Examining the health of subtidal reef environments in South Australia

Part 2: Status of selected South Australian reefs based on the results of the 2005 surveys

By

David Turner, Timothy Kildea and Grant Westphalen

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More information:
Further information about the Reef Health Program along with copies of reports and technical documents may be obtained from the Reef Watch website at http://www.reefwatch.asn.au, or by contacting SARDI Aquatic Sciences.

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Table of Contents

ACKNOWLEDGEMENTS 3
EXECUTIVE SUMMARY 5
1 INTRODUCTION 8
1.1 The 2005 Reef Health survey 9
1.2 Previous studies of reef health in South Australia 9
1.3 The 2005 Reef Health survey program 10
2 INDICES USED FOR THE INTERPRETATION OF REEF HEALTH 11
2.1 Areal cover of canopy-forming macroalgae 12
2.2 Areal cover of turfing macroalgae 13
2.3 Areal cover of mussel mats 14
2.4 Areal cover of bare substrate 16
2.5 Size and abundance of blue-throated wrasse, *Notolabrus tetricus* 16
2.6 Total abundance of site-attached reef fishes 17
2.7 Abundance of mobile invertebrate predators 18
2.8 Presence of invasive taxa 19
2.9 Presence of high sedimentation 20
2.10 Species richness of macroalgae; species richness of mobile invertebrates 21
2.11 Presence/absence of rare species 23
3 METHODOLOGY 24
3.1 Survey sites 24
3.2 Adelaide Metropolitan Survey 24
3.3 Fleurieu Peninsula Regional Survey 26
3.4 Yorke Peninsula Regional Survey 28
3.5 Surveys 30
3.6 Data manipulation - index calculation 30
3.6.1 Indices of areal cover 30
3.6.2 Indices of abundance 31
3.6.3 Binary indicators 32
3.6.4 Indices of species richness 32
3.6.5 Scaling of indices 32
3.5.6 Overall index of reef health 34
3.7 Data analyses 34
3.7.1 Changes to Adelaide metropolitan reefs 1996 - 2005 34
3.7.2 Status of Fleurieu and Yorke Peninsula reefs in 2005 35
4 RESULTS 36
4.1 Reef surveys along the Adelaide metropolitan coastline 36
4.1.1 Structure of Adelaide metropolitan reef assemblages 1996 – 2005 36

4.1.2 Summary of indicator scores for Adelaide metropolitan reefs *42*
4.2 Fleurieu Peninsula regional surveys 44
4.2.1 Site Comparisons *45*
4.2.2 Summary of indicator scores for the Fleurieu Peninsula regional sites *46*
4.3 Yorke Peninsula regional surveys 47
4.3.1 Site comparisons *47*
4.3.2 Summary of indicator scores for Yorke Peninsula regional sites *49*
4.4 Inter-regional comparison 51
5 DISCUSSION 54
5.1 Status of South Australian reefs 54
5.1.1 Status of metropolitan reefs *54*
5.1.2 Status of reefs on Yorke Peninsula and Fleurieu Peninsula *55*
5.2 Factors affecting the health of temperate reef systems 57
5.2.1 Toxicants and reduced salinity *57*
5.2.2 Nutrients, turbidity and sedimentation *58*
5.2.3 Other factors affecting reef status *59*
5.3 Indices of reef status 60
5.3.1 Cover indices *60*
5.3.2 Abundance indices *62*
5.3.3 Invasive species *62*
5.3.4 Sedimentation *63*
5.3.5 Richness indices *63*
5.3.6 Summary index *63*
6 CONCLUSIONS AND RECOMMENDATIONS 65
REFERENCES 68
APPENDIX A: *IN SITU* SAMPLING METHODOLOGY USED IN THE PROGRAM 76
APPENDIX B: REEF HEALTH PROGRAM STANDARD SURVEY PROTOCOL 84
APPENDIX C: REPORTING CODES USED DURING DATA ANALYSIS 86
APPENDIX D: SITE DESCRIPTIONS FOR REEFS INCLUDED IN THE 2005 SURVEYS 88

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Executive summary

Concern over the degradation of Adelaide's metropolitan reefs led to the development of a number of environmental monitoring and research initiatives. The first Reef Health survey was initiated in 1996 and expanded in 1999, with a follow-up survey completed in 2005. The 2005 survey program was considerably extended compared to previous surveys and aimed to achieve a number of objectives including:

1. An up-to-date assessment of the condition of reefs along Adelaide' s metropolitan coast;
2. A comparison of the condition of reefs in 2005 with past observations (1996 and 1999) to determine whether there was any shift in the structure of the biological communities associated with the metropolitan reefs;
3. The development and interpretation of a number of indices to assist in determining the status of reef health;
4. The establishment of baseline information for reefs in non-metropolitan areas (specifically Fleurieu and Yorke Peninsulas); and
5. A comparison of metropolitan with non-metropolitan reefs.

Generally, the north to south gradient in reef health observed across metropolitan reefs in 1996 and 1999 was also observed in 2005. Based on macroalgal functional group composition and cover, northern metropolitan reefs (sites from Semaphore to Broken Bottom) appear to be in poor condition, with red foliaceous and turfing macroalgae dominating. There are signs of further declines (compared to previous surveys) on central metropolitan reefs (from Seacliff to Southport), in particular those at Horseshoe Reef and some sites on Noarlunga Reef, with a loss of robust brown macroalgae, establishment of mussel mats, and in some instances, the development of large areas of bare substrate. Southern reefs (Moana to Aldinga) have remained much the same and appear healthy, retaining most of their robust macroalgal canopy.

Similar analyses of macroalgal cover and composition at sites surveyed during 2005 on Yorke Peninsula (11 sites) and Fleurieu Peninsula (8 sites) found reefs were generally healthy, particularly when compared to metropolitan reefs. However, there was a high level of variability within regions. Some sites (notably Point Souttar and Point Riley on Yorke Peninsula) had a relatively low cover of canopy macroalgal species, but this cannot necessarily be interpreted as poor condition without further information.

In order to obtain a more robust indication of reef status, we developed ten additional health indices. To get an overall value for any particular reef, the set of indices are averaged to obtain an overall score. The reef was then scored, and grouped into one of three categories (Poor Condition, Caution Recommended and Good Condition).

This 'stoplight' approach indicated a more complex picture than simply scoring on macroalgal functional group cover. A large number of sites across the metropolitan region fell into the Caution Recommended category, even within the generally healthier southern zone. Similarly, a few sites on the Fleurieu Peninsula coast (Granite Island and Port Elliot) rated Caution Recommended status, while four sites on Yorke Peninsula rated either Caution Recommended (Troubridge Point and Cable Hut Bay) or Poor Condition (Point Souttar and Point Riley). None of the non-metropolitan sites scored as low as northern sites on the Adelaide metropolitan coast. This is a very preliminary approach and the 'stoplight' method has drawbacks. For example, Point Souttar is in an area of naturally low current flow and high sedimentation, and may never have supported large canopy macroalgal species. Invertebrate diversity was high at this site; however, the nature of the indices used has meant that this reef has ranked low. It is important to remember, for all sites, particularly those sampled for the first time, that the data provide a snapshot of the system. The real value of this type of survey is that it will act as a baseline and enable comparisons over time.

The indices employed are not perfect; however, they are informative, with the summary average probably being the most useful. The use of a range of indices targeting different ecological aspects of reef ecosystems has led to a better understanding of the nature and complexity of these communities. Furthermore, the results and interpretations presented in this report highlight the difficulty associated with producing a robust but practical approach to assessing reef health.

To really understand the overall health of reef systems, a greater understanding of the interactions between the biological assemblages and their environment is needed. This would allow predictions to be made about the types of communities that could be expected in different environments. This would also assist in assessing impacts from anthropogenic sources. The development of indices employed in the interpretation of reef health is an evolving process that will be refined in tandem with increasing knowledge of the dynamics of southern temperate reef systems.

Additionally, different types of putative impact should be targeted, such as industrial areas; reefs in proximity to coastal developments; and reefs subject to different fishing intensities or other extractive industries. Such data will further expand our knowledge of what constitutes a 'healthy' reef, and assist in the development of management and remediation strategies for reef systems.

The following recommendations are made:

- Baseline data needs to be collected from other reefs across South Australia (Eyre Peninsula, West and Southeast coasts). A range of sites including near pristine and potentially impacted areas should be included;
- Data should also be collected from areas of high conservation value as well as those areas likely to be subject to human impact;
- Further (and more focused) monitoring should be carried out for sites which are rated Poor Condition or Caution Recommended by the stoplight approach;
- The link between abiotic factors (e.g. substrata composition, wave exposure) and the biotic assemblages present on a reef requires further investigation. This would allow biologists to make predictions (which can then be tested) about the types of biotic assemblages that should be expected under various conditions;
- The reef health indices need to be further refined, and preferably augmented with data on keystone species. The concept of indicator species should be further investigated;
- The potential influence of climate change on reef ecosystems needs investigation;
- The potential impact of seagrass loss off Adelaide on reef health should be investigated;
- Community-based reef-monitoring initiatives (e.g. Reef Watch) are a cost effective method for increasing the volume of information that can be collected, and should be supported.

1 Introduction

Temperate reef ecosystems have generally been less studied than their tropical counterparts, but are environmentally, socially and economically important. Urban development in Australia is overwhelmingly coastal, and degradation of nearshore reef and seagrass systems is increasingly a cause for concern. Important sources of stress in urbanised coastal environments include (but are not limited to) wastewater and stormwater discharges, fishing, pollution (heavy metals, pesticides, hydrocarbons, etc.) and coastal development (see review in Turner *et al.* 2006a). All of these factors have been variously implicated in the degradation of marine habitats both in Australia and internationally. Nevertheless, the degree to which these impacts are understood and/or managed is highly varied.

State of the Environment reports provide a regular summary of the major issues relating to a variety of natural habitats, including (amongst others) estuaries, seagrass systems, catchments and coral reefs, at both the national and state level (Nicolson *et al.* 2003). As recently as 1995 the section on rocky reefs in South Australia was limited to a single phrase 'Status not known', which highlights the paucity of information available at that time (Edyvane 1995).

Since 1995, a number of initiatives have been taken to address this knowledge gap in South Australia, including government monitoring programs and research undertaken by local universities. Key amongst these activities were the Reef Health surveys conducted in 1996 and 1999. These surveys highlighted some knowledge gaps, and as a result other research programs were developed on specific issues such as sedimentation of reef systems (Greig 2000), the influence of excessive mussel numbers on reef flora (Smith 2000) and the dynamics of macroalgal recruitment (Turner 2004). Fundamental ecological research in temperate systems in South Australia has been greatly boosted by the establishment of the Southern Seas Ecology Laboratories at The University of Adelaide in 1999.

In response to increasing levels of public interest and concern, the Reef Watch Community Environmental Monitoring Program was formed in 1997 as a partnership between the scientific, conservation, and environmental management sectors (Turner *et al.* 2006b). This program uses methods based on those developed for the Reef Health surveys, and indeed many of the scientists who conducted the Reef Health surveys in 1996 and 1999 have served on the Reef Watch Steering Committee. In addition, the recently completed Adelaide Coastal Waters Study has worked to develop an understanding of seagrass losses on the Adelaide metropolitan coast (Westphalen *et al.* 2005), along with related research into effective methods for rehabilitation of degraded seagrass systems (e.g. Seddon *et al.* 2005, Wear *et al.* 2006).

Six years elapsed between the 1999 Reef Health survey along Adelaide's metropolitan coastline and the expanded survey which is the focus of this report, which was completed in the summer of 2005. This most recent investigation was carried out in response to evidence of further reef degradation (especially along the central coast, Turner 2004), despite a number of initiatives to improve coastal water quality (Nicolson *et al.* 2003, Gaylard 2004, Turner *et al.* 2006a). Attempts have also been made to investigate and explain some of the ecological mechanisms behind these changes (e.g. Gorgula and Connell 2004, Turner 2004).

1.1 The 2005 Reef Health survey

The purpose of this document is to provide a summary and interpretation of data collected during the summer 2005 Reef Health survey. The data are examined against a number of objectives:

1. An up-to-date assessment of the condition of reefs along Adelaide' s metropolitan coast;
2. A comparison of the data with past observations (1996 and 1999) to determine whether there has been a shift in the structure of the biological communities associated with the metropolitan reefs;
3. Development and interpretation of a number of indices to assist in determining the status of reef health;
4. The establishment of baseline information for reefs in non-metropolitan areas (specifically Fleurieu and Yorke Peninsulas); and
5. A comparison of metropolitan with non-metropolitan reefs.

1.2 Previous studies of reef health in South Australia

Concern arising from the negative consequences of reef habitat degradation led to the establishment of a survey program in 1996, which aimed to assess the current 'health' of Adelaide's near shore metropolitan reefs (Cheshire *et al.* 1998a, Cheshire *et al.* 1998b, Miller *et al.* 1998[1]). The main objectives of the 1996 study included:

1. The collection of baseline reef health measurements;
2. Formulation of hypotheses surrounding reef community dynamics;
3. Identification of suitable protocols for monitoring future changes to metropolitan reefs.

The 1996 survey program considered six reefs along the metropolitan coastline from Semaphore in the north to Aldinga in the south. Results showed a distinct south to north trend, with southern reefs dominated by large Phaeophycean (brown) macroalgae, similar to reefs elsewhere on the South Australian coast. In contrast, northern reefs primarily comprised smaller (foliaceous and turfing) Rhodophycean (red) macroalgae. This gradient correlated with the level of urbanisation of the 47 km stretch of coast.

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[1] available from the Reef Watch website: http://www.reefwatch.asn.au/reports.html

While acknowledging the possibility that this trend might be a response to a natural gradient within Gulf St Vincent, Cheshire *et al.* (1998a) hypothesised that the lack of robust brown macroalgae on the more urbanised northern reefs was a sign of degradation linked to poorer water quality relative to less modified sites further south.

A second survey was commissioned in 1999 (see Cheshire and Westphalen 2000[2]) which aimed to determine whether the pattern observed in 1996 was the same or if it had changed. The 1999 surveys considered a larger number of reefs (9 compared to 6) but revealed a similar pattern to those in 1996 and 1999. Northern reefs remained ‘degraded’, but the cover of large brown macroalgae on the central and southern reefs had increased by 1999. It was argued by Turner (2004) that this increase was related to more favourable climatic conditions in 1999 relative to 1996, based on an examination of meteorological data (BOM 2002).

Of particular concern in 1999 was the large increase in mussel abundance (*Brachidontes rostratus*[3]) on a few central coastline reefs. The possibility that these mussels were restricting the subsequent recruitment of macroalgae was experimentally tested and confirmed at Horseshoe Reef (Smith 2000); however preliminary observations in 2001 (for an experimental mussel removal program funded by the Coast Protection Board) indicated that few mussels remained at Horseshoe Reef (pers. obs. Sue Murray-Jones).

1.3 The 2005 Reef Health survey program

The 1996 and 1999 Reef Health surveys have provided much useful information about the status of Adelaide’s metropolitan reefs. However, both surveys were somewhat limited in their spatial extent and the number of parameters considered (almost exclusively macroalgal cover), and there were confounding factors (in particular differences in depth) that make a robust interpretation of the data difficult. The 2005 survey addressed these issues, as well as completing an assessment of metropolitan reefs. The number of reefs surveyed was increased to include locations around Fleurieu and Yorke Peninsulas, incorporating a better coverage of different reef environments. Further, the information collected from each site has been expanded and/or modified to include data on fish communities, invasive taxa, sediment levels, and a greater emphasis has been placed on particular aspects of earlier surveys, such as the areal coverage of turfing algae and mussels.

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[2] Also available from the Reef Watch website: http://www.reefwatch.asn.au/reports.html
[3] This species was incorrectly identified in previous reports as *Xenostrobus pulex*.

