be due to nutrient enrichment

Other species of nitrogen are recorded and evaluated during the assessment. However, their inherently high variability in coastal waters and the uptake of dissolved inorganic nitrogen (DIN) by seagrass and other indicators of nutrient enrichment, eg epiphyte, chlorophyll or macroalgal abundance (Apostolaki et al 2012) means that while the links between ambient soluble nutrient concentration and biological response is often very well understood, it is hard to define in the field except under strict experimental conditions. In these cases the use of total nutrients can provide some details about the nutrient dynamics at the site. Other metrics including the total nitrogen to total phosphorus (TN:TP) ratio can also provide useful information, such as whether the system is nitrogen limited at the time of sampling. Investigating multiple metrics including total nutrients in waters has been shown to be a better indicator of nutrient limitation than inorganic nutrients especially in oligotrophic waters (Downing 1997; Guildford and Hecky 2000; Smith 2006).

Open oceans typically fit a well understood molar elemental ratio of 106:16:1 for carbon, nitrogen and phosphorus respectively (Redfield 1958). Marine phytoplankton typically also conform to this molar ratio, and in waters that are nitrogen limited the ratio of N:P will be less than 16:1, while in waters not nitrogen limited the ratio can be higher.

In this MER program the use of the N:P ratio can be used as an indicator helping to explain the nutrient dynamics, but importantly it should be noted that biological activity including epiphyte biomass and opportunistic macroalgae can assimilate nitrogen resulting in a low N:P ratio. However these other lines of evidence show that there is nutrient enrichment occurring (see sections 4.2 and 6.2).

Work on nutrient thresholds for seagrass communities is currently underway and may aid in the determination of criteria for soluble nutrient species in the future.

Water clarity

Light is fundamental to primary producers because it facilitates the photosynthetic process. Seagrasses and algae rely on access to sufficient light to generate energy production, and therefore growth and reproduction (Short and Wylie-Echeverria 1996, Zimmerman 2006). There is evidence to suggest that at times the light penetration through the water column can be hindered by poor water clarity which can include elevated suspended solids and coloured dissolved organic matter. Such events can affect photosynthetic organisms by reducing the time available for photosynthesis (Ralph et al 2006). When the amount of time spent respiring (using energy) exceeds the time photosynthetic organisms can produce energy (photosynthesising) the organism will go into energy deficit, and when the time in deficit is prolonged, seagrasses and algae can die of starvation (Ralph et al 2006).

Sections 2.1 and 2.2 discuss the way in which limited access to sufficient light can cause ecological impairment in seagrass and rocky reef habitats, and the conceptual models in Tables 1–4 demonstrate the model being evaluated in the Nearshore MER program.

Turbidity

Turbidity is a measure of water clarity which assesses the amount of scattering of light in the water column. It is accepted that turbidity varies over relatively small temporal scales, and that two spatially intensive snapshots (autumn and spring) only provide information on the time of sampling rather than any period between sampling and will not capture temporal variability. Collection of water clarity data is still useful to show whether the biounit receives significant terrestrial runoff or experiences sediment re-suspension which may affect light penetration. As stated in Section 4 the duration of this decreased light penetration is critical in determining whether there is a significant risk of impact.

Turbidity data were assessed at reference locations and the 80th percentile for all reference sites was 0.278 NTU and the median was 0.191 NTU (Table 11 and Appendix 1). The median turbidity concentration from sites with possible impacts from poor water clarity from the historical water quality monitoring from 1998–2008 was 1.57 NTU.

Table 11 Turbidity index

Median turbidity (NTU)	Likely result
Less than 0.191	No different to reference locations
Between 0.192 & 0.278	Slightly to moderately disturbed
Greater than 1.57	No different to locations with impacts likely to be due to nutrient enrichment

6.2.2 Biological parameters

Chlorophyll a

Chlorophyll *a* is used as a measure of phytoplankton biomass in the water column. Chlorophyll complements chemical nutrient monitoring as algal growth may be more representative of the immediate-past nutrient status of the environment than periodic sampling of ambient nutrient concentrations. There are a number of instances where nutrient levels are very low but the site has significant evidence of eutrophication because available nutrients have been converted to algal growth (Cloern 2001; Apostolaki et al 2012). If nutrients were sampled alone this impact may not become apparent. However, measuring phytoplankton chlorophyll does not take into account macroalgae, which can be a significant part of the ecosystem, especially when considering the impacts of eutrophication.

There may be a need for regional specific scales to account for natural elevations in chlorophyll due to non-anthropogenic factors such as upwellings, eg Coffin Bay and the South East (Schahinger 1987; Pattiaratchi 2007), but this will need to be assessed in conjunction with other data collected in the Nearshore MER (and other) monitoring programs, eg SAIMOS and will be documented in the assessment reports.

The assessment of reference condition (Appendix 1) indicated that the median chlorophyll *a* value was 0.591 µg/L and the 80th percentile was 0.719 µg/L (Table 12). Analysis of historical monitoring undertaken by the EPA from 1998–2008 at locations with known impacts from nutrient enrichment resulted in a median chlorophyll *a* concentration of 2.09 µg/L (Appendix 2).

The median chlorophyll *a* for each biounit will be compared to the values outlined in Table 9.

Table 12 Chlorophyll *a* index

Median chlorophyll *a* (µg/L)	Likely result
Less than 0.591	No different to reference locations
Between 0.592 & 0.719	Slightly to moderately disturbed
Above 2.10	No different to locations with impacts likely to be due to nutrient enrichment

Epiphyte load

Space is at a premium in the marine environment (Connell and Keough 1985; Butler and Chesson 1990). Organisms will attempt to grow on any suitable substrate. In oligotrophic environments, seagrasses generally have some growth of epiphytic organisms on leaves. In mesotrophic (or eutrophic) environments the amount of epiphytic organisms increases to a point where they can shade seagrass and weigh the leaf down resulting in it breaking off. Seagrass epiphyte load can also depend on the species of seagrass and is also often related to the age of the individual leaves (Bulthuis and Woelkerling 1983), the season (Kendrick and Burt 1997), nutrient availability (Frankovich and Fourqurean 1997) and amount of water flow (Koch et al 2006).

Different species of seagrass have different leaf turnover rates; *Posidonia* spp generally have long leaf turnover rates, typically greater than 200 days (Marbà and Walker 1999) and as such the leaves can accumulate considerable amounts of epiphytic organisms. *Halophila* sp. on the other hand have relatively short leaf turnover rates ~30 days (Hemmings et al 1991) and as such may have less epiphytes on each leaf compared to *Posidonia* if it was in a similar environment. While epiphyte load may be related to nutrient availability, other factors can be important for epiphyte production (eg (Bryars et al 2011), in circumstances where these factors may be significant caution will be applied through deliberation during the panel assessment of condition (see Section 7).

Each 50-m belt transect will be analysed for epiphyte load where the evaluation of epiphytes is relative to the amount of seagrass rather than an estimate of epiphyte biomass (examples can be seen in Appendix 6). This measure represents the load of epiphytes measured as being sparse, moderate or dense. If there is no seagrass present then the result is a null. Calcareous epiphytes are a natural part of seagrass ecosystems (James et al 2009) and literature suggests that there is a shift from calcareous to filamentous epiphyte species with increasing nutrient enrichment (Frankovich and Fourqurean 1997; Cambridge et al 2007).

The Nearshore MER program recognises that some degree of calcareous epiphytes may be a natural occurrence and not necessarily related to proximity to an anthropogenic nutrient source. Locations with high loads of epiphytes which do not correlate with other lines of evidence regarding nutrient enrichment may be further interrogated using the high definition video to provide more information on the type of epiphytes and whether it is an indicator of excess nutrients. Examples of sparse, moderate and dense epiphyte loads on various densities of seagrass are shown in Appendix 6.

Opportunistic macroalgae

Opportunistic macroalgae are species that have been shown to be indicative of nutrient enrichment including *Ulva* spp (Steffensen 1976; Connolly 1983; Campbell 2001; Cambridge et al 2007) and *Hincksia sordida* (Campbell 2001; Bryars et al 2006a; Lovelock et al 2008). Fifty-metre belt transects will be analysed for the abundance of opportunistic macroalgae and scored semi-quantitatively as sparse, moderate or dense. Visual examples of opportunistic macroalgae are included in Appendix 6.

6.2.3 Invasive species and marine debris

Through the video analysis process trained operators will record any occurrence of invasive species that may be identified using the analogue or high definition video systems. The geo-encoded position of the organisms will be recorded and given to BiosecuritySA for confirmation and field validation. If the presence of an invasive species can be confirmed then this will be reflected in the AECR condition assessment by reducing the condition score to reflect the ecological ramifications of the presence of a new or already established invasive species.

Marine debris can cause a variety of ecological and aesthetic issues in nearshore marine waters. The type of debris can range from large plastics (bags, packing tape, etc), rope, wood from pallets or crates, glass and fishing gear including buoys, nets, pots and bait baskets (Edyvane et al 2004). If any marine debris is sighted in the video assessment or during the fieldwork, the type and location will be recorded for future reference to identify trends or sources. If any material is sighted during the fieldwork, and it is safe to do so the field personnel will retrieve the material and dispose of it when ashore. The location and type of material will still be recorded and included in the assessment.

7 Panel assessment of nearshore marine condition

The methods in Section 6 outline a risk-based, tiered framework to broadly assess ecological condition across a large spatial unit with the ability to investigate at a finer level of detail if required. The indicators and disturbance gradient models presented are based on available information from peer-reviewed scientific journals, government, universities and industry, and represent an assessment of the likely reference condition of nearshore (2–15 m) marine waters and patterns in degradation due to excess nutrients or poor water clarity in South Australia. The proposed Nearshore MER program is designed to be iterative where information from ongoing work can be fed into the program to refine the conceptual models or refine the criteria for each parameter. Advances in underwater video technology may also allow for higher resolution video collection and remote sensing techniques for assessment of more factors.

The Nearshore MER program aims to assess habitat condition at a biounit scale (10s to 100s of km^2) but can also be scaled up to larger spatial units such as bioregions or whole of gulf, where assumptions and limitations are outlined. However, there may be instances or particular biounits that do not fit the models and using the models described here would result in erroneous conclusions.

For example, a complex situation occurs in Coffin Bay on the western coast of the lower Eyre Peninsula where it is well known to be influenced by cool nutrient rich upwellings from the continental shelf (Pattiaratchi 2007). As such the area typically has increased total and soluble nutrient concentrations (EPA, unpublished data), higher phytoplankton biomass (Dimmlich et al 2004; Pattiaratchi 2007) and likely flow-on effects throughout the food web. It is quite possible that the local ecology in Coffin Bay has adapted to a higher nutrient and phytoplankton environment, thereby possibly maintaining a good state despite conditions typical of nutrient enrichment.

The terrigenous habitats throughout the region however, has also been heavily cleared for agriculture, and water quality and ecological condition in the local streams has been shown to be poor with evidence of nutrient enrichment, poor riparian habitat, silt deposition and bank erosion (EPA 2012). The ability to tease out the differences between ‘natural’ upwelling-derived nutrient loads compared to runoff from agricultural catchments is fundamental in being able to define ecological condition in Coffin Bay. While this case is likely to require a Tier 3 type investigation to tease out these aspects as a broad overview, the sole use of the process outlined in this document would likely lead to an underestimation of ecological condition.

In order to overcome this problem for the Tier 2 broad assessment, the condition of each region as well as the habitat modifiers will be assessed by a panel of marine ecology and water quality scientists. This panel will review the results of the program as the Tier 1 assessment to determine whether the results of the monitoring are an accurate reflection of the current condition. The use of a panel of experts allows extraneous information to be included in the review of the assessment such as point source discharge information, more detailed scientific literature review (eg hydrodynamic modelling, etc) and aerial photography which can aid in the interpretation of results (as well as in the Tier 1 assessment) and the wealth of knowledge provided by years of experience in the field in order to assess whether the results are an accurate reflection of the condition at that point in time.

The panel will be made up of the EPA staff specialising in marine water quality and a number of external independent scientists. The composition of the panel members will be confirmed at the time of writing the assessment report for each bioregion. In order to maintain transparency in the methodology, the assessment report will be published with details of changes to the raw results following the panel review.

8 Reporting

Reporting the results of the Tier 2 assessment of broad habitat condition at the biounit scale is an important step in the program and will be undertaken using two communication tools. The first is the Aquatic Ecosystem Condition Report (AECR) which is a series of interactive web pages using the grading system discussed in Section 4.2. Each biounit is classified according to the conceptual model of habitat condition discussed in Tables 1–4 and Figure 6.

The AECR is a communication tool designed for exposure to the general public, rather than conveying scientific concepts to highly educated audiences. Each biounit will have an AECR published which shows the predicted condition (Tier 1) and the observed condition (Tier 2) and a basic summary of key findings, a description of the location, a discussion of results, an identification of key pressures on the location and a number of management actions which are currently being undertaken by government or industry to reduce the pressure exerted on the ecosystems in the location (see example AECR in Appendix 7). AECRs will be published each year on the EPA website along with an interactive map of the sampling sites <www.waterconnect.sa.gov.au>, a link to download the raw data and a two-minute representative compilation of the benthic habitat video.

It is accepted that even though there are problems 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 concept. It is hoped that a diagnosis of poor condition will raise community and political concern, and result in action to manage the issues highlighted (Deeley and 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 means, and where possible this will be conveyed through the AECR format.

The second publication format is an assessment report as discussed in section 5.2.5, which outlines in more detail the key results from the monitoring for that region, and which is focused on a more scientific audience. These reports are still accessible to the general public and will be available on the EPA website.

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10 Glossary

Ambient: an adjective that means surrounding, completely enveloping, encompassing. In terms of ambient water quality it generally refers to the environment not immediately impacted by a point source discharge. This means it is possible to describe the ambient water quality in a national park, broad-scale agricultural setting or even some urban catchments, but not immediately downstream from an industry discharge point or pipe.

Condition: refers to the state in which things exist and can be quantified as a quality or rank. The term relies on defining the criteria to rate different states or levels of assessment in the case of water quality indicators.

Diffuse pollution: typically refers to non-point source pollutants that run off or seep into waterways from broad areas of land such as agriculture or urban settings, as well as dispersal from airborne pollutant sources. Non-point sources are generally the largest contributors to water pollution at the catchment scale.