2 Indices used for the interpretation of reef health

Temperate reef systems are often diverse and complex, and present a particular challenge when it comes to research and/or monitoring for management purposes. Traditionally, research and monitoring of temperate reef systems has relied on destructive methods (Turner 1995), but for investigations of reef status ('health'), such approaches are counterproductive. The indices developed for the present study are non-destructive, rigorous, and can be replicated, which is particularly important if community groups are to assist with reef-monitoring programs (Turner *et al.* 2006b). Non-destructive temperate reef monitoring is relatively new, compared with its counterparts in coral reef systems, and consequently the parameters employed to describe the nature of these systems will continue to evolve both in terms of composition and structure. The development of an overall value as an indicator of a reef's health (see methods) will also be refined.

Two workshops (16 June and 12 December 2005) were held to develop the approach to analysis, and the identification of potential indicators. In addition, a series of meetings were held, involving a range of local experts. Most participants agreed that a temperate reef might be broadly considered to have four fundamental components:

- abiotic factors (substrate composition, topography, wave action, etc)
- macroalgal cover and community structure
- invertebrate abundance and diversity
- abundance and diversity of fish and other higher order consumers

The indices developed for this project (Table 1) mainly focus on these components. Eleven indicators were eventually selected (Table 1). Details of the justification for their selection are included in the next section, while details of the way they have been applied are included in the methods (see Section 3). Different members of the Reef Health steering committee contributed a rationale for the various indices, and these authors are credited for the relevant part of Section 2.

Table 1. Eleven indices considered from the 2005 Reef Health survey

Index
Areal cover
Areal cover of canopy-forming macroalgae
Areal cover of turfing macroalgae
Areal cover of mussel mats
Areal cover of bare substrate
Abundance
Size and abundance of blue-throated wrasse
Abundance of site-attached fish
Abundance of mobile invertebrate predators
Presence
Presence of invasive taxa
Presence of high sedimentation
Species richness
Richness of macroalgae
Richness of mobile invertebrates

2.1 Areal cover of canopy-forming macroalgae

(written by David Turner, SARDI Aquatic Sciences[4])

Temperate reefs are often visually dominated by large Phaeophycean (brown) macroalgae that form dense, often closed, canopies. Within southern Australia, these dominants comprise members of the Fucales (*Scytothalia, Seirococcus, Cystophora* and *Sargassum*) and Laminariales (*Ecklonia* and *Macrocystis*). In South Australia, these canopy-forming macroalgae are well understood in terms of primary productivity (Fairhead and Cheshire 2004) and in creating habitat complexity in support of substantial faunal communities (e.g. Turner *et al.* 2006a). These taxa have also been shown to be susceptible to 'urbanisation' with several local studies discussing their loss from reefs under conditions of declining water quality (e.g. Cheshire and Westphalen 2000, Gorgula and Connell 2004, Turner 2004).

Given their importance on temperate reefs and their apparent susceptibility to common anthropogenic impacts (specifically water quality), the presence and cover of canopy-forming macroalgae is an ideal indicator of reef health.

The areal cover of these taxa is variable under natural conditions but generally exceeds 40% on reefs not exposed to anthropogenic inputs (Cheshire and Turner 2000, Cheshire and Westphalen 2000, Turner and Cheshire 2003, Turner 2004). Therefore, reefs with a percentage cover greater

[4] Currently Regional Conservation, DEH

than 40% received an index score of 100. Reefs exhibiting less then 40% cover are given a scaled index score between 0 and 100 (corresponding to 0 to 40% canopy cover respectively).

2.2 Areal cover of turfing macroalgae

(written by Tim Kildea, Australian Water Quality Centre)

Turfing algae are defined as a taxonomically complex assemblage of small filamentous or short fleshy algae (usually < 1 cm), but can also be composed of articulated or encrusting corallines (Wanders 1977, Borowitzka *et al.* 1978, Klumpp and McKinnon 1989). Turfs include early developmental/alternative life history stages of larger macroalgae, all primarily attached to rock or coral substrates (Copertino 2002). On most temperate reefs, turfs normally compete with kelps, fucoids and other large macroalgae for substrate (Kennelly 1983, Westphalen and Cheshire 1997). Turfing algae are highly productive (Copertino 2002) and are often the first algal group to settle and establish on cleared substrate, gradually being replaced (often via shading) by slower-growing foliose algae and then later by the larger kelps (*Ecklonia* and *Macrocystis*) and fucalian algae (e.g. *Cystophora* and *Sargassum*) (Kennelly 1983, 1987, Kennelly and Underwood 1993).

In terms of areal cover, turfing communities can be as abundant as larger macroalgae (Turner 1995, Fowler-Walker and Connell 2002). Turfs have a fundamental role in the colonisation of naturally disturbed areas (e.g. by wave action, sediment accretion, storms, intense grazing (Littler and Littler 1984, Sala and Boudouresque 1997, Airoldi 1998, Baynes 1999) or areas impacted by anthropogenic influences (e.g. dredging, sewage, power station waste, eutrophication and sedimentation) (Hatcher 1998, Aseltine-Neilson *et al.* 1999, Turner and Cheshire 2002, Gorgula and Connell 2004). Turfs are also recognised as important agents in the physical stabilisation of marine sediments, having the ability to trap sediment (Neumann *et al.* 1970). Turfs are thus an important component of healthy temperate reef communities, incorporating high levels of biodiversity and productivity that in turn support a wide range of fauna and act as an important early phase in macroalgal succession.

It has been hypothesised that the eutrophication and sedimentation of temperate nearshore coastal environments has led to an increase in the cover of turf-forming algae at the expense of canopy-forming algae (Airoldi *et al.* 1995, Benedetti-Cecchi *et al.* 2001, Eriksson *et al.* 2002, Airoldi 2003, Gorgula and Connell 2004, Russell *et al.* 2005). The ability of turfing algae to dominate is due to life-histories and physiologies better suited to high nutrient loading (Worm *et al.* 1999) and morphologies that enable them to dominate space under heavy sedimentation regimes (Airoldi and Virgilio 1998).

The mechanisms that switch subtidal habitats from canopy- to turf-dominated communities are not clearly understood, but the presence of large areas of turf on Adelaide's metropolitan coast correlates with areas that have been impacted by elevated levels of nutrients and/or sediments (Gorgula and Connell 2004, Russell *et al.* 2005).

A percentage cover of turfing algae greater than 25% is thought to be symptomatic of an impacted ecosystem (Gorgula and Connell 2004, Russell *et al.* 2005). The level of turf cover may provide an indication of the health of a reef, and reef systems dominated by turfing algae (>25% areal coverage) may be showing symptoms of poor health. Previous reef health reports (see Cheshire and Westphalen 2000) suggest that a percentage cover of turfs greater than 40% is indicative of a heavily impacted ecosystem.

Therefore, as a biological indicator, reefs with a percentage cover of turf ≥ 40% received a score of zero, representing a highly impacted reef, whereas a cover < 25% was coded as a 'null' value, as the occurrence of turfs at this level does not necessarily equate to good health. Percentage turf cover between 25 and 40% scored between 50 and 0 on a linear scale.

2.3 Areal cover of mussel mats

(written by Tim Kildea, Australian Water Quality Centre)

Mussels are a common component of temperate reef systems around the world (Edgar *et al.* 1997), coexisting and interacting with other species including macroalgae (Sousa 1979, Dethier 1984). Mussels are broadcast spawners, releasing planktonic larvae into the water column either seasonally or in response to specific stimuli (Suchanek 1978, Petraitis 1995). Juvenile mussels are able to respond to both chemical and physical cues to induce settlement onto empty substrate (Davis and Moreno 1995), and are able to grow rapidly to occupy large areas of the available primary space (Carroll and Highsmith 1996). Turfing algae are also a good substrate for secondary settlement of post-larval juveniles, and thus have the potential to promote mussel recruitment as well (P.J. Fairweather pers. comm.).

Previous studies have shown that mussels have the potential to dominate the substrate by blanketing the surface, excluding competitors (Lubchenco and Menge 1978, Petraitis 1995). Dominance is maintained either through high rates of recruitment or low rates of mortality, although growth can play an important role in determining the persistence of spatial dominance by a single species. Mussels can utilise an increase in growth rate to offset mortality and thus maintain a competitive advantage (Petraitis 1995), although this is dependent upon the availability of resources such as food and space.

Several mussel species are found on southern Australian reefs, but the two species of interest are *Xenostrobus pulex* and *Brachidontes rostratus*. Both species are small in size (<40 mm) and tend to live as dense, mat-forming populations on exposed rock platforms (Edgar 1997). Their habitat is generally confined to the intertidal zone but along the Adelaide metropolitan coast, *B. rostratus* has been observed growing to depths of 3 to 5 m (Smith 2000, Turner and Cheshire 2002)[5].

Reef Health surveys conducted by the University of Adelaide in 1996 and 1999 showed an increase in the percentage cover of *B. rostratus*[5] (1-2% to 15-25%) on a number of reefs. This was attributed to deterioration in water quality (increase in nutrients and sediment) in the region which affected macroalgae, leading to conditions favourable for mussel growth (Cheshire and Westphalen 2000).

The mechanisms that have resulted in the change from a macroalgal to mussel-dominated community are not clearly understood, but may be via one or more of several factors including: a change in the availability of limited resources (e.g. food or substrate); a trophic switch in predators from carnivores to herbivores; and/or a reduction in competitors.

Mussel species tend to be opportunistic (Petraitis 1995) and the reduction in macroalgal canopy cover may have provided an opportunity for mussels to establish and slowly dominate the available substrate. Mussels are filter feeders that rely on water circulation in order to obtain planktonic food that would include algal spores (Denny *et al.* 1985). Mussel filtration has the potential to greatly reduce the abundance of algal spores in the water column, limiting the capacity for algae to settle and compete with mussels for space (Santelices and Martinez 1988).

The surface heterogeneity of the mussel bed also provides secondary substrate for a variety of herbivorous grazers, which have the potential to feed on macroalgal spores, resulting in major impacts on algal assemblages (Lubchenco and Menge 1978, Schiel *et al.* 1995). Equally, the presence of a significant macroalgal canopy can adversely affect juvenile mussel settlement, with the constant sweeping of the substrate surface by the macroalgal thallus physically dislodging mussels (Chapman 1995).

Whether an ecosystem dominated by mussels is any less healthy than one covered with macroalgae is a point of conjecture. However, a substantial switch in community structure from macroalgae to mussels over a timeframe of a few years should be a cause for concern. Consideration of previous Reef Health survey data indicates this change is likely to occur when mussel cover is in the vicinity of 15 to 30% (Cheshire and Westphalen 2000, Turner and Cheshire 2002). As the cover of mussels increases, macroalgae are less likely to settle and grow, leading to a cycle of flora loss and ultimately, a significant change in the trophic structure of the reef.

Using mussel cover as a biological indicator, reefs with a percentage cover ≥ 30% received a score of zero, representing a highly impacted reef. A percentage cover < 15% recorded a 'null'

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[5] The species was wrongly reported as *Xenostrobus pulex*.

value, as the absence of mussels does not necessarily equate to good health. Percentage cover between 15 and 30% scored between 50 and 0 on a linear scale.

2.4 Areal cover of bare substrate

(written by Sam Gaylard, Environment Protection Authority)

Competition for space within Australian temperate reef systems has been a topic of research for some time (Kay and Keough 1981, Keough 1984, Butler and Chesson 1990). Most benthic substrate is covered by a diversity of organisms, which compete for space by overgrowing, undercutting, poisoning and mechanically removing competitors (Butler 2000). Cleared patches are rapidly occupied, and space is often considered to be a limited resource on temperate reefs (Connell and Keough 1985, Butler and Chesson 1990).

Substrata in marine ecosystems are constantly awash with a soup of propagules or larvae from reproductive organisms (Butler 2000). Any fresh, completely clean, substrate will be covered in a fine biofilm of microalgae and bacteria within hours (Jesus *et al.* 2006). Apart from a biofilm cover, persistently free habitable substrate may be indicative of an on-going disturbance, either natural (e.g. sand abrasion, high water movement, grazing pressure) or anthropogenic (e.g. release of toxicants, high turbidity), which disrupts natural recruitment processes. The presence of bare substrate is not in itself necessarily a cause for concern as natural disturbances that create free space are common. The protracted persistence or expansion of large areas devoid of marine life (other than biofilms) however suggests that underlying processes affecting natural recruitment and settlement have been disrupted. A high level of bare space on a reef or, in particular, a persistent trend of increasing bare substrate over a number of years, may be indicative of a 'disturbed' system. Bare substrate is thus a potentially useful indicator to monitor reef health.

Data collected from past Reef Health surveys (Cheshire and Turner 2000, Cheshire and Westphalen 2000, Turner 2004) indicate that impacted reefs along the metropolitan coast have, on an areal basis, a high proportion of persistently bare substrate (generally > 20 %). Thus, an areal cover between 0 and 20% received a null value, reflecting the argument that the presence of space does not necessarily represent an unhealthy reef. The lower and upper threshold values were set at 20% and 40% respectively, with the index score linearly scaled between 50 and 0. Reefs with an areal cover > 40% were given a score of zero, indicating a highly disturbed ecosystem.

2.5 Size and abundance of blue-throated wrasse, *Notolabrus tetricus*

(written by Scoresby Shepherd, SARDI Aquatic Sciences)

The blue-throated wrasse is a site-attached species with high fidelity to a home range of 2 - 3000 m^2. The species occurs ubiquitously on exposed and semi-exposed reefs, but does not penetrate far into the Gulfs (northern limits were thought to be Christies Beach in Gulf St Vincent and Cape Elizabeth in Spencer Gulf).

Visual censuses conducted between 10 am and 4 pm at one site recorded an average of 62% of the total population (i.e. at any one time a diver doing a fish census will on average see only 62% of the total numbers present). The remainder are cryptic or in shelter. Abundance of wrasses on any reef is dependent mainly on rocky bottom relief (i.e. low reefs have fewer fish compared to reefs that have a higher relief). The diet of the species is cryptic reef invertebrates (Shepherd and Clarkson 2001).

Two measures, total length of blue-throated wrasse adults and mean size alone, would be expected to give a reliable index of total biomass and an estimate of fishing intensity, respectively. This is based on the premise that fishers selectively target larger fish. Reef health could be inferred either from comparative studies with control reefs or from temporal studies of changes in abundance. Ideally, both studies should be conducted to gain a greater understanding of the mechanisms that control population size.

The effect of anthropogenic influences was of greater interest to the present study and hence only the measure total length was used as the index to quantify reef health. The total length of blue-throated wrasse was calculated by summing the lengths of individual adults (>15 cm) at each site, standardised to per metre value.

Any reef for which the presence of blue-throated wrasse was not recorded received a null value as absence does not necessarily relate to poor health but may be indicative of a reef that is outside the accepted range of the species. A deficiency of the index is that the abundance of the species naturally decreases along the northern limits of its distribution (e.g. Christies Beach for Gulf St Vincent), with the result that reefs in this region may receive a low score for blue-throated wrasse. The low score may not necessarily reflect an anthropogenic impact, but is a result of limits in the species natural distribution. Fish surveys were generally done only when visibility was sufficient (see Appendix A).

2.6 Total abundance of site-attached reef fishes

(written by Scoresby Shepherd, SARDI Aquatic Sciences)

Site-attached species of fish are herbivores, omnivores or carnivores, which feed variously on macroalgae, turf algae and invertebrates (see Appendix C). Some planktivorous species (e.g. sweep) are also associated with reefs, but the extent to which they are site-attached is poorly understood. Within a healthy reef there are normally 10 - 15 such site-attached fish species with total abundances of up to 0.2 m^{-2}, and an overall abundance more or less linearly related to rocky bottom relief (see Shepherd and Baker 2006). This would suggest that similar reef topography should have similar abundances in site-attached fish. The presence and/or abundance of the site-attached fish species might be expected to reflect the overall health of the reef and fishing intensity combined, subject to reservations stated in the previous section (see above). Total length was calculated by summing the lengths of individual adults (>15 cm) at each site, standardised to per metre value.