Disturbance: refers to the negative change in condition of a system from a previous desirable state, can be mediated by natural (eg storms) or anthropogenic (eg pollution) forces.

Rating: refers to the condition classification or grade. This is intended as a simple communication tool, to summarise the relative condition in a single, simple-to-understand phrase.

Point source pollution: refers to the entry of pollutants from well-defined locations, such as a pipe or sewer outflow. Factories, sewage treatment plants and stormwater outflow pipes are common point sources of water pollution.

Reference: a benchmark that is typically used as a comparison of a state or condition against a natural, unaffected, preferred or desired state or range of states.

Spatial: an adjective that refers to the nature of space, size, area or position.

State: another term for condition, stage, rank or circumstances at any time.

Stressor: refers to a physical, chemical, or biological pressure that can act on an ecosystem and drive changes in the condition of the system.

Sub-program: refers to the monitoring and assessment activities relating to different South Australian water types—nearshore marine, creeks, lakes and wetlands, groundwaters and estuaries.

Temporal: an adjective that refers to the timeframe(s) over which monitoring and reporting occurs.

Trend: refers to change with respect to time. This has typically been interpreted and assessed using standard statistical analyses which rely on showing a linear relationship between an indicator and time. There are a number of assumptions that need to be considered when using different statistical approaches, and in many cases it is arguable whether they are appropriate for highly variable water quality data. An increasing or decreasing linear trend in individual water quality parameters is of limited value in the context of assessing ecosystem status. With regard to the South Australian aquatic ecosystems MER program, changes in condition over time will be used to show trends in environmental quality for different waters in the state.

Water: refers to all waters including inland, groundwater, estuaries and marine waters.

Water quality: in view of the complexity of factors and the large choice of variables used to describe the status of water bodies in quantitative terms, it is difficult to provide a simple definition of water quality. For the purposes of the EPA, water quality is a technical term that refers to the suitability of water to sustain various uses and processes. It is typically thought of and described in terms of the biological, physical and chemical properties of an aquatic environment.

Appendix 1 Assessment of reference condition

Ecological condition is not easily measured particularly in marine systems, and there are numerous factors that correlate to condition; these measures can vary depending on the spatial and temporal scales that are investigated. Cheshire et al (1998) suggest one method of defining ecological condition is through the attributes of a 'baseline' or unimpacted state, and that the level or severity of disturbance can be judged against deviation from that baseline state. This approach has been adopted here where the Nearshore MER program was trialled at a number of locations that were considered far removed from current risk factors.

The aim of the assessment of reference condition was to:

- Observe and document habitats in a natural or unimpacted state within the bounds of what unimpacted habitats are thought to be currently available in South Australia in order to further contribute to the conceptual models and our understanding of what constitutes reference condition in shallow South Australian nearshore coastal waters.
- Develop a range of indices that relate to unimpacted shallow nearshore marine environments in South Australia. This data will be supplemented by historical EPA data from locations with known impacts of nutrients and/or poor water clarity to define an index for a range of parameters used in the MER program to describe ecological condition.

The surveys were undertaken while trialling methods to determine logistical and practicality of monitoring marine environments. Methods utilised in the survey were designed to provide observations of ecological assemblages and the key features of each of the habitat types using underwater video equipment.

Methods to assess reference condition

In order to characterise habitats over a site, 10 x 50-m underwater video belt transects were undertaken at randomly chosen locations in water between 2–15 m deep. Transects were undertaken using a geo-referenced 450-line analogue video camera (Scielex/Kongsberg) angled at 90 degrees to the seafloor, in a custom-made housing. A live video feed to a surface screen viewed by a trained operator ran directly from the camera into an audio and video encoding system (Geostamp) which overlays a GPS location, direction, speed, date and time strings overlaid to the video and recorded to a hard drive. The surface screen and trained operator allowed the camera to be positioned approximately 1 m from the substrate in order to maximise image quality and resolution. This set-up provided a field of view of approximately 1 m^2, where each belt transect equates to approximately 50 m^2. Videos were analysed upon return from the field using an inhouse video analysis software package. The operating procedure is outlined in Appendix 4. At times, the resolution of the analogue video camera can appear pixilated when finer detail may be warranted. Therefore a full high definition (HD) video camera (GoPro Hero 2) was synchronised with the analogue camera, and when analysis demands a higher resolution for taxonomic identification or finer detail (eg rocky reef assessment), the HD footage can be used.

Quantifying water chemistry at each site was undertaken by sampling three replicate 2.5-litre water grab samples at each transect location mixed into a 25-litre container. After three transects the water in the container was mixed thoroughly and sub-sampled. This process is repeated across the site for all transects (n = 9) to provide a snapshot 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 a 2-litre grab sample is taken 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 frozen and analysed within the laboratory holding times.

Compositing water samples is a method that 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). The method for compositing water samples incorporated into the Nearshore MER program is outlined in Appendix 5.

A multi-parameter sonde (YSI 6920 v2) was used to log water quality parameters including electrical conductivity, pH, dissolved oxygen and chlorophyll *a* at 10-second intervals for a total of approximately 2.5 minutes at each location (n = ~15 per transect and ~150 per site). At Reference Site 1 an in-situ multi-parameter sonde (YSI 6920 v2) was set at 1 m below the water surface logging at a frequency of one sample every five minutes in addition to the sonde used at each transect. This sonde was left in place while stationary at each of the nine locations for approximately one hour.

Water samples were analysed at the Queensland Health Scientific Services Laboratory in Queensland with a random selection of samples analysed by the Australian Water Quality Centre in Adelaide and ALS laboratories in Melbourne for quality assurance. Samples that were below the limit of reporting were given a value of half the reporting limit (Ellis and Gilbert 1980). This arbitrary method does have its limitations (see Helsel 1990) but in the Nearshore MER program, it was considered appropriate due to the amount of data generated, the low number of ‘non-detects’ and the unbiased nature of using half the reporting limit.

Results

Reference Site 1 – Pearson Island and Flinders Island

In June 2009 the EPA sampled nine locations comprised of 72 randomly-positioned nearshore transects around Flinders and Pearson Islands in waters less than 15 m deep (Figure 11 and Figure 12). These islands are located 17 and 35 nautical miles southwest of Elliston in the Southern Ocean. The marine environment surrounding these islands is largely considered to be pristine (Bryars and Wear 2008; Miller and Wright 2008) as they are far removed from any anthropogenic input other than the occasional fishing vessel. Flinders Island has been cleared for agricultural use, although the current level of farming is unknown as access to and from the island for produce is extremely limited.

Descriptive statistics from the sampling trip are contained in Table 13 and key observations are highlighted below.

Of the 72 sites, 28 (38%) had seagrass on at least some part of the site. These sites showed that:

- Meadows were generally continuous *Posidonia* spp and a number of small areas of *Amphibolis* spp
- The seagrass was in very good condition with an average habitat structure index of 95 out of 100.
- Epiphyte loads were considered to be ‘low’.
- There were no observations of opportunistic macroalgae (eg *Ulva* spp, *Hincksia sordida*).

The results are consistent with the field surveys in 2006 which assessed seagrass and reef using both SCUBA, acoustic and video methods (Bryars and Wear 2008; Miller and Wright 2008). Additionally, observations here were also consistent with studies of seagrass distribution (Gobert et al 2006), colonisation (Clarke and Kirkman 1989) and condition (Shepherd et al 1989b; Collings et al 2008) as outlined in section 2.1.1.

Seven 1-m^2 video quadrats were assessed to broadly characterise key features in rocky reef composition based on the life-form codes utilised by the Reef Health Program (Turner et al 2007; Collings et al 2008). Seven samples were considered to be a low number to characterise a site. Limitations in the random sampling design encountering rocky reefs and technical difficulties with camera equipment dictated the available data.

The reef results can be summarised as:

- Rocky reef canopies were dominated by large brown ‘robust’ macroalgae (eg *Cystophora* spp). The average density was 54% and the range was between 42–73%.
- There were very low densities of turfing algae, typically less than 1%.
- There were some areas of bare substrate (16% of the area monitored). The bare substrate was likely reflecting the reasonably high wave energy environment seen at a number of the sites.

Figure 11 Position of 49 sampling locations at Flinders Island in June 2009.

Even though the amount of replication in the EPA survey was low, the observations are consistent with studies of reef communities at Flinders and Pearson Islands undertaken between 1969–2006 (Shepherd and Womersley 1971; Baker et al 2008) and conclusions about rocky reef condition in other areas of South Australia (Shepherd and Sprigg 1976; Shepherd and Womersley 1981; Turner 2004; Connell 2007; Turner et al 2007; Collings et al 2008; Connell et al 2008).

Throughout the 2009 observations at Flinders and Pearson Islands there were only three locations where bare sand was encountered in waters less than 15 m deep that enabled an opportunity to classify biological components of this habitat type. These sites showed very little variation with a very low density of visible bioturbation (holes/quadrat^{-1}) and minimal visible epi-fauna.

Water quality data showed that throughout the islands, only 3 samples were below the analytical limit of reporting (LOR) for ammonia (LOR = 0.002 mg/L) and oxidised nitrogen was detected at all locations; suggesting that these locations were not nitrogen limited at this time of year (Table 13). (Bryars et al 2006b) demonstrated changes in nitrogen limitation in South Australian coastal waters throughout the year, with soluble nutrients routinely detected during autumn/winter. In addition to the nutrient values reported here the chlorophyll samples showed good agreement between the stationary (in-situ) sonde and the sonde measurements along each transect.

Results of the water samples showed that at the time of sampling the total nitrogen concentrations were in the order of 0.130 mg/L and the dissolved inorganic nitrogen were in the order of 0.011 mg/L (Table 13). Both parameters had very low standard deviations indicating low variability in the data. Chlorophyll *a* concentrations were typically in the order of 0.510 µg/L.

Figure 12 Position of 21 sampling locations at Pearson Island during June 2009.

Table 13 Summary statistics for water chemistry parameters at Flinders and Pearson Islands

	Total nitrogen (mg/L)	Total ammonia (mg/L)	Oxidised nitrogen (mg/L)	Dissolved inorganic nitrogen (mg/L)	Total phosphorus (mg/L)	Soluble phosphorus (mg/L)	Turbidity (NTU)	Chlorophyll *a* (µg/L)
Median	0.130	0.005	0.006	0.011	0.011	0.003	0.160	0.516
80th Percentile	0.188	0.007	0.007	0.0128	0.012	0.004	0.208	0.569
Standard deviation	0.083	0.002	0.002	0.003	0.002	0.001	0.047	0.073
Count	27	27	27	27	27	27	27	3,968

Reference Site 2 – Sir Joseph Banks Islands

In order to further validate an assessment of reference condition a second survey was undertaken at the Sir Joseph Banks Group of Islands (SJB). The SJB is compromised of 20 or so islands located approximately 15–20 nautical miles from Port Lincoln in Spencer Gulf. A risk assessment process has highlighted that the islands are less removed from anthropogenic influences than the Flinders and Pearson Islands, but could still be considered to be relatively unimpacted due to the distance (~5 nm) from the nearest southern bluefin tuna (*Thunnus maccoyii*) aquaculture lease. Nonetheless, the net direction of water movement has been shown to be from the tuna farming zone generally towards the top of the SJB group (Hertzfeld et al 2009). Spilsby Island has a number of residential dwellings on the northern coast and the area is popular for recreational boaters, particularly fishing. At the time of sampling these islands were considered likely to be as unimpacted as any other location in Spencer Gulf or Gulf St Vincent, provided an opportunity to assess condition of a relatively ‘pristine’ environment within the gulf systems.

In May 2010, EPA marine scientists undertook condition surveys at 14 sites in the SJB (Figure 13). This assessment of reference condition used slightly modified methods to those outlined in section 5 where the replication within a site was increased to 10 replicate 50-m belt transects. Each site was nominally defined as a 20-hectare area in waters between 2–15 m deep.

Figure 13 **Location of 14 sampling sites in Sir Joseph Banks Group of Islands during May 2010. The upper inset shows the start and finish of 10 randomly located transects within one of the sites (SJB109).**

A total of 140 x 50-m video transects were recorded throughout the Islands to assess the key features of habitat condition and ambient water chemistry that focused on total/soluble nutrients and turbidity. The key observations from the EPA survey based on the assessment criteria are detailed below.

The EPA survey found that of the 140 sites, 118 (84%) had seagrass on some part of the site with the following results:

- Seagrass meadows were typically continuous *Posidonia* spp but there were some areas that were interspersed with rocky reef.
- The habitat structure index was on average 90 out of 100.
- Epiphyte load for the region as a whole was considered to be 'low' however there were a number of sites that had moderate epiphyte loads, particularly sites to the north of Reevesby, Winceby and Marum Islands, and to the north of Spilsby Island (SJB 101, 106, 113 & 115, Figure 13).

Throughout the SJB, 40 video transects had a proportion of macroalgal reef on the transect. Subsequently 59 quadrats measuring 1 m^2 were analysed using functional life form codes (Cheshire and Westphalen 2000; Turner et al 2007; Collings et al 2008) to make the following broad observations regarding macroalgal reef composition:

- Rocky reefs were dominated by large brown 'robust' macroalgae (eg *Ecklonia radiata, Cystophora spp*), which formed dense canopies across much of the reefs. The mean density of robust brown macroalgae was 72%.
- There were very low densities of turfing algae, typically less than 1%.
- There were few areas of bare substrate which made up approximately 11% of the quadrats.
- There were no observations of opportunistic macroalgae (eg *Ulva* spp, *Hincksia sordida*).
- There were no transects considered to be unvegetated soft sediments.

Results of the water sampling indicated that both total and soluble nitrogen and phosphorus were relatively consistent with low standard deviations indicating that the measurements were relatively precise (Table 14). At almost all locations oxidised nitrogen (nitrate + nitrite) was below the reporting limit, suggesting very low oxidised nitrogen concentrations and possible nitrogen limitation. Typically total nitrogen values were in the order of 0.170 mg/L and dissolved inorganic nitrogen was generally 0.025 mg/L both with small standard deviations again indicating low variability (Table 14).