2.7 Abundance of mobile invertebrate predators

(Kirsten Benkendorff, Flinders University of South Australia)

Predators are useful indicators of reef health because, being at the top of the food chain, they can only persist in significant numbers in areas where there is substantial primary and secondary productivity. Thus, production levels for top marine predators can be indicative of the maximum potential yield of the ecosystem (Robertson and Hatcher 1994). Unhealthy or heavily impacted ecosystems often show a reduction in the length of the food chain with predators being the first to disappear (Estes 2005, Steneck and Sala 2005).

Predators can also be important in structuring reef communities. Large generalist predators, such as seastars, whelks and large decapod crustaceans, can be important determinants of the distribution and abundance of many reef species (Paine 1974, Chilton and Bull 1984, Fairweather 1985, Barkai and McQuaid 1988, Fairweather 1988, Estes 2005, Steneck and Sala 2005). Indeed, some reef predators have been described as 'keystone species' due to their crucial influence on the structure and composition of assemblages (e.g. Dayton 1971, Paine 1974). Whilst not all invertebrate predators are necessarily 'keystone species', they can play some role in moderating the effects of competitive or other interactions amongst their prey and usually have localized effects on prey abundance and size distributions. By altering the patterns of prey domination, predators can influence the recruitment of both sessile and mobile species on reefs. This may result in enhanced patchiness in the spatial and temporal patterns of occupancy of the substratum (e.g. Fairweather 1988) or even alternative stable states in the communities (e.g. Barkai and McQuaid 1988, Petraitis and Dudgeon 2005). The exclusion of predators has been shown to lead to lower diversity in reef communities, suggesting that predators help maintain epifaunal communities in a high diversity state (Paine 1974, Russ 1980), although effects can be subtle (Keough and Butler 1979).

The total invertebrate predator abundance was selected to represent productivity at this top trophic level rather than numbers of any one predator. This is because there is a lot of variation in species composition between reefs (i.e. there is likely to be replacement of specific taxa under different environmental conditions) and, to date, no particular predator species has been shown to function as a 'keystone species' in SA (Keough and Butler 1979). Invertebrate predators are relatively large and conspicuous compared to the richer diversity of cryptic grazers and detritivores. Thus, the abundance of all invertebrate predators should provide a reliable measure of the integrity of the food chain on local reefs. To improve the repeatability of this measure between researchers, juveniles and predator species < 5 cm in size were excluded to decrease the variability due to experience or uncertain capabilities in discriminating amongst smaller specimens. Counts of purely predatory species could also be combined with those for omnivores, scavengers, and 'browsing carnivores' to represent total energy flow from animals to all consumers.

This indicator uses the mean abundance of all mobile invertebrate predators taken from quadrats at each site, then ranked sites on a continuous scale from 0 - 100, such that a higher abundance of predators indicated greater integrity in the reef communities and lower numbers or an absence of predators indicated a less healthy reef. This indicator is only functional with native species. The presence of invasive invertebrate taxa should not be recorded for this index (e.g. *Asterias amurensis*)

2.8 Presence of invasive taxa

(written by Sue Murray-Jones, Coastal Protection, DEH)

Coastal and marine habitats are among the most heavily-invaded ecosystems in the world and until recently there has been limited research on how invasive species affect these habitats (Grosholz 2002). The ecological consequences of an invasive species to an ecosystem can be extensive and include: competitive displacement; impacts on growth, survivorship and reproduction; and a range of impacts at community and ecosystem levels (see review by Grosholz 2002). The establishment of opportunistic and exotic taxa can drastically change the structure of marine communities, reducing overall biodiversity within an ecosystem (Boero and Guidetti 2004).

There is evidence to suggest that anthropogenic impacts, such as industrial discharges and overfishing, influence the susceptibility of an ecosystem to the invasion of opportunistic species (Levine 2000, Harris and Tyrrell 2001). Exotic and 'feral' species are often opportunistic, and capable of rapidly adapting to changes that occur to an ecosystem because of an anthropogenic impact.

Once established, pest species are difficult to control. Early warning of the presence of pest species has the potential to considerably reduce the cost of control and eradication. Bio-economic risk assessment models suggest that preventing invasions by pest species may be economically beneficial, even when the cost of preventing the invasion is high (Leung *et al.* 2002). Therefore monitoring for the presence/absence of marine pests has additional benefits, well above their utility as an indicator of health.

As an indicator of reef health, the presence of an invasive or feral species at a reef receives a score of zero (representing poor health), whereas an absence of these species receives a null value. The indicator score is not cumulative if more than one invasive species is observed growing on a reef. The 'null' value reflects the argument that the absence of a pest species does not necessarily imply that the reef is healthy. The reef can be 'unhealthy' without the presence of invasive or feral species. It must also be noted that this index is only valid if the invasive taxa is not present in "plague" abundances.

There may also be an inability to detect or recognise the presence of an invasive or feral species, even though it is growing on a reef. The National Introduced Marine Pest Information System (NIMPIS, Hewitt *et al.* 2002) lists 43 exotic organisms for South Australia, comprising many species that are small, innocuous and/or difficult to differentiate from natives. Consequently, only eight marine pests and two nuisance species were considered (Appendix A).

2.9 Presence of high sedimentation

(written by David Turner, SARDI Aquatic Sciences[6])

Sedimentation is a natural process whereby fine particulate matter (sand, silt and organic debris) is carried in suspension within the water column and then deposited. Although a natural process, anthropogenic activities have led to increasing rates of sedimentation and this process is now considered to pose a threat to marine ecosystem function (United Nations Environment Program 1995).

In South Australia, coastal development and catchment modification have both led to increasing rates of sedimentation, with the situation further exacerbated by sediment (re)mobilisation due to seagrass loss (Fotheringham 2002).

Rocky reef environments are very susceptible to sedimentation, the effects of which are reviewed by Airoldi (2003), who identified numerous studies wherein elevated levels of sedimentation led to dramatic changes to reef assemblages. Increased sedimentation was found to exert an influence on the composition, structure and dynamics of benthic communities. Under heavy sediment loads, reef assemblages generally shifted towards more sediment-tolerant algal species and suspension feeders such as mussels. On Adelaide's metropolitan coast, recent research has generally found that high sedimentation leads to dominance by more adventitious species, such as turfs, at the expense of the larger macroalgal taxa (e.g. Connell 2003, Gorgula and Connell 2004, Turner 2004, Connell 2005).

Knowledge of sediment dynamics is therefore important for sustainable management of reef assemblages, particularly those adjacent to coastal developments (Airoldi 2003). However, a detailed investigation of sedimentation levels requires a substantial commitment in resources that was beyond the scope of the present study. Nonetheless, a snapshot estimate of the amount of

[6] Currently Regional Conservation, DEH

sediment present at each site was possible, with data scaled from absent through to high. Although somewhat coarse, reefs that recorded 'high' sedimentation scored zero, which represents a negative impact. All other sediment estimations recorded a 'null' value, as it was not clear whether the levels observed were capable of causing an ecological shift in the structure of the reef community.

2.10 Species richness of macroalgae; species richness of mobile invertebrates

(written by Grant Westphalen, SARDI Aquatic Sciences)

The species richness of a community may be both an indicator of the health of an ecosystem as well as an influence on the health of that system. Healthy ecosystems tend to support a large number of species, which in turn assists healthy system function by increasing stability (Worm and Duffy 2003).

Species richness can be a contentious index to use for the spatial comparison of different reefs. Reefs vary in many ways, all of which are likely to vary across geographic scales (Naeem *et al.* 1999, Gawn 2004). At best, these variations add noise to the results, obscuring the reality of the situation. At worst, they confound the interpretation, leading to spurious conclusions about the 'health' of the reef, which are in fact a reflection of natural physical and biological conditions. With a limited choice of sites making adequate spatial replication problematic, this may be an issue. For example, discovering that putatively impacted reefs have lower species richness may be a natural result of lower species richness in the area. This is essentially a confounding factor, particularly if the supposedly impacted sites are clustered within a relatively limited geographic area and isolated from sites that might act as controls. If a region has a naturally limited species pool then the sites within, regardless of their 'health' or level of exposure to external influences, are likely to demonstrate lower species richness. Particular care is thus required in the interpretation of data, given the nature of impacts reported in previous surveys.

A potentially less problematic approach would be to use species richness as an indicator of change (i.e. a temporal comparison within sites). Natural temporal variation in biological systems is often less significant than geographic variability (e.g. Collings 1996), although whether this holds for species richness is unknown. Secondly, whilst there is temporal variability, much of it is seasonal with annually repeated patterns (Collings 1996) for which allowances need be made in the experimental design (i.e. sampling in a given month each year).

It is worthy of note that much of the debate about the stability of an ecosystem centres on the interactions involved between species of different trophic levels. Thus, assessment of the diversity of a single group, such as macroalgae, may be misleading because a restricted range of interactions is possible (i.e. largely competition). Having said this, many of the hypotheses that explain the apparent link between species richness and stability focus on the ability of one species to replace another if the original is removed through disturbance. In this way, richness within a group is important because it is a measure of the number of species that may be lost before that niche is left empty and ecosystem function is subsequently altered.

Biodiversity amongst reefs is highly variable, even amongst those considered to be unimpacted (e.g. Collings 1996, Collings and Cheshire 1998, Shears and Babcock 2004). Many healthy reefs in southern Australia are dominated by monospecific stands of the relatively small kelp, *Ecklonia radiata* (Collings 1996, Shears and Babcock 2004, Turner 2004), which tends to associate with encrusting coralline species, but lacks turfing species (Edwards 1998, Fowler-Walker and Connell 2002, Copertino *et al.* 2005). This situation poses problems for using species richness as an index of health , as the decline in health of an *Ecklonia*-dominated system my be manifest as an increase in biodiversity as the canopy dominants are replaced by a species-rich turfing community.

Diversity within a functional group (e.g. macroalgae or mobile invertebrates) has the potential to indicate ecosystem health through the provision of redundant species that can fill an ecological niche should the 'usually-dominant' species disappear. In addition, it can be argued that under conditions of high disturbance, it is usually a few species that are able to take advantage, and as a result, low species richness will be evident. Similarly, the advent of an invasive species can result in a loss of diversity. However, despite the truth of these propositions, it is undeniable that there are a great many other natural factors that determine species richness. This is amply demonstrated by the high degree of variability of species richness across sites that are largely not influenced by anthropogenic activity. In the light of this variability, and the fact that the present study has demonstrated that apparently healthy reefs demonstrate low diversity within the macroalgal assemblage, the use of within-group taxa richness should be applied with great caution as an indicator of reef health.

Considering the above arguments, a degree of caution was applied to interpretation of the data, particularly for macroalgal species richness. Canopy macroalgal species such as *Cystophora* and *Ecklonia* reduce species richness due to the simple fact that they visually dominate the Line Intercept Transect assessment of the reef (see Appendix A) and thus reduce the species richness score. Macroalgal species richness was therefore calculated using both quadrat and LIT data. To counteract the bias due to the visual dominance of canopy species, sites with a high proportion (e.g. >40%) of Fucoids and Laminariales, received a maximum score. Invertebrate and macroalgal species richness was ranked between 0 and 100, where 0 represents poor health.

2.11 Presence/absence of rare species

One indicator that could be used is the presence of rare and/or endangered species; however, indices of this nature are of limited use as indicators of reef health. By their very nature, rare species are difficult to find. For example, a rare (and cryptic) species is the Harlequin fish (*Othos dentex*). This species was originally considered for use as an indicator of reef health. Anecdotal evidence suggested that this fish was found throughout much of South Australia. During the 2005 surveys, however, only one sighting of this species was recorded, at West Island. In the absence of *a priori* data and/or a consistent model describing the components of a reef, it is difficult to conclude whether the absence of a particular species provides any indication of the status of a reef, particularly for cryptic species. Thus, the Harlequin fish was excluded from the indices describing reef health, and the presence/absence of rare species was not used as an index of reef health.

3 Methodology

3.1 Survey sites

From January to May 2005, 39 sites over 31 different reefs were surveyed across the Adelaide Metropolitan, Fleurieu Peninsula and Yorke Peninsula regions. This compares to 8 sites over 6 reefs in 1996 and 17 over 9 reefs in 1999 (Cheshire *et al.* 1998b, Cheshire and Westphalen 2000).

3.2 Adelaide Metropolitan Survey

Sites along the Adelaide Metropolitan coastline (Table 2; Figure 1), surveyed during previous Reef Health studies (Cheshire *et al.* 1998a, Cheshire and Westphalen 2000), were re-surveyed in the 2005 program. Additional sites were also included to provide better spatial coverage and to encompass a broader range of depths, which was a major limitation of previous Reef Health surveys. Nineteen sites on 11 reefs along the Adelaide metropolitan coastline were surveyed (Table 2; Figure 1). Sites were selected to complement previous surveys, and were chosen to encompass a range of reef types as well as to provide a broad geographical spread across the region (see descriptions in Appendix D). Given the importance of depth as a determinant of community structure (Shepherd and Sprigg 1976), sites were divided into deep (8 m+) and shallow (c. 5 m).

Table 2. Survey locations of the Adelaide metropolitan reefs. Coordinates are based on the WGS84 datum.

	Site name	Abbreviation	Zone	Depth (m)	Latitude (°)	Longitude (°)
Deep sites (8+ metres)	Semaphore Reef ♠⊗	SEM	North	8	34 50.826	138 26.757
	Broken Bottom ♠⊗	BRB	North	9	34 57.801	138 28.817
	Glenelg Barge ♠⊗	GBG	North	16	34 58.745	138 26.398
	Glenelg Dredge ♠⊗	GDG	North	16	34 58.851	138 26.445
	Seacliff Reef	SEA	Central	12	35 02.398	138 29.491
	Noarlunga Deep ♠⊗	NDP	Central	10	35 09.160	138 27.884
	Aldinga Deep ♠⊗	ADP	South	10	35 16.296	138 25.859
Shallow sites (c. 5 metres)	Glenelg Blocks	GBL	North	5	34 58.706	138 30.198
	Hallett Cove ♠⊗	HAL	Central	5	35 04.418	138 29.661
	Horseshoe Inside ⊗	HSI	Central	5	35 08.276	138 27.775
	Horseshoe Outside ⊗	HSO	Central	5	35 08.365	138 27.483
	Noarlunga North Inside ⊗	NNI	Central	5	35 08.930	138 27.695
	Noarlunga North Outside ♠⊗	NNO	Central	5	35 08.849	138 27.782
	Noarlunga South Inside ⊗	NSI	Central	5	35 09.420	138 27.979
	Noarlunga South Outside ⊗	NSO	Central	5	35 09.415	138 27.925
	Southport ⊗	SOU	Central	4	35 10.065	138 27.736
	Moana Inside ⊗	MSI	South	5	35 12.551	138 27.863
	Moana Outside ⊗	MSO	South	5	35 12.390	138 27.733
	Aldinga Shallow ♠⊗	ASH	South	5	35 16.254	138 25.971

♠ Indicates sites previously surveyed in 1996
⊗ Indicates sites previously surveyed in 1999

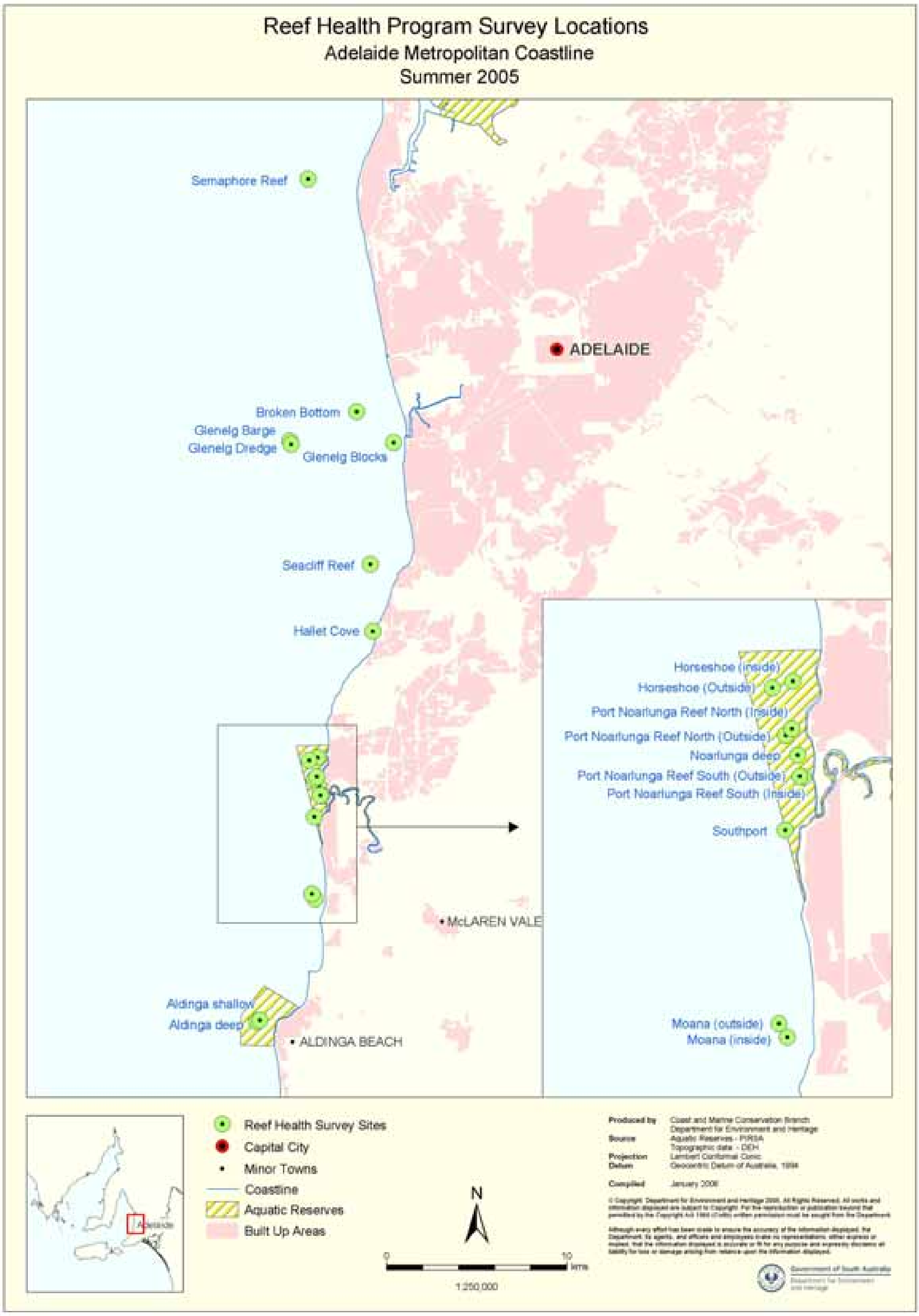

Figure 1. Adelaide Metropolitan reefs surveyed during the 2005 field program. 19 sites on 11 reefs were considered. Map produced by Coast and Marine Conservation Branch Department for Environment and Heritage.

3.3 Fleurieu Peninsula Regional Survey

Non-metropolitan sites were chosen to provide a representative coverage across the Fleurieu and Yorke Peninsula regions (Figure 2, 3). Site choice was determined by considering several pragmatic issues associated with locating reefs at suitable depths (for purposes of comparison), and weather condition at the time of each survey.

The Fleurieu Peninsula regional surveys incorporated four sites within Gulf St Vincent, south of the Adelaide Metropolitan area (i.e. not including Aldinga), and four sites on the other side of the Peninsula within the Encounter Bay area (Table 3, Figure 2). None of the eight Fleurieu sites were evaluated in previous Reef Health surveys. Site descriptions appear in Appendix D.

Table 3. Survey locations of Fleurieu Peninsula reefs. Coordinates are based on the WGS84 datum.

Reef name	Abbreviation	Depth (m)	Latitude (°)	Longitude (°)
Carrickalinga	CAR	5	35 25.519	138 19.235
Second Valley	SEC	5	35 30.583	138 12.889
Cape Jervis North	CJN	5	35 34.724	138 06.827
Cape Jervis South	CJS	5	35 38.035	138 06.620
West Island	WEI	5	35 36.327	138 35.559
The Bluff	BLU	5	35 35.332	138 36.332
Granite Island	GRA	5	35 33.970	138 37.679
Port Elliot	PTE	5	35 32.263	138 41.555

Figure 2. Fleurieu Peninsula reefs surveyed during the 2005 field program. Eight sites were considered. Map produced by Coast and Marine Conservation Branch Department for Environment and Heritage.

3.4 Yorke Peninsula Regional Survey

Eleven sites were sampled on both sides of Yorke Peninsula, to include both Gulf St Vincent (5 sites) and Spencer Gulf (6 sites, Table 4, Figure 3). As with Fleurieu Peninsula, this is the first time these sites have been surveyed for health. See Appendix D for full site descriptions.

Table 4. Survey locations of Yorke Peninsula reefs. Coordinates are based on the WGS84 datum.

Reef name	Abbreviation	Depth (m)	Latitude (°)	Longitude (°)
Edithburgh Pool	EDP	4	35 04.989	137 44.926
Troubridge Point	TRP	5	35 10.173	137 40.443
Point Yorke	YKP	5	35 13.853	137 11.289
Marion Bay	MAB	5	35 15.185	136 58.892
Cable Hut Bay	CAH	5	35 17.620	136 53.892
Corny Point	CPT	5	34 53.671	137 01.133
Point Souttar	PTS	4	34 53.607	137 14.803
Wardang Island	WAI	5	34 32.171	137 21.359
Goose Island	GOI	5	34 27.183	137 22.062
Cape Elizabeth	CEL	4	34 10.779	137 27.861
Point Riley	PTR	4	33 52.579	137 35.915

Figure 3. Yorke Peninsula reefs surveyed during the 2005 field program. Eleven sites were considered. Map produced by Coast and Marine Conservation Branch Department for Environment and Heritage

3.5 Surveys

Four 50 metre transect lines were haphazardly located at each site. On each transect, SCUBA divers conducted a standard set of sampling protocols (Appendix B), consisting of:

- habitat descriptions incorporating information on reef topography, environmental conditions, and dominant life forms in the area of the transect;
- a pelagic fish survey, in which length and species were recorded for all fish within a 5 m belt of the transect;
- cryptic fish and large mobile invertebrate survey within a 1 m belt from the transect;
- a Line Intercept transect (LIT) recording benthic assemblages over the first 20 m of the transect;
- quadrats (0.25 m^2) recording additional benthic information at 2, 5, 8, 11, 14, 17, 20, 30, 40, and 50 m along the transect;
- an invasive species survey specifically searching for the presence of a range of known invasive taxa across the area.

Data from the four transects were pooled prior to analysis.

Complete descriptions of the sampling regime are presented in Appendix A, with the standard survey protocols in Appendix B. Complete site descriptions are in Appendix D.

3.6 Data manipulation - index calculation

Data collected during the field surveys were manipulated according to the following protocols to produce a raw value for each indicator. These values were then compared against threshold levels and scaled to produce a final index score.

3.6.1 Indices of areal cover

Four areal cover indices were considered including canopy macroalgae, turfing macroalgae, mussel mats, and bare substrate. Other than macroalgae, all of the indices based on cover (turf algae, mussels and bare substrate) are to some degree affected by the layered structure of the community, which makes them less useful - these measures are underestimated at sites with dense canopy. While this has been countered by leaving all sites below threshold levels with ‘null’ responses, the actual information value of these parameters is reduced, as they were only employed for a few reefs.

Areal cover values were derived from LIT data, with percent cover determined using Equation 1.

Equation 1. Conversion of LIT data to percent cover

$$\text{Percent cover of Index } A = \frac{\sum L_A}{Y - \sum D} \times 100$$

Where: $\sum L_A$ is the sum of the individual lengths of Index *A* on the LIT transect,
Y is the total length of the transect,
$\sum D$ is the sum of the lengths for which no data were recorded

Data from each transect at a site was pooled to produce a mean percent cover value for each indicator at each site, and these raw values were used in subsequent calculation of the scaled final index score.

3.6.2 Indices of abundance

The fish species considered as being site-attached are listed in Appendix C (Table 16). Abundance of site-attached fish was based on fish survey data expressed as average number per square metre. Abundance values for each site were converted to an index using Equation 2. Any values >100 were considered equal to 100.

Equation 2. Calculation of index of abundance

$$\text{Index of abundance } I = 50 \times \left(\frac{A - \min(Abund)}{\text{median}(Abund)} \right)$$

Where: *A* is the average abundance at any particular site, and min (Abund) and median (Abund) are the minimum and median abundances recorded from all sites respectively.

This equation functions in the context of the data set used to assess reef health in South Australia. Caution needs to be taken if this equation is translated to other data sets to calculate an index of abundance without first testing the validity of the model to local conditions.

Mobile invertebrate predators encountered during the surveys are listed in Appendix C (Table 17). Calculation of the index of abundance for mobile invertebrate predators was based on data obtained from the invertebrate transect expressed as average number per square metre. Calculation of the index follows the same procedure as employed for site-attached fish.

The raw value for total length of blue-throated wrasse was calculated by summing the lengths of individual adults (>15 cm) at each site, standardised to a per metre value. Calculation of the index from this value followed the same procedure as for fish abundance.

3.6.3 Binary indicators

Two binary indices were used in the study: presence of invasive species; and high levels of sedimentation. Data underpinning each of these were extracted from the fish, invasive species, and habitat surveys (note that fish surveys also considered the presence of benthic exotics, see Appendix B). For each index, sites were given a raw value of zero when the indicator was present; otherwise, a null score was recorded.

3.6.4 Indices of species richness

Macroalgal species richness was based on a combination of data obtained from the line intercept transects and quadrats. Mobile invertebrate species richness was calculated from a combination of the invertebrate transect and quadrat survey data. In both cases, raw species richness for each site was converted to an index using the method employed for fish abundance (Equation 2).

If the scaled areal cover index for macroalgae was at the maximum value (100), the corresponding species richness index was scaled to the highest score (100). Otherwise, high macroalgal cover might have prevented species from being observed, and hence resulted in an underestimate of species richness (see Section 2.10).

3.6.5 Scaling of indices

Upper and lower threshold values were determined for each index based on available information and expert advice (see Section 2). Appropriate values for the index at each of these thresholds were then determined based on ecological significance. Thus, for indicators in which higher raw scores imply better 'health', upper thresholds corresponded to a maximum index value (Table 5), with the opposite occurring for negative indicators. Under certain circumstances, it was not appropriate to give a score for an indicator and in these cases; a null value was recorded. For example, a large amount of bare substrate (>40) was considered to be indicative of 'poor' condition whereas the reverse (small areas of bare substrate) was not taken to necessarily indicate 'good' condition.

For each indicator, raw figures matching or falling outside of the threshold range were given values as defined in Table 5. In the absence of any quantitative basis for the relationship between raw values and their respective health index, where raw figures lay between the lower and upper threshold values the index score was linearly scaled between the corresponding lower and upper values (Figure 4).

Table 5. Critical thresholds and index parameters used for scaling the indices of reef health

Index name	Threshold Lower	Threshold Upper	Index value <Lower	Index value Lower	Index value Upper	Index value >Upper
Areal cover indices						
Areal cover of canopy macroalgae	0	40	NA	0	100	100
Areal cover of turfing macroalgae	25	40	Null	50	0	0
Areal cover of mussel mats	15	30	Null	50	0	0
Areal cover of bare substrate	20	40	Null	50	0	0
Abundance indices						
Abundance of site-attached fish	0	Median	NA	0	100	100
Abundance of mobile invertebrate predators	0	Median	NA	0	100	100
Abundance of blue-throated wrasse	0	Median	NA	0	100	100
Presence indices						
Presence of invasive taxa	0	1	Null	Null	0	0
Presence of high sedimentation	None	High	Null	Null	0	NA
Species richness indices						
Richness of macroalgae	0	Median	NA	0	100	100
Richness of mobile invertebrates	0	Median	NA	0	100	100

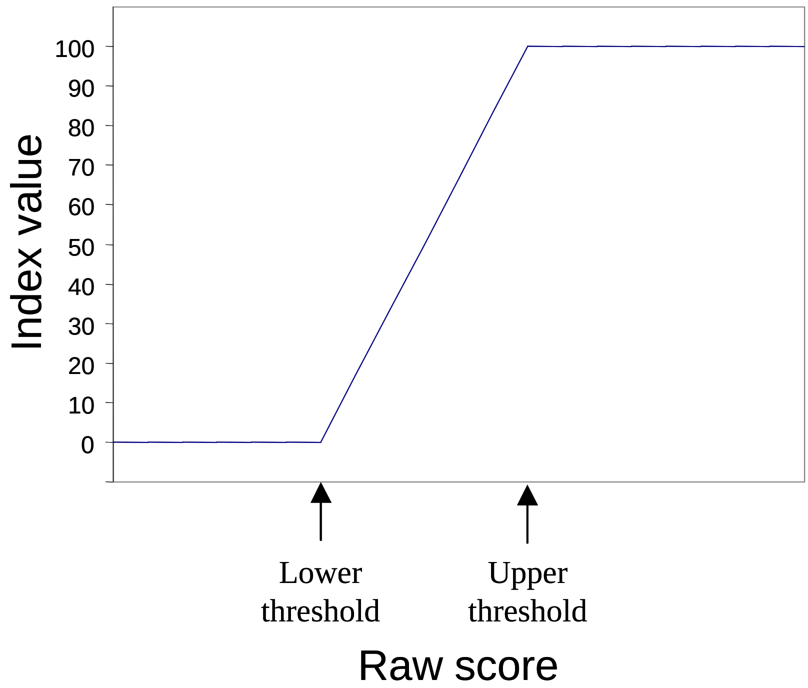

Figure 4. Example scaling of a positive index (a negative index would be the mirror of this plot). Raw figures less than the lower threshold received the minimum value (0), between the two thresholds the index was scaled linearly between the minimum and maximum (100), and for raw scores greater than the upper threshold the maximum index value was recorded.

3.5.6 Overall index of reef health

For each site, all of the non-null indicators were averaged to produce a single composite score, ranging between zero and 100. This score provided a relative measure of health in that sites with higher scores were considered to be in better condition than those with low scores. Reef health was set at three break points: Poor Condition (0-34); Caution Recommended (35-65); and Good Condition (66-100). Inclusion of the intermediate classification (Caution Recommended) highlights reefs that may be in a state of flux, but should not necessarily be allocated to the Poor Condition category. For example, a site with a markedly lower macroalgal cover may be a result of either an anthropogenic or a natural disturbance, or might be located in an area of very different physical characteristics. Reefs in the Caution Recommended category should be the focus of further monitoring and research.

3.7 Data analyses

Analyses comprised three broad sections:

1. Changes to Adelaide metropolitan reefs, 1996- 2005
2. Composition and status of Fleurieu Peninsula reefs in 2005
3. Composition and status of Yorke Peninsula reefs in 2005

An examination and re-interpretation of the various indices developed from the 2005 survey was also undertaken.

3.7.1 Changes to Adelaide metropolitan reefs 1996 - 2005

An examination of changes to Adelaide metropolitan reefs was made through direct comparison of the 2005 survey data with that collected previously by Cheshire *et al.* (1998a) and Cheshire and Westphalen (2000). The previous Reef Health surveys were limited to data collected using the LIT methods, restricted to functional group classification. To allow for direct comparison with previous surveys, 2005 survey data were distilled into a set of standard reporting codes based on structural characteristics to give six functional groups (see Appendix C, Table 15):

- Robust brown algae
- Foliaceous brown algae
- Foliaceous red algae
- Turfing & encrusting algae
- Animals
- Bare substrate

Sites were divided into deep (8+ m), and shallow (5 m), and within each group the relationship between sites / surveys examined using ordination. The relative abundance of each of the six functional groupings was graphed cumulatively for each site and survey date.