Table 14 Summary statistics for water chemistry parameters at Sir Joseph Banks group of islands

	Total nitrogen (mg/L)	Total ammonia (mg/L)	Oxidised nitrogen (mg/L)	Dissolved inorganic nitrogen (mg/L)	Total phosphorus (mg/L)	Soluble phosphorus (mg/L)	Turbidity (NTU)	Chlorophyll *a* (µg/L)
Median	0.170	0.010	0.015*	0.025	0.015	0.007	0.210	0.666
80th Percentile	0.240	0.017	0.015*	0.032	0.017	0.015	0.318	0.868
Standard deviation	0.063	0.007	0.003	0.008	0.004	0.005	0.090	0.090
Count	42	42	42	42	42	42	42	2,309

* Oxidised nitrogen in the marine environment commonly results in values that are below the analytical reporting limit. At the Sir Joseph Banks group of Islands there were 40 out of 42 (95%) values below the limit of reporting. As stated in section 5.1.1 a value of half the reporting limit was adopted for values below the reporting limit (Ellis and Gilbert 1980, Helsel 1990). However, caution should be applied when interpreting such data. In the future laboratories with lower reporting limits will be selected for analysis.

There were a number of locations that showed some elevated epiphyte loads compared to other reference site locations (SJB 101, 106, 113 & 115). Analysis of the imagery suggested that these epiphytes were typically comprised of filamentous brown or red epiphytes with similar morphology to epiphytes from areas with known impacts from nutrient enrichment (Figure 13). The similar morphology of epiphytes would suggest that the epiphytes at the SJB are not a natural calcareous type that is sometimes seen in areas far removed from a nutrient source (James et al 2009). These epiphytes may be indicating that these islands are experiencing some degree of nutrient enrichment.

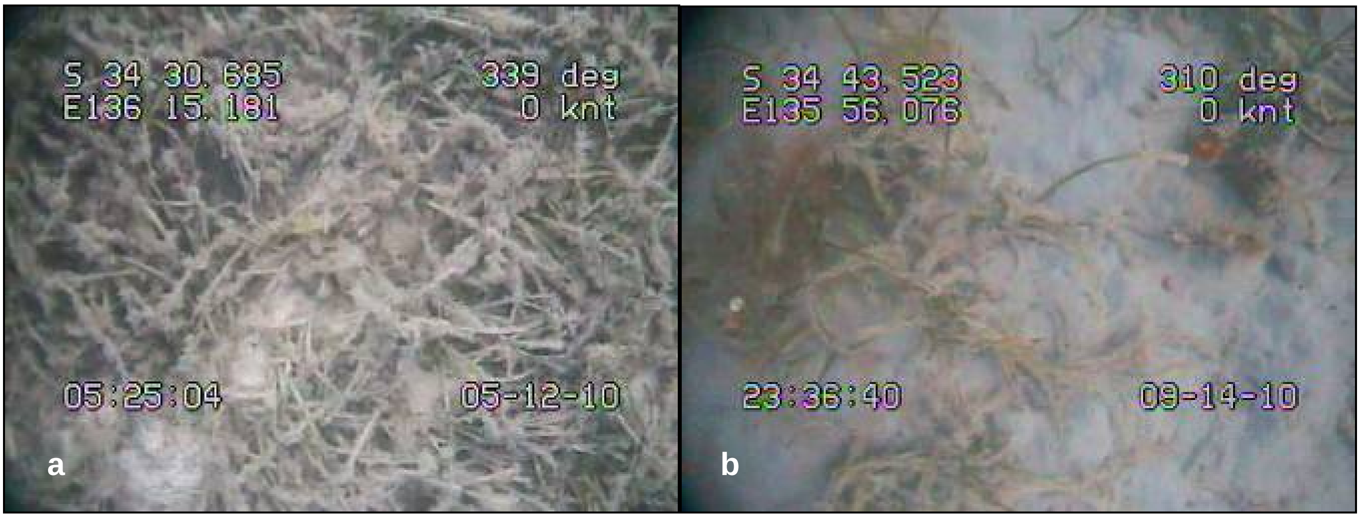


Figure 14 **Screen capture of epiphytes on *Posidonia* seagrass at (a) SJB 113 and (b) a location with potential impacts from excess nutrients (Rotten Bay, Lower Spencer Gulf).**

Summary of reference site results

Any monitoring and evaluation program that assesses temporal change requires an adequately quantified baseline condition (or benchmark) against which potential changes can be measured and quantified in relation to natural variability (Dayton et al 1998). Pauly (1995) describes how a poorly quantified baseline can create a ‘shifting baseline’ syndrome where a benchmark is accepted as being the norm when monitoring commences, while in fact this benchmark has actually shifted from the ‘real’ starting point prior to the benchmark being set. The shifting baseline syndrome results in an accommodation of the new baseline as being the norm and the gradual decline in natural resources goes undetected (Pauly 1995).

It should again be reiterated that the Nearshore MER program is assessing condition at a broad regional scale. The key findings from the assessment of reference sites were:

- Seagrass habitats were generally continuous meadows of *Posidonia* or *Amphibolis* spp.
- Seagrass condition using the habitat structure index (Irving et al 2013) was typically greater than 90 out of 100.
- Epiphyte loads were very low, however, there were a number of locations in the Sir Joseph Banks Group of Islands that may be showing initial signs of nutrient enrichment.
- Rocky reefs were typically dominated by a canopy of robust brown macroalgae, typically *Cystophora* spp or *Ecklonia radiata*.
- The proportion of canopy forming robust brown algae was typically higher than 50%.
- There were no observations of opportunistic macroalgae (eg *Ulva* spp, *Hincksia sordida*).
- Unvegetated sediments were generally sparsely inhabited with no observations of the taxa identified by Cheshire et al (1996) to indicate organic enrichment, and the level of bioturbation was considered to be low.
- Typical total nitrogen values are in the order of 0.150 mg/L and dissolved inorganic nitrogen was typically in the order of 0.018 mg/L.
- Chlorophyll *a* concentrations were typically to the order of 0.591 µg/L.
- Water clarity was high with typical turbidity results in the order of 0.190 mg/L.

Observations from Flinders and Pearson Islands as well as the Sir Joseph Banks Group of Islands are consistent with findings from the scientific literature detailing the ecology for these particular regions (Shepherd and Womersley 1971;

Baker et al 2008; Bryars and Wear 2008; Miller and Wright 2008) as well as for the model of anthropogenic impacts from eutrophication (Section 2).

Table 15 shows the results of the pooled reference location water chemistry. A comparison to values from the long term data set from the historical EPA ambient water quality monitoring program (Table 15) shows that all parameters in reference locations were significantly lower (Mann Whitney test, $p < 0.001$) than at the metropolitan nearshore waters (Gaylard 2004), EPA unpublished data). The standard deviation of the data indicates that there was lower variability in the reference locations than at the metropolitan sites, which fits with the notion that variability increases as the level of impact increases (Odum et al 1979; Warwick and Clarke 1993).

Table 15 Descriptive statistics for reference sites

	Total nitrogen (mg/L)	Total ammonia (mg/L)	Oxidised nitrogen (mg/L)	Dissolved inorganic nitrogen (mg/L)	Total phosphorus (mg/L)	Soluble phosphorus (mg/L)	Turbidity (NTU)	Chlorophyll *a* (µg/L)
Median	**0.150**	**0.007**	**0.015***	**0.018**	**0.012**	**0.005**	**0.190**	**0.591**
80th Percentile	**0.228**	**0.014**	**0.015***	**0.029**	**0.016**	**0.015**	**0.278**	**0.719**
Standard deviation	**0.072**	**0.007**	**0.005**	**0.010**	**0.004**	**0.005**	**0.085**	**0.120**
Count	**69**	**69**	**69**	**69**	**69**	**69**	**72**	**6,277**

* Oxidised nitrogen data affected by 58% (40 out of 69) of values being below the reporting limit.