Changes to sites over time were then examined by calculating the difference in abundance for each functional group for two surveys (1996 – 1999 and 1999 – 2005), based on Collings (1996, Chapter 5). The changes in assemblage composition between the two periods were plotted on an ordination. Classification analyses were used to identify groups of sites with similar temporal trends.

Ordinations used in the analysis were generated through non-metric multi-dimensional scaling (nMDS), based on a Bray-Curtis similarity matrix of the raw data. Group classification employed a hierarchical agglomerative clustering algorithm using the average similarity of individual nodes. Groups were determined based on similarity of between 40 and 85%, based on the groupings from the associated nMDS plot. Statistical computation was undertaken using a combination of the R Statistical package (R Development Core Team 2004) and PRIMER Version 5 (Clarke and Gorley 2001).

3.7.2 Status of Fleurieu and Yorke Peninsula reefs in 2005

In the absence of prior information from the Fleurieu and Yorke Peninsula sites, a comparison of all sites from the 2005 survey allowed interpretation of the data for each of the indices. This allowed the metropolitan reefs to be placed in context with those elsewhere on the South Australian coast, as well as giving a broader understanding of what algal compositions might comprise healthy reefs. The various indices used to classify reef status are also described for each zone (Fleurieu, Yorke and Metropolitan), which ultimately led to a review of the usefulness or otherwise of each index as an indicator of reef status.

4 Results

4.1 Reef surveys along the Adelaide metropolitan coastline

4.1.1 Structure of Adelaide metropolitan reef assemblages 1996 – 2005

The biotic composition of reefs along the metropolitan coastline from the 2005 survey was correlated with their geographic position in a similar way to that reported in the 1996 and 1999 surveys (Cheshire *et al.* 1998a, Cheshire and Westphalen 2000).

Spatial associations

Differentiation of the northern metropolitan reefs was most noticeable for comparisons involving the deep reefs (8 m+), with five groups being evident from the classification (Figure 5 A – E; Stress = 0.08). Where data from previous Reef Health surveys were available, they were included (1996 data based on Cheshire *et al.* 1998a, and 1999 data based on Cheshire and Westphalen 2000) alongside the 2005 data (Figure 5).

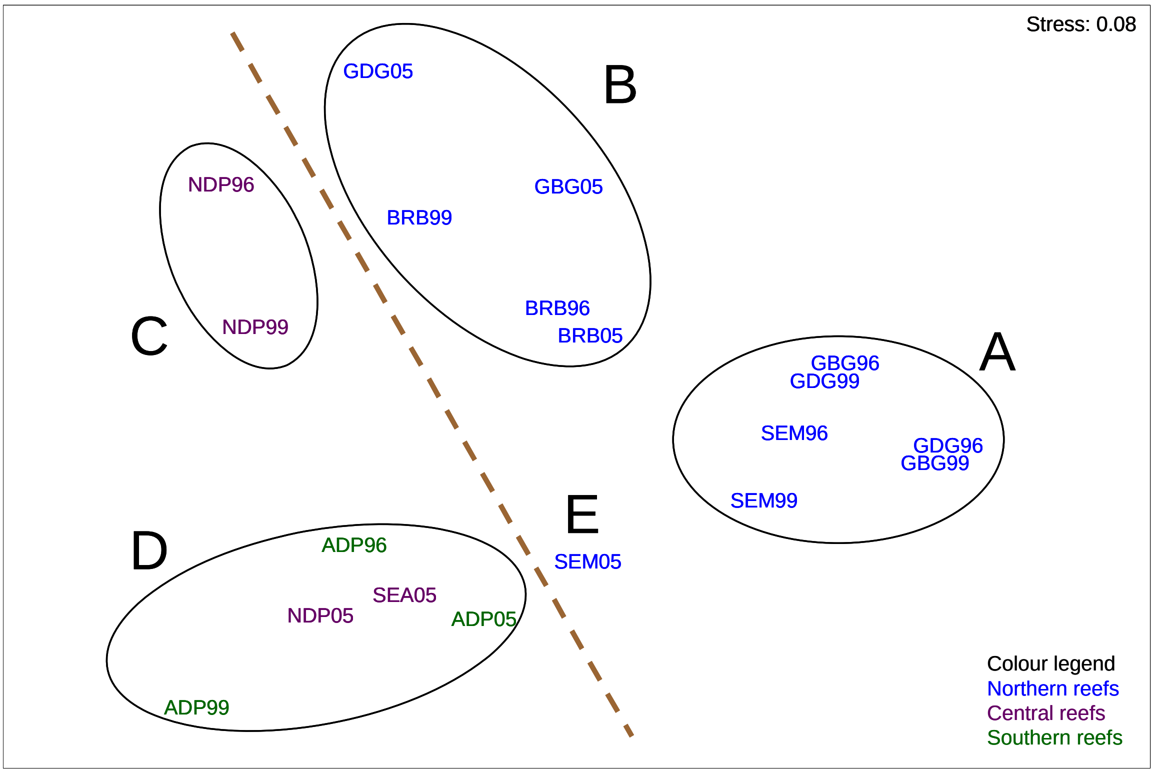

Figure 5. Non-metric MDS comparison of deep sites (8 m+) along the Adelaide Metropolitan coastline. Abbreviations used on the chart and their corresponding locations are listed in Table 2. Numbers with abbreviations indicate year of survey (e.g. SEM96 indicates Semaphore Reef, 1996 data). Dashed line shows the separation of northern reefs from central and southern reefs. Groups are derived from hierarchical agglomerative clustering of sites at 50%.

Within the classification, the northern reefs were represented by Groups A, B and E. Group A comprised three sites in 1996 and 1999, the Dredge and Barge (GDG & GBG, 18 m) and Semaphore Reef (SEM, 10 m). Group B contained the Dredge and Barge in 2005, and Broken

Bottom Reef (BRB, 10 m) for all years. Group E consisted of Semaphore Reef during the 2005 survey (Figure 5). The central and southern reefs were represented by: Group C, being the deep site at Noarlunga (NDP, 10 m) for earlier surveys (1996/99); and Group D which encompassed all years for the Aldinga Deep site (ADP, 10 m), and 2005 data for Noarlunga Deep (NDP05) and the new survey site at Seacliff Reef (SEA05, 10 m).

A common difference between the northern and other sites was the almost total absence of the larger canopy taxa (comprising the robust brown macroalgae) in the north. Northern reefs were instead dominated by smaller foliaceous and encrusting taxa (Figure 6).

The separation between Groups A and B appeared to be based on the dominance of red foliaceous macroalgae at Group A and large amounts of turfing and encrusting taxa at Group B. Semaphore 2005 (Group E) grouped separately from previous years due to the higher cover of bare substrate and higher cover of robust and foliaceous brown taxa (~10% for each) and a reduction in foliaceous reds (~25% versus 60% or more in other surveys; Figure 6).

Group C was also primarily composed of smaller life forms but differed from Group B due to the higher proportional cover of robust brown canopy taxa. Group D comprised an even larger proportion (40%) of robust browns interspersed with larger foliaceous taxa (Figure 6).

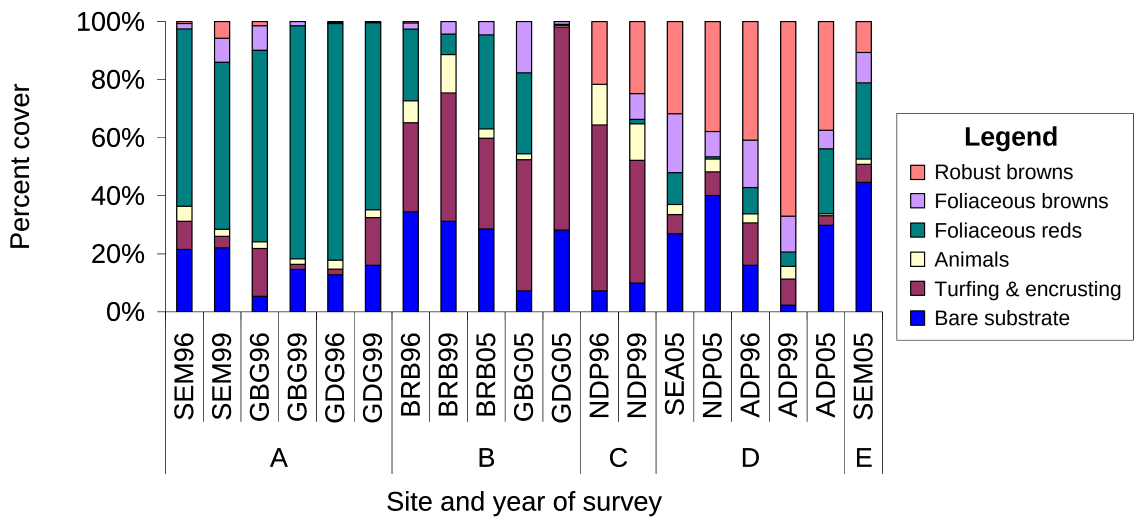

Figure 6. Comparisons of deep sites (8 m+) along the metropolitan coastline in terms of percentage cover of life forms. Sites are grouped according to the ordination in Figure 5. The order of the life forms on the chart represents their approximate location within the structure of the canopy. Abbreviations used on the chart and their corresponding locations are listed in Table 2.

Ordination of the shallow (5 m) sites showed that a similar north – south trend was present between the central and southern reefs across five groups, and that site composition was variable between surveys (Figure 7, F – J; Stress = 0.1). The majority of the southern metropolitan sites and some central sites were clustered together in Group F, while the Aldinga site (ASH) appeared in Group G separately from previous surveys. Group H contained most of the remaining central metropolitan sites with the exception of Horseshoe Reef inside (HSI) 2005, which occurred on its own (Group J), and the site on the northern outside of Noarlunga Reef (NNO) 1996, which grouped with the only northern reef site (Glenelg Blocks, GBL) 2005 to form Group I.

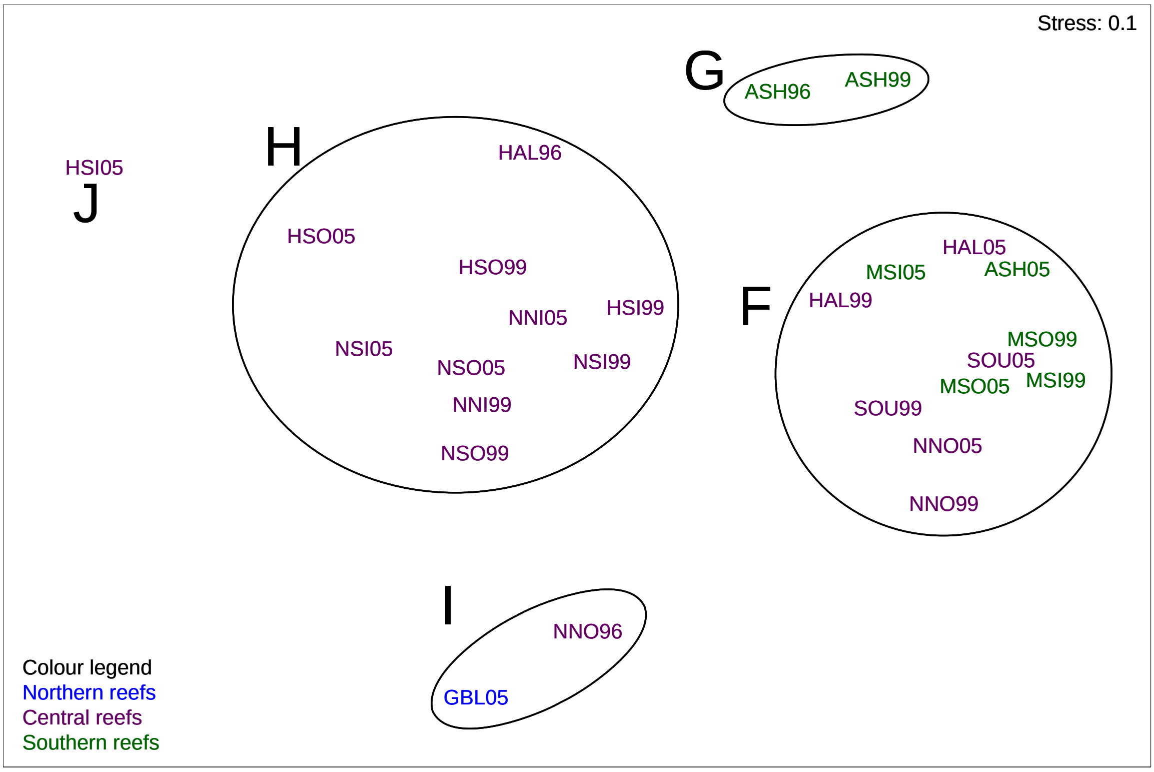

Figure 7. Non-metric MDS comparison of shallow sites (~5 m) along the Adelaide Metropolitan coastline. Abbreviations used on the ordination and their corresponding locations are listed in Table 2. Groups are derived from the hierarchical agglomerative clustering of sites at 50%.

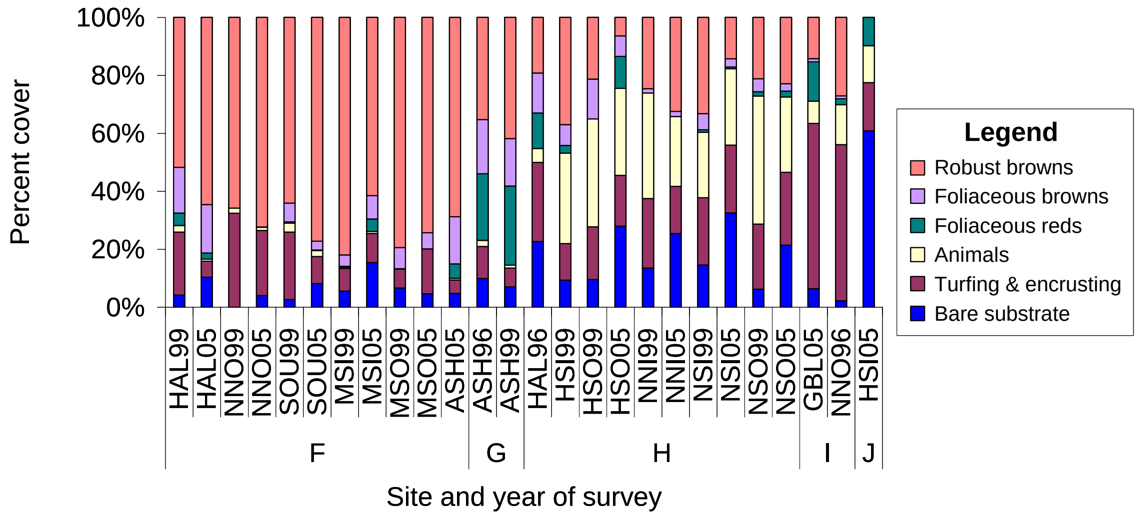

Figure 8. Comparisons of shallow sites (5 m) along the Adelaide Metropolitan coastline in terms of percentage cover of life forms. Sites are grouped according to the ordination in Figure 7. The order of the life forms on the chart represents their approximate location within the structure of the canopy. Abbreviations used on the chart and their corresponding locations are listed in Table 2.

Comparison of the major life form groups (Figure 8) showed that the southern reefs (Figure 7, Groups F and G) generally comprised high proportions of the larger brown macroalgae (40 – 80% in previous surveys and >60% in 2005). Three of the central sites (Hallett Cove, HAL; Southport, SOU; and the northern outer reef of Noarlunga, NNO) also had similarly high proportions of robust brown macroalgae in 2005. The shallow Aldinga site formed a separate group (Figure 7, Group G) due to a higher proportion of foliaceous taxa (Figure 8).

Several of the central sites (Figure 7, Group H) had lower covers of robust brown macroalgae, instead being characterised by the presence of smaller foliaceous and turfing taxa, but also a significantly high abundance (20 – 40%) of sedentary animals (primarily the mussel *Brachidontes rostratus*, Figure 8). A comparatively large cover of turfing and encrusting species differentiated Group I (Figure 7), while the inside site at Horseshoe Reef appeared separate in 2005 (Group J) because of the large proportion of bare substrate (60%, Figure 8).

Temporal changes

From 1996 to 1999 there were substantial changes in site composition in terms of changes in the areal cover of functional groups. Ordination of the difference data for 1996 to 1999 revealed four groups (based on clustering at 40%; Figure 9 K – N; Stress = 0.05), with a substantial increase in the cover of robust brown macroalgae (30 - 40%) at the three Group K sites (Hallett Cove, Noarlunga North Outside, and Aldinga Deep; Figure 10). A further three sites (Group L, comprising Semaphore Reef, Noarlunga Deep, and Aldinga Shallow) showed only a slight increase in brown macroalgal cover (<10%). For both groups, these changes were matched by a decrease in the reported areal cover of small turfing macroalgae and bare substrate (Figure 10).

None of the sites in the remaining groups (Group M – Glenelg Barge & N – Glenelg Dredge and Broken Bottom) had more than a few percent cover of larger brown taxa (Figure 10). Broken Bottom and the Glenelg Dredge (Group N) showed a small reduction in foliaceous red macroalgae (15%) in favour of turfing species (Figure 10) but the opposite trend occurred at the Glenelg Barge (Group M) with a small (15%) increase in foliaceous red macroalgae and an increase in the total amount of bare substrate (10%; Figure 10).

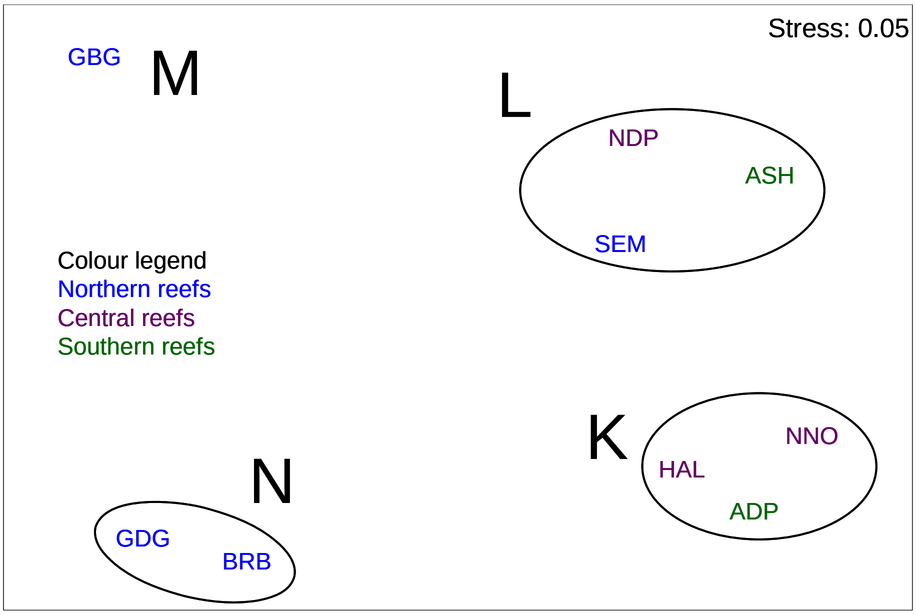

Figure 9. Changes in site composition along the Adelaide Metropolitan coastline from 1996 – 1999 through an analysis of differences in cover for each functional group between surveys. Site abbreviations used on the chart and their corresponding locations are listed in Table 2. Groups are derived from the hierarchical agglomerative clustering of sites at 40%.

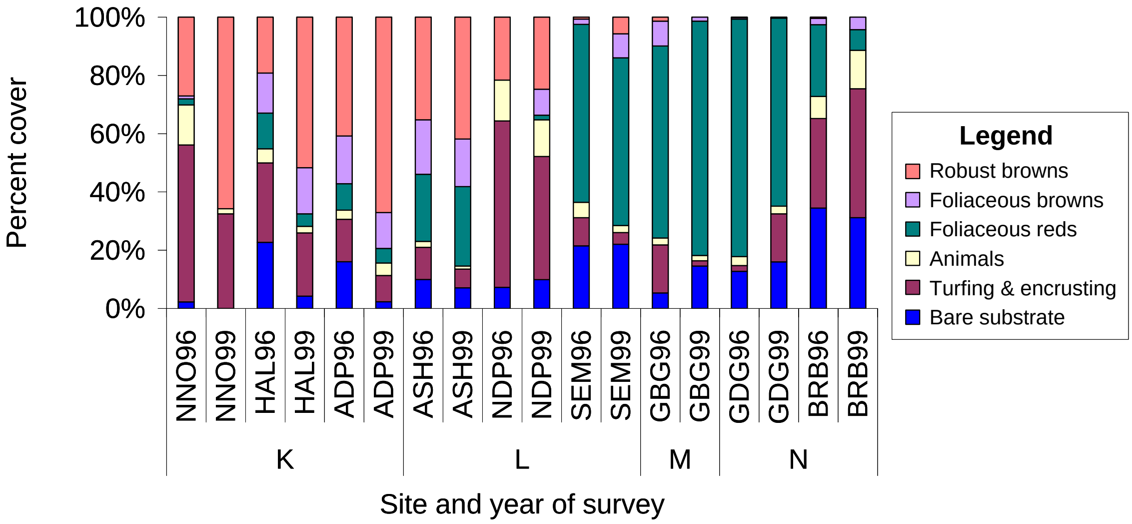

Figure 10. Comparison of sites along the Adelaide Metropolitan coastline from 1996 – 1999
Sites are grouped according to the ordination in Figure 9. The order of the life forms on the chart represents their approximate location within the structure of the canopy. Abbreviations used on the chart and their corresponding locations are listed in Table 2.

Ordination of the differences between sites from 1999 to 2005 with clustering at 40% revealed an array of six groups (Figure 11 O – T; Stress = 0.13). There was an increase (10 - 20%) in robust brown macroalgal cover at the five Group P sites (Hallett Cove, Noarlunga North Inside, Noarlunga North Outside, Noarlunga Deep and Southport; Figure 12). A slight to moderate increase in bare substrate was also observed in Group P, as for Noarlunga North Inside and Noarlunga Deep (15 and 30% respectively) and 5% for the remainder (Figure 12). A concomitant increase in bare substrate with canopy algal cover might suggest that individual plants in 2005 are relatively larger and more broadly spaced than in 1999. In itself, this could be a change in composition to bigger species or simply that the extant community has grown taller and thinner.

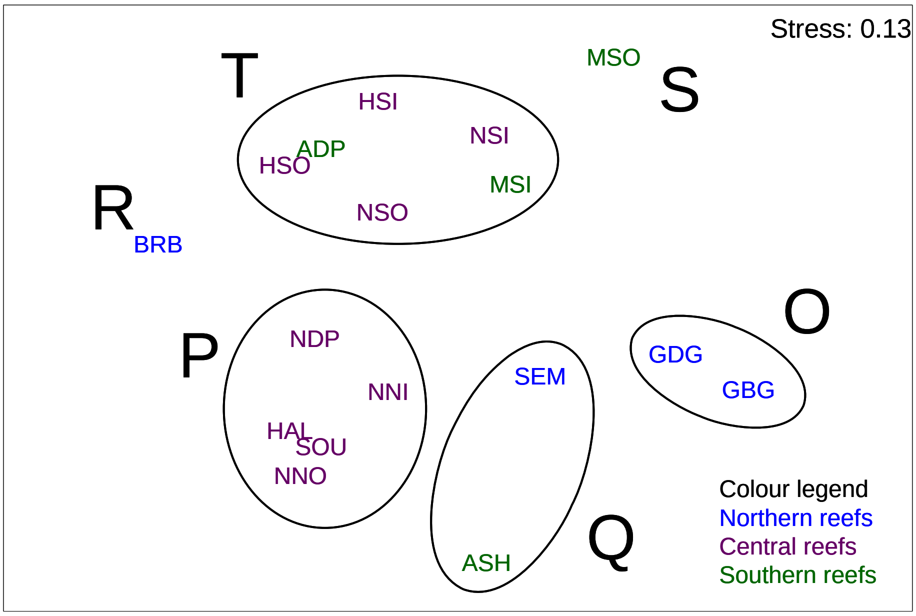

Figure 11. Changes in site composition along the Adelaide Metropolitan coastline from 1999 – 2005 through an analysis of differences in abundance for each functional group between surveys. Site abbreviations used on the chart and their corresponding locations are listed in Table 2. Groups are derived from the hierarchical agglomerative clustering of sites at 40%.

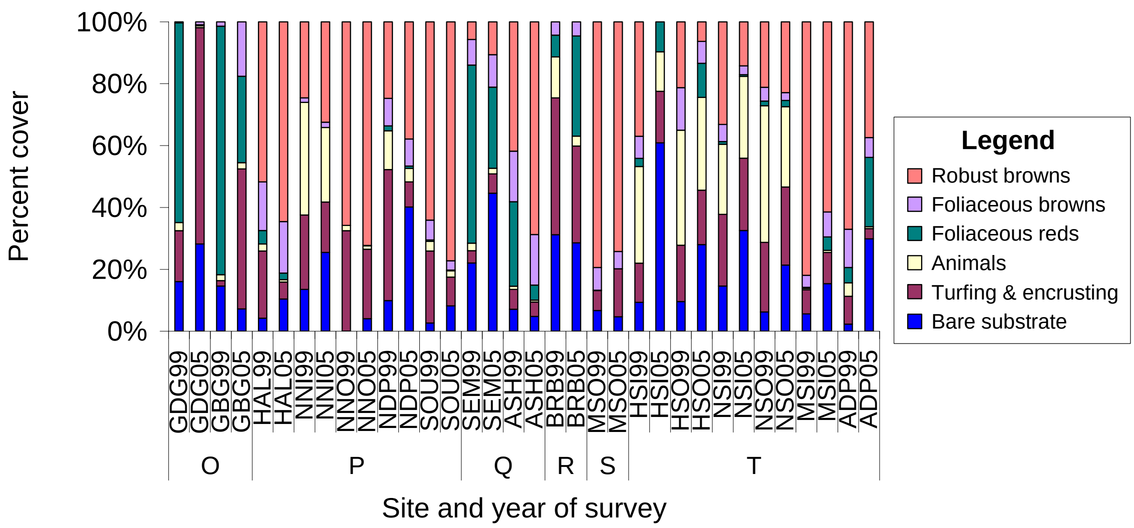

Figure 12. Comparison of sites along the Adelaide Metropolitan coastline from 1999 – 2005. Sites are grouped according to the ordination in Figure 11. The order of the life forms on the chart represents their approximate location within the structure of the canopy. Abbreviations used on the chart and their corresponding locations are listed in Table 2.

Increases in the cover of robust brown macroalgae were also observed at Aldinga Shallow (30%) and Semaphore (5%: Group Q), but were largely matched by decreases in the cover of red foliaceous algae (20 - 30%; Figure 12). The cover of bare substrate at Semaphore almost doubled to 44% of the total area. Hence, although the hierarchical agglomerative clustering algorithm grouped Semaphore with Aldinga, presumably because of the common reduction in red foliaceous macroalgae and increases in robust brown cover, the nature of their changes was fundamentally different, as Semaphore is now dominated by bare substrate and Aldinga by robust brown macroalgal cover. The latter may be reflected in the relative position of Aldinga in the ordination relative to Semaphore and the other sites (Figure 11).

Substantial increases in the areal cover of bare substrate (ranging from 10 - 50%) were observed at the six Group T sites (Horseshoe Inside, Horseshoe Outside, Aldinga Deep, Noarlunga South Inside, Noarlunga South Outside, and Moana South Inside; Figure 12). Of these, the first three also had reduced cover of robust brown macroalgae of 20 - 30%, while the others lost 0 - 10%. In contrast to the above, the small reduction in the areal cover of robust brown macroalgae at Moana South Outside (Group S) was in favour of turfing species rather than bare substrate (Figure 12).

The remaining groups (O & R) lacked a significant cover of robust brown macroalgae. At the Glenelg Barge and Dredge (Group O), 60 - 70% of the red and brown foliaceous taxa were lost to turfing macroalgae (Figure 12), while there was a 25% increase in the cover of red foliaceous macroalgae at Broken Bottom (Group R; Figure 12).

The changes observed in the interval from 1999 to 2005 are greater than those from 1996 – 1999 in that there are a larger number of groups (6 vs. 4), which suggests a broader range of change. These differences are likely due to the larger number of sites being considered and the larger interval encompassed, but also differences in season at the time of the surveys (late spring for 1999 and summer-autumn for 2005). Substantial differences between surveys could also relate to spatial differences. It is certain that at most sites the survey repetition is only in the general vicinity of prior LITs. There is thus a risk of interpreting spatial patchiness as being a temporal change. However, this is not the situation on either the Glenelg Dredge or Barge as repeated surveys are definitely across the same patch of reef.

4.1.2 Summary of indicator scores for Adelaide metropolitan reefs

Overall, index scores (the average of all non-null indices) implied the existence of a north – south gradient along the coastline, with the northern reefs appearing to be the most degraded, and those further south being better (although often only marginally so). It is of concern that only 16% (3 of 19) of metropolitan sites were classified as Good Condition, and that 31% (6 of 19) were deemed to be in Poor Condition, while the bulk of sites (10 of 19 or 53%) fell into the Caution Recommended category (Table 6). Further, three central reef sites, Horseshoe Inside and Outside, and Noarlunga South Inside fall into the Poor Condition category, which may suggest that the decline in reef health observed on northern metropolitan reefs is advancing southwards. This result was in contrast to the non-metropolitan sites where average index scores were considerably higher (see Table 7).

There was no macroalgal cover at any of the Poor Condition sites; hence, algal biodiversity was also low (ranging from 29 – 61; Table 6). Site-attached fish were low at some Poor Condition sites (Semaphore, Horseshoe Inside and Outside ranging from 2 – 4) but high at others (Glenelg Broken Bottom, Noarlunga South Inside; 76 – 100; Table 6).

Table 6. Summary of indicator and overall scores (from 0-100 in all cases except for the presence/null indices) for Adelaide metropolitan sites. Note that blank = null.

Site	Areal cover of canopy macroalgae	Areal cover of turfing macroalgae	Areal cover of mussels	Areal cover of bare substrate	Abundance of site attached fish	Abundance of mobile predators	Median abundance of wrasse	Presence of invasive taxa	Evidence of high sedimentation	Richness of mobile invertebrates	Richness of macroalgae	Score	Status
Semaphore Reef	0			0	2	64	14	0	0	36	61	**20**	**Poor**
Broken Bottom	0				76	35	14		0	47	46	**31**	**Poor**
Glenelg Barge	0				100	28	0			26	22	**29**	**Poor**
Glenelg Dredge	0	0			100	14	98			15	20	**35**	**Caution**
Glenelg Blocks	0	0			42	85	100	0		57	22	**38**	**Caution**
Seacliff Reef	25				100	28	39			36	62	**48**	**Caution**
Hallett Cove	100				47	78	73			57	100	**76**	**Good**
Horseshoe Inside	0			0	4	57	0		0	31	29	**15**	**Poor**
Horseshoe Outside	0	0			4	100	0		0	42	49	**24**	**Poor**
Noarlunga North Inside	40		44		100	71	0			31	30	**45**	**Caution**
Noarlunga North Outside	100				100	85	0			68	100	**76**	**Good**
Noarlunga South Inside	0		0	0	100	100	0			52	23	**34**	**Poor**
Noarlunga South Outside	4				49	64	44			36	32	**38**	**Caution**
Noarlunga Deep	46			0	100	71	29			52	22	**46**	**Caution**
Southport	100				39	42	34		0	42	100	**51**	**Caution**
Moana Inside	99				58	19	0			42	100	**53**	**Caution**
Moana Outside	100				100	66	26		0	73	100	**66**	**Good**
Aldinga Shallow	100				51	28	54			36	100	**62**	**Caution**
Aldinga Deep	16			19	69	28	29		0	63	52	**35**	**Caution**

Within the Caution Recommended category, there is little or no pattern to particular indices. Five sites (Glenelg Dredge and Barge, Aldinga Deep, Noarlunga South Outside and Seacliff) had low macroalgal cover (from 0 – 25), but the others (Noarlunga North Inside, Noarlunga Deep, Moana Inside, Southport and Aldinga Shallow) had substantially higher macroalgal cover (from 40 – 100; Table 6). Similarly, the blue-throated wrasse index was zero at two Caution Recommended sites, Noarlunga North Inside and Moana Inside, but intermediate (29 – 54) at six others and high (98 & 100) at the remaining two.