Appendix 2 Historical EPA monitoring – impacted sites

In order to determine the upper bounds of the water chemistry parameters of the ecological condition gradient, descriptive statistics for the EPA's historical water quality monitoring program were evaluated for the data from 1998–2008 (Gaylard 2004; EPA unpublished data). Data prior to 1998 were excluded due to uncertainties in the quality of the analytical results for nitrogen (Gaylard 2004).

The sites chosen for the assessment of 'impacted' sites were Largs Bay, Semaphore, Grange, Henley Beach, Glenelg and Brighton jetties. These six sites were historically sampled monthly as a part of the EPA's ambient water quality program. The six impacted sites have been shown to have elevated nutrient and/or chlorophyll *a* concentrations (Gaylard 2004) and are located adjacent to large areas of seagrass loss as shown in Figure 15 (Hart 1997; Cameron 2008) and degraded rocky reefs (Collings et al 2008). The multiple lines of evidence from these reports suggest that these locations are in poor condition and the EPA's long-term data set is likely to provide the best available estimate of long-term ambient water chemistry at these locations. Table 16 summarises the averaged descriptive statistics from the six impacted sites.

Table 16 Descriptive statistics for 'Impacted' sites from 1998–2008

	Total nitrogen (mg/L)	Total ammonia (mg/L)	Oxidised nitrogen (mg/L)	Dissolved inorganic nitrogen (mg/L)	Total phosphorus (mg/L)	Soluble phosphorus (mg/L)	Turbidity (NTU)	Chlorophyll *a* (µg/L)
Median	0.285	0.021	0.005	0.035	0.015	0.036	1.570	2.090
80th Percentile	0.427	0.046	0.028	0.075	0.029	0.054	3.906	3.714
Standard deviation	0.179	0.030	0.041	0.057	0.012	0.024	3.795	2.847
Count	657	695	711	646	60	710	853	724

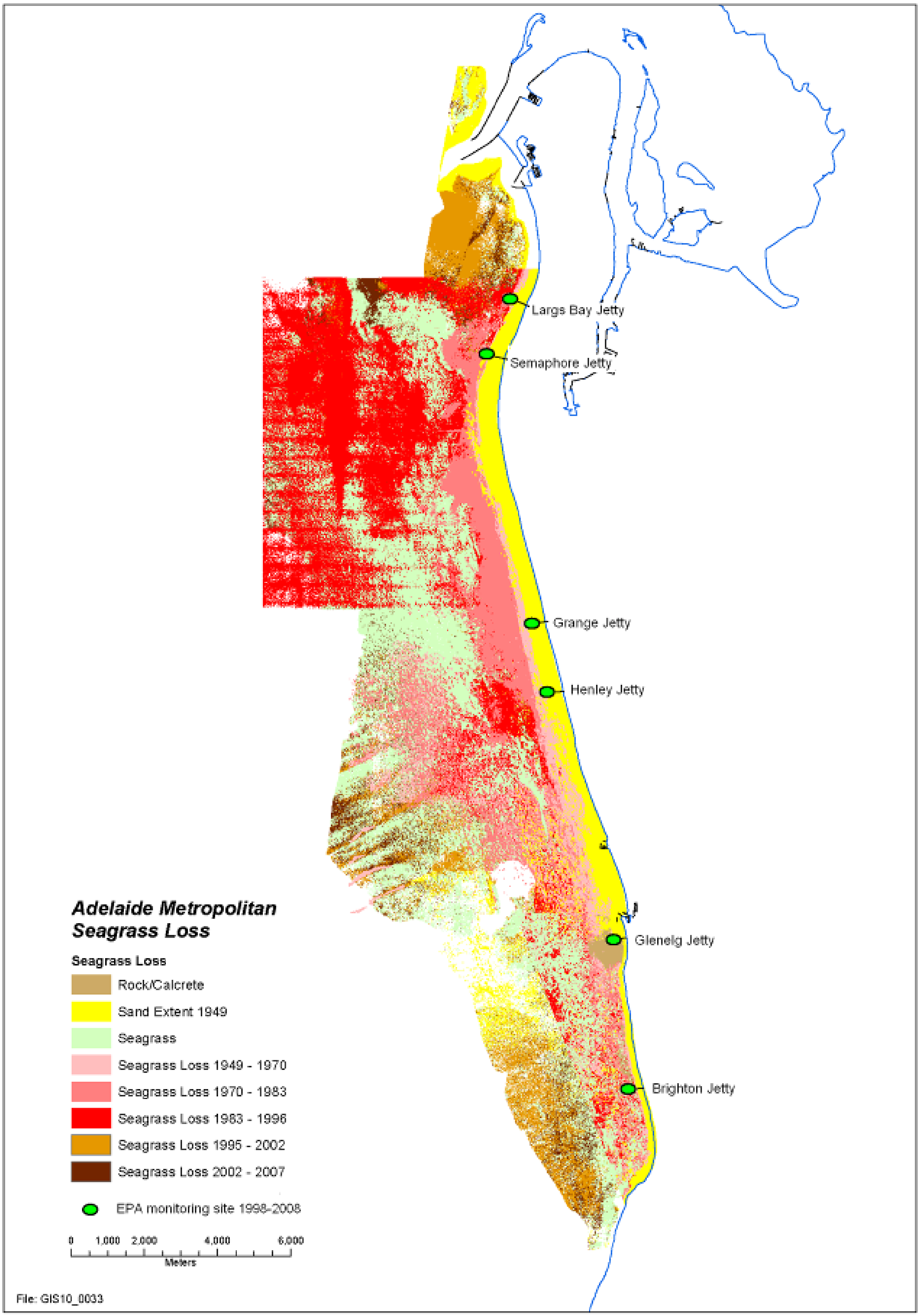


Figure 15 **Epochs of seagrass loss along the Adelaide metropolitan coast showing the location of the EPA ambient water quality monitoring sites used in the assessment of impacted water chemistry (modified from Cameron 2008).**

Appendix 3 Power analysis for seagrass condition

A pilot study testing the proposed ecological condition monitoring methods were undertaken in 2009. The study tested not only the logistics of the methods but enabled an assessment of the variability likely to be encountered. A total of 10 transects measuring seagrass condition were undertaken at each of 61 sampling sites (site = 20 ha) throughout Gulf St Vincent in spring 2009. A retrospective power analysis was undertaken to determine how much power there was to detect various levels of change throughout seagrass habitats in Gulf St Vincent. This information was then used to determine whether the number of transects would need to be increased to detect change in seagrass condition for subsequent monitoring periods.

A power analysis can be undertaken to show whether the amount of replication is sufficient to be able to detect change in the parameter monitored over the amount of variability within the samples (Fairweather 1991). Power is inversely related to the probability of making a type II error, where a type II error is concluding that there is no significant change in the environment when there actually is one, which from an environmental perspective can be considered disastrous in comparison to a type I error (Fairweather 1991).