Sites within the Good Condition category (Moana Outside, Hallett Cove and Noarlunga North Outside) were very high in canopy macroalgal cover (100), which carries over to algal species richness as part of the data manipulation (Table 6). The score for site-attached fish was high (100) at two of the three sites, but the score for blue-throated wrasse was varied (0-73) across the good sites. Interestingly, the northern limit for blue-throated wrasse in Gulf St Vincent was previously thought to be Christies Beach (see the above index description), but observations of this species were made as far north as Semaphore. Otherwise, as for the Caution Recommended category, there is little pattern to the remaining indices (Table 6).

4.2 Fleurieu Peninsula regional surveys

All sites within the region were characterised by a high percentage cover of robust brown macroalgae (60 – 80%), and low areas of bare substrate (< 10%, Figure 13). The ordination of sites within the Fleurieu Peninsula region distinguished three groups (Figure 14, Groups U – W; Stress = 0.0). Granite Island and Pt Elliot (Group U) differed from the others due to the presence of approximately 15% cover of red foliaceous macroalgae, substantially higher than other sites (Figure 13). The remaining two groups differentiated according to the ratio of robust to foliaceous brown macroalgae, with Group V (Second Valley, Cape Jervis North, and Cape Jervis South) having the highest cover of canopy macroalgae (>80%), while Group W (West Island, The Bluff and Carrickalinga) contained slightly more brown foliaceous species (Figure 13).

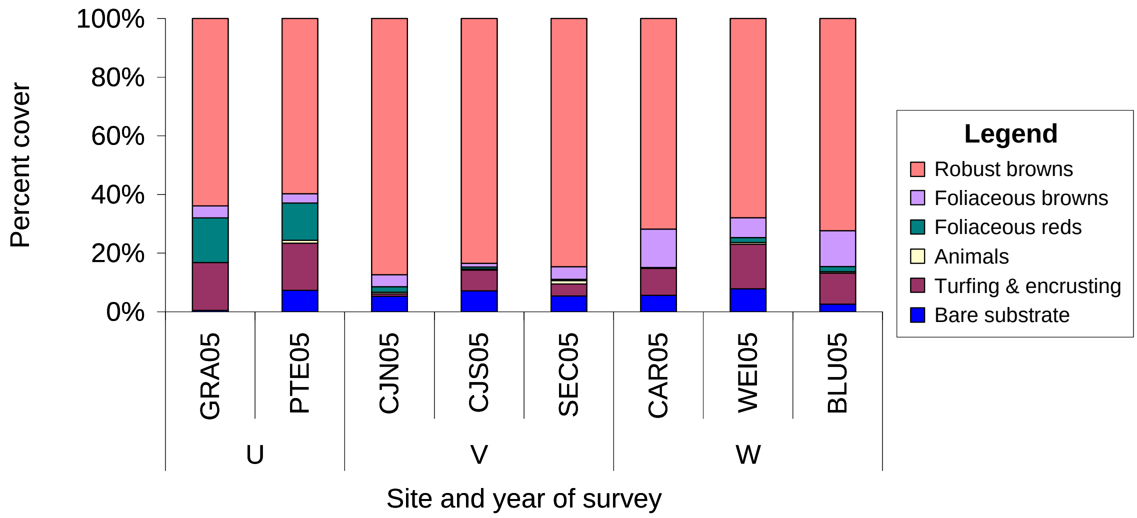

Figure 13. Comparison of sites within the Fleurieu Peninsula region.
Site abbreviations used on the ordination and their corresponding locations are listed in Table 3.

4.2.1 Site Comparisons

There is some pattern to the ordination groupings relative to geographic position. Group V sites (Second Valley, Cape Jervis North and South) are consecutive along the coast (Figure 2, Figure 14) and the two most eastern sites (Granite Island and Port Elliot) comprise Group U. However, Group W (Carrickalinga, West Island and The Bluff) is spread across either side of Fleurieu Peninsula (Figure 2, Figure 14). The separation of Group U from other sites may relate to the more urbanised coast in and around Encounter Bay. Certainly there is a relatively marked change in overall cover composition (shift from Group W sites into Group U) across a relatively short stretch of coast (Figure 2, Figure 14). However, there is no indication in the ordination and related cover data of a north – south gradient along the coast, although there may be one from east to west (Group U to Group W and then Group V).

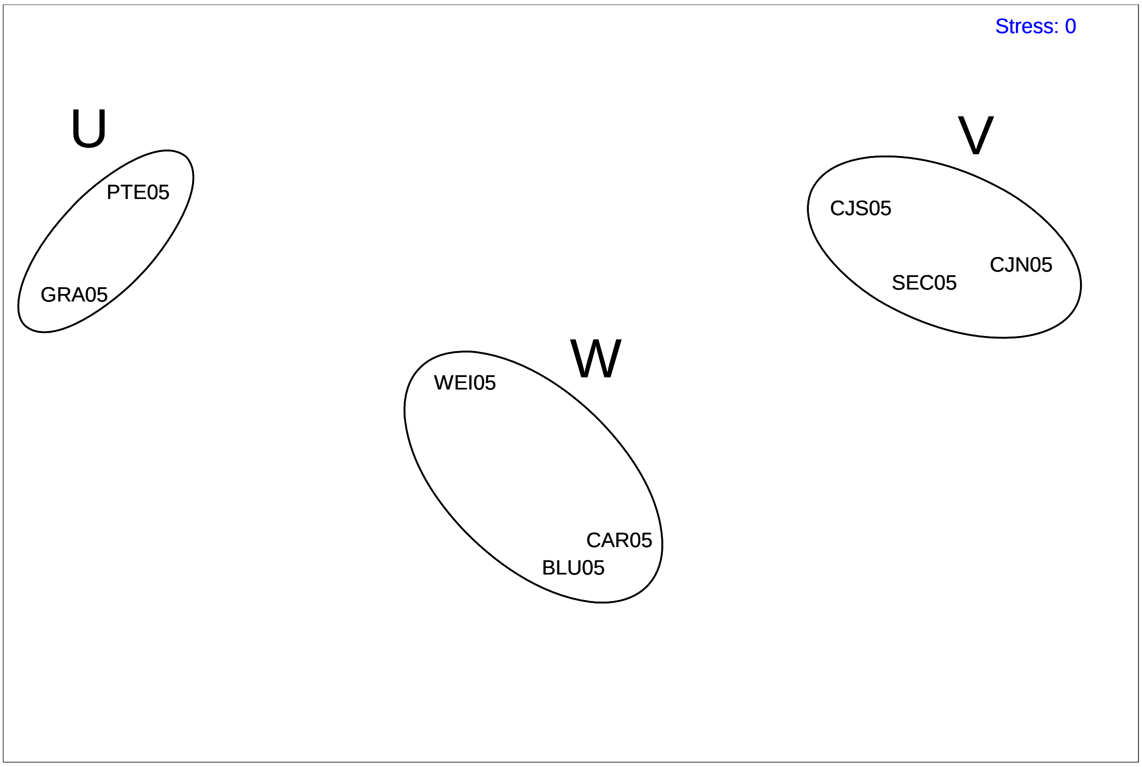

Figure 14. Non-metric MDS comparison of sites within the Fleurieu Peninsula region. Site abbreviations used on the ordination and their corresponding locations are listed in Table 3. Groups are derived from the hierarchical agglomerative clustering of sites at 85%.

4.2.2 Summary of indicator scores for the Fleurieu Peninsula regional sites

Based on the overall average, six sites (Carrickalinga, Second Valley, Cape Jervis North, Cape Jervis South, West Island and The Bluff) were in Good Condition with two (Granite Island and Port Elliot) falling into the Caution Recommended category (Table 7). The latter was probably due to low levels of site-attached fish at both sites (9 & 5 respectively). A low site-attached fish index for The Bluff is offset by this site maintaining the highest level of mobile invertebrate predators (71 compared to 7 – 50 across other sites; Table 7). The blue-throated wrasse index was lowest at Second Valley (39), but this was offset by a high score for site-attached fish (100). Granite Island had a wrasse index of 63, while all other sites were 100. There was no apparent pattern in the mobile invertebrate richness across sites (Table 7). None of the indices targeting factors for of concern (areas of mussels, turf and bare substrate as well as high sediment load and marine pest presence) registered a numeric score in the Fleurieu Peninsula area.

Table 7. Summary of indicator and overall scores (from 0-100 in all cases except for the presence/null indices) for Fleurieu Peninsula sites. Note that blank = null.

Site	Areal cover of canopy macroalgae	Areal cover of turfing macroalgae	Areal cover of mussels	Areal cover of bare substrate	Abundance of site attached fish	Abundance of mobile predators	Median abundance of wrasse	Presence of invasive taxa	Evidence of high sedimentation	Richness of mobile invertebrates	Richness of macroalgae	Score	Status
Carrickalinga	100				100	14	100			21	100	**73**	**Good**
Second Valley	100				100	50	39			42	100	**72**	**Good**
Cape Jervis North	100				47	21	100			57	100	**71**	**Good**
Cape Jervis South	100				29	50	100			57	100	**73**	**Good**
West Island	100				17	42	100			63	100	**70**	**Good**
The Bluff	100				8	71	100			52	100	**72**	**Good**
Granite Island	100				9	7	63			36	100	**53**	**Caution**
Port Elliot	100				5	28	100			42	100	**63**	**Caution**

4.3 Yorke Peninsula regional surveys

4.3.1 Site comparisons

Ordination and hierarchical agglomerative cluster analysis revealed five groups (Figure 15, Groups X – B1; Stress = 0.03). Group X (Troubridge Point, Point Yorke and Wardang Island) had high (80% or higher) robust brown macroalgal cover with only marginal levels of other groups (Figure 16). The largest group (Group Y) comprising Cable Hut Bay, Goose Island, Edithburgh Pool and Corny Point, had marginally less canopy macroalgae (around 70% or higher), which would explain this group's similarity to Group X (Figure 15, Figure 16).

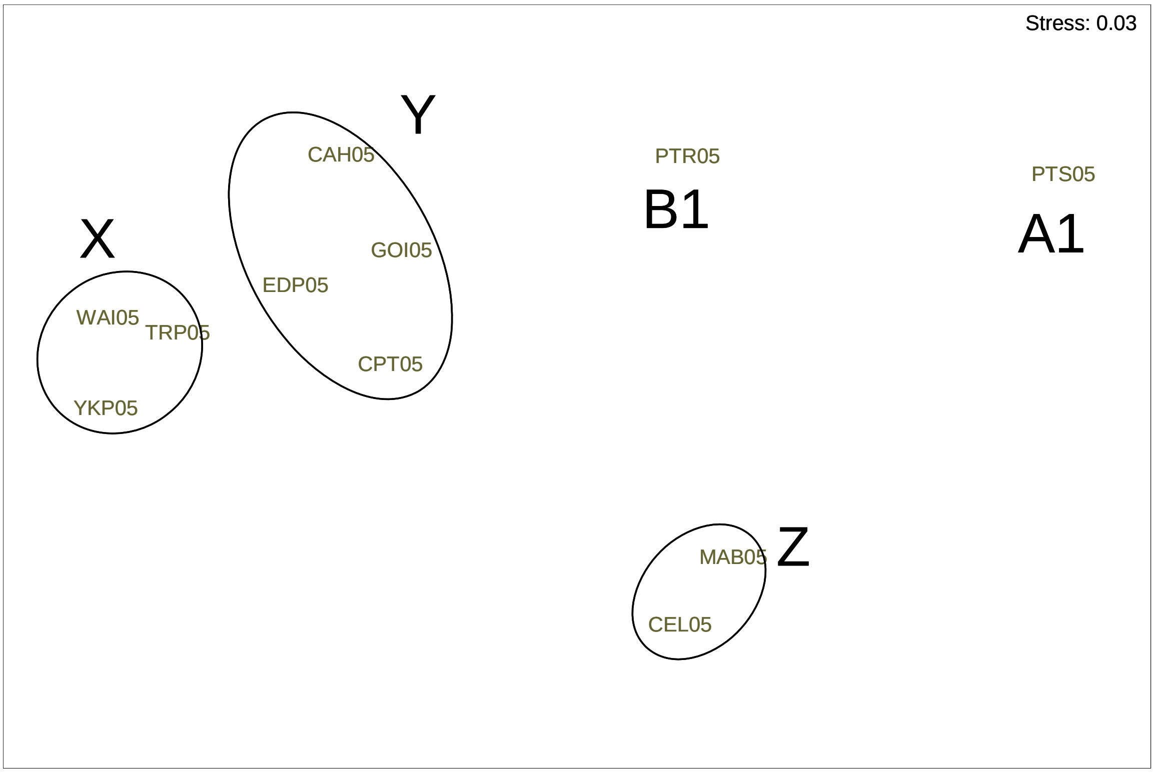

Figure 15. Non-metric MDS comparison of sites within the Yorke Peninsula region. Site abbreviations used on the ordination and their corresponding locations are listed in Table 4. Groups are derived from the hierarchical agglomerative clustering of sites at 80%.

Group X sites contain a high proportion of robust browns (Figures 17 and 18), generally greater than 80% cover. Bare substrate was higher in Group Y sites relative to Group X (from 10 – 20% compared with less than 5%; Figure 16). Group Z comprised two sites (Cape Elizabeth – and Marion Bay) that were somewhat isolated from other groups (Figure 15), probably because of their high cover of foliaceous brown macroalgae (> 20% at both sites) at the expense of robust canopy species (Figure 16). The remaining groups (A1 and B1) contain only one site each (Point Souttar and Point Riley), both of which had relatively low cover (20% and 50% respectively) of robust browns and substantial areas of bare substrate (around 30%; Figure 16). However, Point Souttar also had the highest cover of foliaceous reds (around 20%; Figure 16), which may indicate a slightly more sheltered location in this instance.

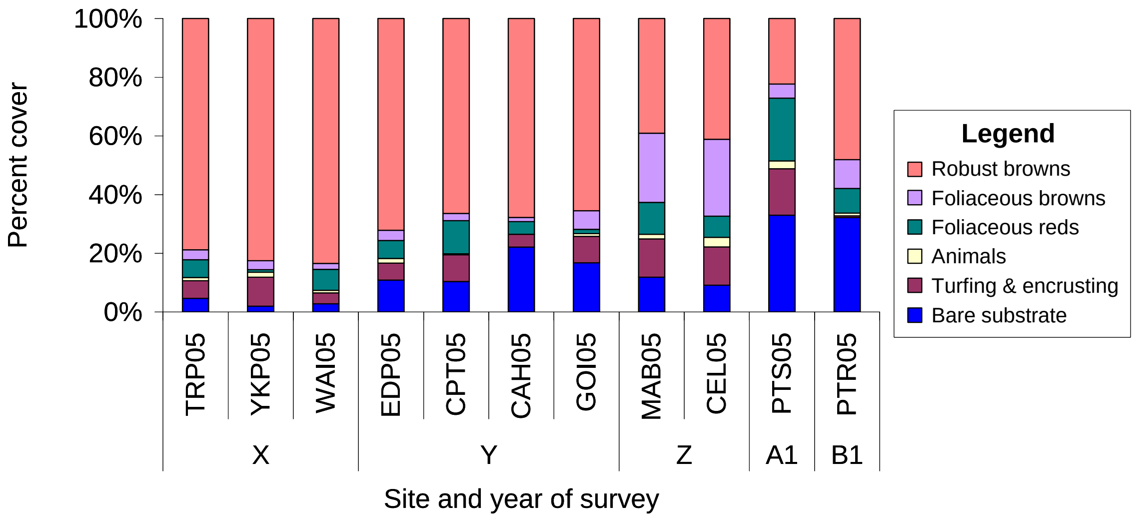

Figure 16. Comparison of coastal reefs within the Yorke Peninsula region. Site abbreviations used on the ordination and their corresponding locations are listed in Table 4.