A one sample Z test (Minitab 14) was used to retrospectively determine the adequate sample size to detect various changes in seagrass condition. The power analysis was undertaken using the Gulf St Vincent data rather than the reference condition data as it encompasses sites ranging from locations of known significant nutrient enrichment and water clarity impacts through to areas with very little anthropogenic impacts. The Gulf St Vincent data are considered to be a more representative sample of the overall variability possible in seagrass systems, with variability likely to increase as the level of disturbance increases (Odum et al 1979; Warwick and Clarke 1993).

The power of any test should be as large as possible to minimise the risk of type II error, however logistical and financial considerations will always have an influence on the amount of replication that is able to be undertaken. The emerging conventions suggest that power should be equal to at least 0.8 to have a high degree of confidence of not committing a type II error (Cohen 1988)

Table 17 shows that when looking for a 10% change in the habitat structure index within a 20-ha site, the ideal sample size is 20 transects with a power of 0.99, 14 transects with a power of 0.95, 12 transects with a power of 0.9 and nine transects with a power of 0.80. Twenty transects within each site was determined to be logistically unfeasible to cover an entire bioregion in one season with the allocated resources. Trialling the method determined that 10 transects at each site was logistically workable and resulted in sufficient confidence (0.9) to detect a 10% difference in seagrass condition (Table 18).

Table 17 Power analysis of habitat structure index throughout Gulf St Vincent 2009

Difference	Transects	Target power	Actual power
10%	9	0.80	0.833
10%	12	0.90	0.922
10%	14	0.95	0.954
10%	20	0.99	0.992
15%	4	0.80	0.831
15%	5	0.90	0.904
15%	7	0.95	0.971
15%	9	0.99	0.992
20%	3	0.80	0.921

Difference	Transects	Target power	Actual power
20%	3	0.90	0.921
20%	4	0.95	0.973
20%	5	0.99	0.992

Table 18 Retrospective power analysis using 10 replicates within a site

Transects	Power	Difference (%)
10	0.8	9.1
10	0.9	10.5
10	0.95	11.7
10	0.99	13.9

For future monitoring this level of power and degree of change in seagrass condition was considered acceptable to detect changes for a broadscale survey of seagrass condition. While the level of power for each site may change over time and across different bioregions, it is anticipated that as the program continues, the statistical power will be periodically reassessed. The number of transects may also change to provide a higher degree of confidence or a finer resolution to determine change if a Tier 3 program is required. Such change is deemed to be a separate and site-specific exercise and will be assessed at the time of design.

Appendix 4 Towed video standard operating procedure

The following methods are used by the EPA for capture of video footage using towed, geo-referenced underwater video to record and assess benthic habitats in shallow waters (<15 m LAT). The methods are designed to simply and easily evaluate and monitor benthic habitats without the use of SCUBA. The method requires at least two operators, one to control the vessel and the other to control the video equipment.

This operating procedure details the method of collecting data of benthic habitats in shallow waters. It does not look at camera set up (this will vary depending on individual systems), data analysis or archiving of data.

Once the vessel has arrived at the desired location the following procedure will allow for the collection of a 50-m underwater belt transect.

1 On a visual cue board record the location and site codes, date, transect number and operators (Figure 15) and film the board to record a visual reference point for the transect.

2 With the boat stationary, lower the camera via the umbilical cable through the water until a clear picture of the benthos is attained. The video should maximise the area of benthos in the field of view but be close enough to identify species to at least genus level in the case of seagrass. This is generally approximately 0.5–1.0 m from the substrate depending on substrate complexity and water clarity.

3 Once camera orientation is stabilised and the picture on the live feed video screen is clear, drift or put the vessel in gear to maintain a consistent movement across the benthos. The camera operator must continuously watch the live feed screen to optimise the speed of the vessel and communicate with the vessel operator to vary the speed based on the video picture attained. This speed is typically between 0.5 and 1 knot.

4 At least two minutes of clear video footage of the benthos should be captured for each transect to equate to a transect of approximately 50 m in length.

5 If the substrate depth or complexity changes it may be necessary to raise or lower the camera slightly to maintain optimal positioning relative to the benthos. Any obvious features should be communicated by the vessel operator to ensure camera safety.

6 At the end of the transect, the video should be paused or stopped, and the camera raised to the surface until the next transect location is reached.

7 Repeat for each transect in a site, depending on the degree of replication, ensuring that information on the visual cue board is recorded for each transect.

Figure 16 Visual cue board used in EPA underwater video collection

Appendix 5 Compositing water samples standard operating procedure

The following methods are used by the EPA to sample water chemistry using a composited sample with multiple replicates. These methods are designed to average water chemistry across a larger area, with the aim of reducing analytical costs and potentially averaging out peaks in some parameters to reduce variability across the site, which is inherent with many water chemistry parameters in the marine environment.

This operating procedure details the method of collecting water samples in shallow waters for the EPA's Aquatic Ecosystem Condition Reports (AECRs). It will sample three locations in each composite, with three composites (replicate samples) within the site. It does not look at analytical methods or statistical analysis.

Once the vessel has arrived at the desired location the following procedure will allow for the collection of three replicate composited batches of water which can be sampled into individual bottles for analysis.

1 A clean 2-litre PET bottle is used on the end of a 2-m telescopic pole to sample marine waters 0.5 m below the surface at each location. Prior to each new sample the bottle is rinsed three times with ambient seawater.

2 A clean 20-litre wide neck HDPE barrel is used to composite water samples.

3 Two 2-litre water samples are taken from each location. Each 2-litre sample is immediately transferred to the 20-litre composite barrel.

4 After three locations there is 12=litre of water in the 20-litre barrel. This water is thoroughly mixed and the laboratory-supplied water sample containers are filled from the tap on the 20 L barrel according to the laboratory instructions. This constitutes one replicate within the three replicates taken across an AECR site ~20 ha (Figure 16).

5 At each new replicate location the barrel is rinsed three times with ambient sea water.

6 This procedure is repeated three times within each AECR site (Figure 16).

7 Immediately after collection, water samples are placed on ice and in darkness prior to delivery to the laboratory.

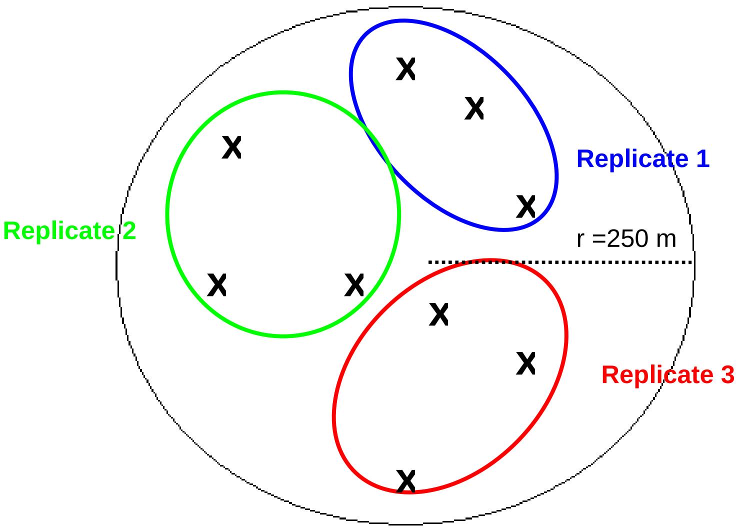


Figure 17 Schematic of composite sampling process at each AECR site

Appendix 6 Visual estimates of cover on belt transects

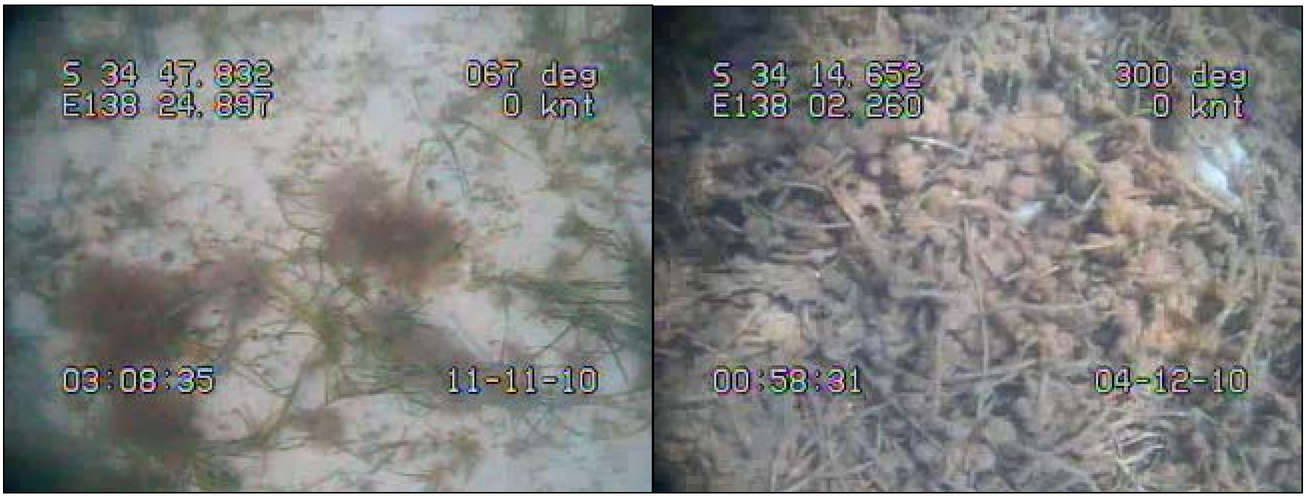
Sparse seagrass (quadrat seagrass density 1–35%)

Moderate seagrass (quadrat seagrass density 35–70%)

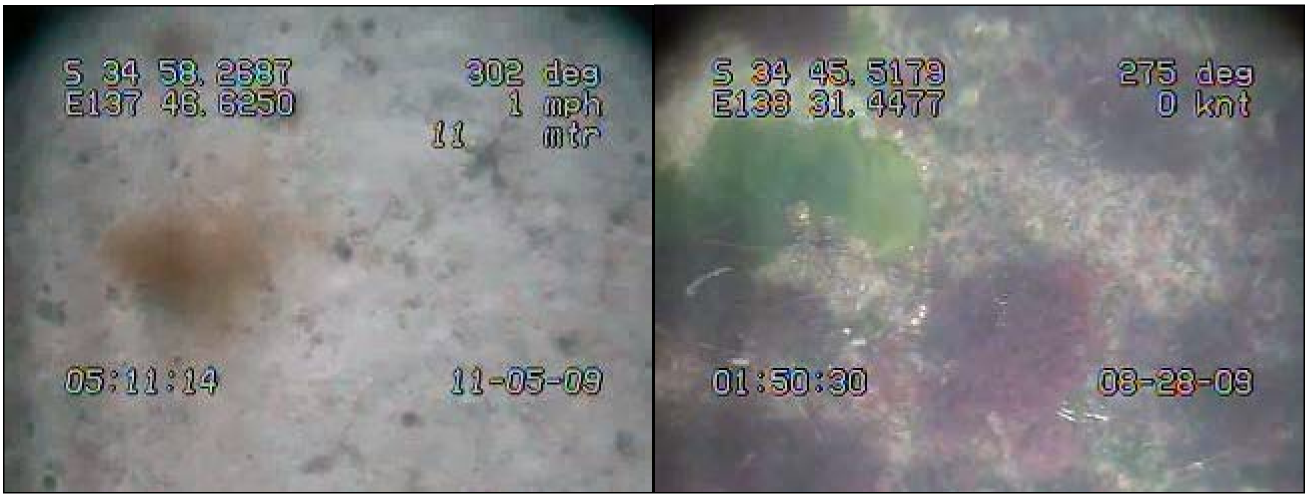
Dense seagrass (quadrat seagrass density 70–100%)

Epiphyte load sparse

Epiphyte moderate



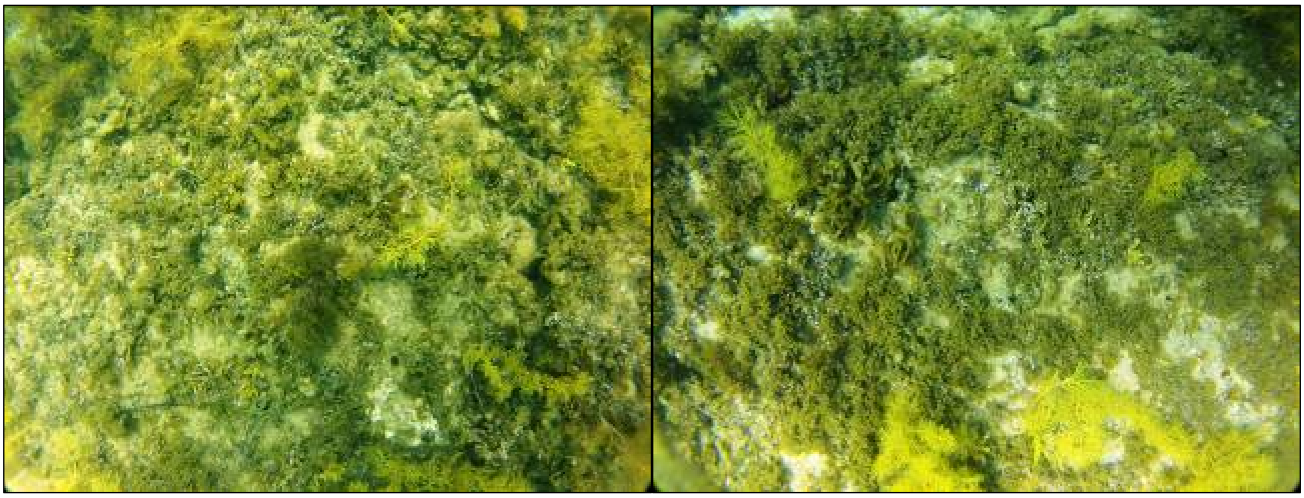
Epiphyte dense



Opportunistic macroalgae (eg *Hincksia sordida* and *Ulva sp*)

Opportunistic macroalgae is measured as a frequency of observation along the belt transect rather than density per 1 m^2

Macroalgae >40%



Turfing algae >25%

Appendix 7 Example of the online Aquatic Ecosystem Condition Report

Please click on the link
<www.epa.sa.gov.au/environmental_info/water_quality/aquatic_ecosystem_monitoring_evaluation_and_reporting>