There was no relationship between ordination results and geographic position around Yorke Peninsula (Figure 3, Figure 15), and hence no apparent north to south gradient of sites. This is similar to Fleurieu Peninsula, and in contrast to Adelaide's metropolitan coast. The observed groupings of Yorke Peninsula sites possibly reflect local differences in underwater topography, geology and hydrology.

4.3.2 Summary of indicator scores for Yorke Peninsula regional sites

Two sites from Yorke Peninsula, Point Souttar and Point Riley, were classified as being in poor health. Both locations registered on the bare substrate index (41 & 31 respectively) and received zero on the sediment index (Table 8). Macroalgal cover rated low at both sites (37 & 65), although Marion Bay rated lower (31). The latter probably remained in the Good Condition category on the strength of site-attached fish (73), abundance of mobile predators (71) and the blue-throated wrasse index (100, Table 8).

Site-attached fish scores at Point Souttar and Point Riley were relatively low (10 & 3 respectively), but again, other sites were similarly low (Corny Point = 3, Cable Hut Bay = 4 and Troubridge Point = 8). Corny Point had an overall average index in the Good Condition range, with index values of 100 for mobile predators and blue wrasse (Table 8). Both Cable Hut Bay and Troubridge Point fell into the Caution Recommended category. Apart from site-attached fish, Cable Hut Bay and Troubridge Point were also weak in mobile predator abundance (14 & 21 respectively). The richness of mobile invertebrates was also low for Cable Hut Bay (26), while Troubridge Point received zero for sedimentation (Table 8). Point Riley was also weak for richness of mobile invertebrates (31) and macroalgal richness (32, Table 8).

All other sites had overall Good Condition reef scores, which were generally based on high macroalgal cover and richness (although not at Marion Bay – see above). The blue-throated wrasse index was 100 for all sites except those that ranked as Poor Condition (Point Souttar and Point Riley) and Cape Elizabeth (45). However, Cape Elizabeth was strong in mobile predator abundance (95) and richness of mobile invertebrates (84, Table 8). Cape Elizabeth is considered to be the northern limit for blue-throated wrasse in Spencer Gulf (see the index descriptions), and so this species might not be expected at Point Riley. However, the same limit for blue-throated wrasse in Gulf St Vincent (Christies Beach) was found to be incorrect, with observations as far north as Semaphore (Table 6).

Table 8. Summary of indicator and overall scores (from 0-100 in all cases except for the presence/null indices) for Yorke Peninsula sites. Note that blank = null.

Site	Areal cover of canopy macroalgae	Areal cover of turfing macroalgae	Areal cover of mussels	Areal cover of bare substrate	Abundance of site attached fish	Abundance of mobile predators	Median abundance of wrasse	Presence of invasive taxa	Evidence of high sedimentation	Richness of mobile invertebrates	Richness of macroalgae	Score	Status
Edithburgh Pool	100				100	57	100		0	57	100	**73**	**Good**
Troubridge Point	100				8	21	100		0	63	100	**56**	**Caution**
Point Yorke	100				31	85	100			100	100	**86**	**Good**
Marion Bay	31				73	71	100			84	73	**72**	**Good**
Cable Hut Bay	100				4	14	100			26	100	**57**	**Caution**
Corny Point	90				3	100	100			57	84.5	**72**	**Good**
Point Souttar	37			41	10	42	0		0	42	40.5	**27**	**Poor**
Wardang Island	100				64	42	100			89	100	**83**	**Good**
Goose Island	100				74	50	100			68	100	**82**	**Good**
Cape Elizabeth	78				50	95	45			84	64.5	**69**	**Good**
Point Riley	65			31	3	21	0		0	31	32	**23**	**Poor**

As with the cover data, the sites that scored as Poor Condition , Point Riley and Point Souttar, are somewhat different to other locations on Yorke Peninsula. Both sites had substantial areas of bare substrate and were high in sediments. These factors are interrelated and to determine a causal mechanism would be problematic, particularly as these data present a snapshot of the system at one point in time. Within the Caution Recommended sites (Troubridge Point and Cable Hut Bay), the cover data do not suggest that these sites were different to others in the same groups (X and Y respectively), all of which had overall indices in the Good Condition range. Speculation as to the potential causes for this result would also be problematic. Sites in both the Poor Condition and Caution Recommended categories may be naturally different to other sites, and the rankings are very much part of a preliminary look at these reefs.

4.4 Inter-regional comparison

An ordination of LIT data from all sites in the 2005 survey was undertaken to assist in placing the status of reefs on the metropolitan coast in an appropriate context, as well as in an attempt to encompass the level of variation that might occur across healthy reefs. The resultant plot, combined with a hierarchical agglomeration at ~60% indicated four groups (Figure 17, Groups C1 – F1; Stress = 0.11).

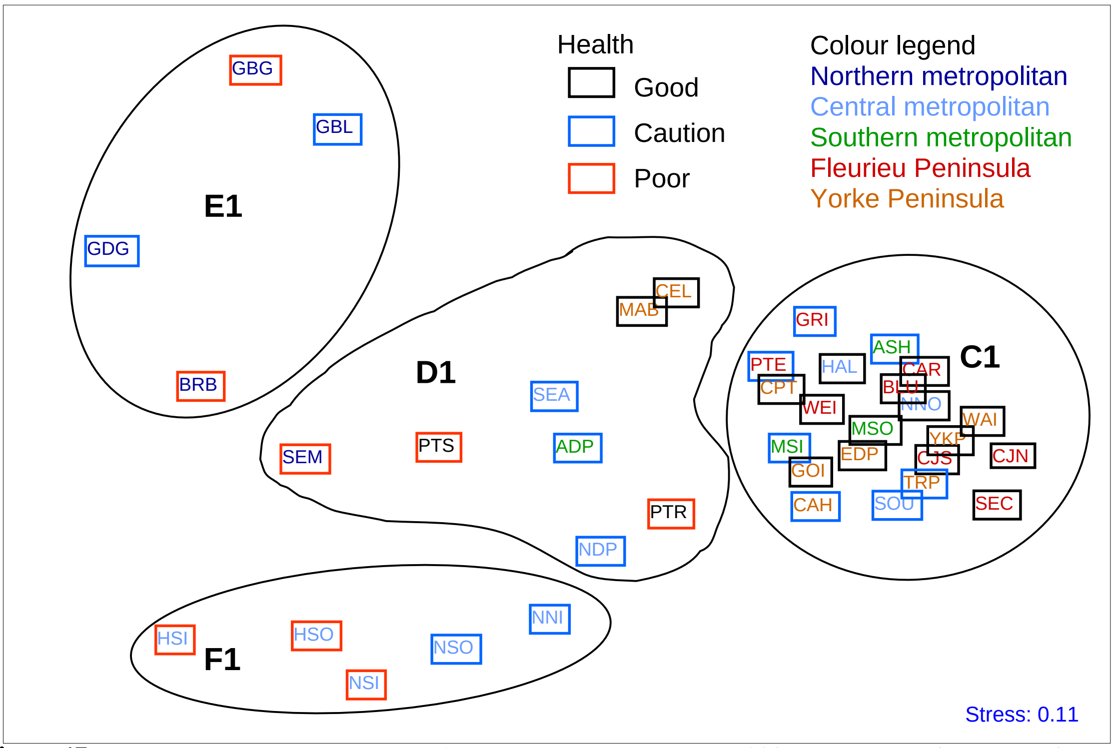

Figure 17. Non-metric MDS comparison of all sites in the 2005 survey. Abbreviations used on the ordination and their corresponding locations are listed in Table 2, Table 3, and Table 4. Groups are derived from the hierarchical agglomerative clustering of sites at ~60%.

All Good Condition sites from the Adelaide metropolitan coast, Fleurieu Peninsula and Yorke Peninsula except Marion Bay and Cape Elizabeth occurred in Group C1 (Figure 17). This group also contained some caution category sites, including Troubridge Point, Cable Hut Bay, Southport, Moana Inside, Aldinga Shallow, Granite Island and Port Elliot (Figure 17, Table 6, Table 7, Table 8). All Fleurieu Peninsula sites are thus incorporated in Group C1 regardless of their health index score. In compositional terms, all Group C1 sites are overwhelmingly dominated by robust brown macroalgae (at least 60%) with low if any animal cover (Figure 8, Figure 13, Figure 16).

The only Good Condition category sites not in Group C1, Marion Bay and Cape Elizabeth (Figure 17, Group D1) are separated on the basis of relatively low levels of robust browns (~ 40%), and high levels of foliaceous brown macroalgae (Figure 16). These sites were also an isolated group in the ordination of Yorke Peninsula sites (Figure 15, Group Z) indicating that they were not typical of this region. However, when combined, the robust and foliaceous algal cover help to give these sites their Good Condition rating.

Apart from the Good Condition sites of Marion Bay and Cape Elizabeth, Group D1 contains three Caution Recommended sites (Seacliff, Noarlunga Deep and Aldinga Deep) and three reefs ranked Poor Condition (Semaphore, Point Souttar and Point Riley; Figure 17). These sites all had ≤40% cover of robust browns (Semaphore had <10%), but were highly varied in other components, as reflected in the spread of this group on the ordination (Figure 17). Marion Bay and Cape Elizabeth had relatively higher foliaceous brown covers (see above), while other sites had at least 20% of bare substrate (as high as ~50% at Semaphore) and Point Souttar and Aldinga Deep had ~20% cover of foliaceous reds (Figure 6, Figure 13, Figure 16). Interestingly, all of the metropolitan sites in this group (Semaphore, Seacliff, Noarlunga Deep and Aldinga Deep) are deep (8 - 12 m) while all other sites are shallow (5 m or less), which may suggest water quality issues may be prevalent at the latter (Table 2, Table 3, Table 4).

Group E1 includes probably the most degraded reefs in the survey, comprising all North Metropolitan reefs except Semaphore (Glenelg Dredge, Glenelg Barge, Glenelg Blocks and Broken Bottom; Figure 17). All sites in this group were classified as in Poor Condition (Broken Bottom and Glenelg Barge) or Caution Recommended (Glenelg Dredge and Blocks; Table 6), with little to no robust brown algal cover (highest was <20% at Glenelg Blocks) and high cover of foliaceous reds and/or turfing macroalgae (Figure 6, Figure 8).

Finally, Group F1 also exclusively comprised reefs from the metropolitan coast, albeit from the central zone, including Horseshoe Inside, Horseshoe Outside, Noarlunga South Inside, Noarlunga South Outside and Noarlunga North Inside (Figure 17). Like Group E1, these reefs are classified as Poor Condition (Horseshoe sites and Noarlunga South Inside) or Caution Recommended (Noarlunga South Outside and North Inside). However, these sites were rather different to Group E1 in terms of composition; although they had similarly low or no cover of robust browns (<40% at Noarlunga North Inside down to 0% at Horseshoe Inside), these sites all had a relatively high animal cover (~10 – 30%; Figure 8).

While the ordination appears to support the concept of the reef health index, with most Poor Condition and Caution Recommendation ranked reefs forming a diverse array of points relative to a more discrete group of healthy reefs (Figure 17), this interpretation may be too simplistic. The inclusion of seven Caution Recommendation reefs in the Good Condition group and the exclusion of two Good Condition reefs might suggest that the index is too rigid in its application.

Further, nearly all sites in the Poor Condition and Caution Recommended categories were from the metropolitan coast, with Point Souttar and Point Riley the only two non-metropolitan sites to occur outside the Good Condition group (Figure 17). The influence of highly degraded reefs on the metropolitan coastline may mask more subtle regional health issues. Further, this strongly implies that the gradients observed on the metropolitan coast are anthropogenic.

5 Discussion

Many of the processes that characterise reef assemblages are still poorly understood (Underwood and Kennelly 1990, Keough and Butler 1995), and it is difficult to state with confidence which patterns represent 'healthy' and which represent 'unhealthy' environments. Temperate macroalgal-dominated reefs in southern Australia are diverse (Cheshire *et al.* 2000) and structurally complex (Turner *et al.* 2006a), incorporating up to three or more levels of canopy and compositional variation across all spatial scales (Turner 1995, Collings 1996). Relative to coral reefs, the methodologies for assessment of temperate reef systems are new and evolving.

The major challenge in characterising the health of a reef is the need to define a set of indices for which reliable, replicable and consistent measurements can be obtained, leading to robust interpretations and conclusions (Turner *et al.* 2006a).

5.1 Status of South Australian reefs

5.1.1 Status of metropolitan reefs

On the Adelaide coast, most sites had a similar biotic composition between 1996 and 1999, but many shifted in structure between 1999 and 2005. While this may represent a dynamic shift in composition, previous surveys were undertaken in late spring (November), whereas the 2005 survey occurred during late summer and autumn (February – May). Hence, some of the observed differences may represent seasonal fluctuations in community structure. Many algal species have seasonal changes in biomass relative to their reproductive state (Edgar 1983). Future Reef Health surveys should be carried out at a particular time of year (late summer/autumn tends to provide warmer water and calmer diving conditions). If not, a greater understanding of the seasonal changes that occur with reef communities would be needed to incorporate them into the model. Both approaches have limitations. As the number of reef surveyed increases, the time required to complete the survey becomes a factor. However, collecting seasonal data from a large number of reefs would entail substantial costs. The solution to both issues may be through greater investment in community monitoring through programs such as Reef Watch.

In general terms, the macroalgal composition of the metropolitan reefs has remained relatively unchanged. There is still a distinct south to north trend, with the southern reefs dominated by the large brown (Phaeophycean) macroalgae and the northern reefs composed of the smaller foliaceous and turfing red (Rhodophycean) algae. Northern reefs appeared more variable than in the previous surveys. The benthic community observed on the central reefs remained intermediate between those of the northern and southern reefs, but there was a further loss of robust brown macroalgae from Noarlunga and Horseshoe Reefs. The 'healthiest' reefs on the metropolitan coast were in the south, with macroalgal community structures similar to those found on the Fleurieu and Yorke Peninsulas.

The fact that little change has been observed on the northern metropolitan reefs indicates that:

- Factors inhibiting recovery of the robust brown taxa are still in effect; or
- Recovery times are significantly longer than the period between Reef Health surveys; or
- The current composition of these reefs represents a stable state.

Consideration of the reef index data for reefs on the metropolitan coast is less clear. Many of the southern reefs that appeared healthy when considering macroalgal composition still received a Caution Recommended status, because of low fish numbers, sedimentation levels and/or bare substrate. Water quality factors, specifically sedimentation and turbidity, may have a profound influence on reef community composition as high sedimentation implies high turbidity (see below), which may lead to smothering or a loss of light restricting macroalgal cover. Increased bare substrate was observed at two of the deeper southern sites (Noarlunga and Aldinga).

Horseshoe Inside has declined substantially in health since 1999, with a change from a mussel / robust brown community to a reef comprised of 60% bare substrate; it was subsequently rated as the poorest site in the 2005 survey. This may be due (at least in part) to past disturbance events that have disrupted biotic processes. In particular, a large sedimentary impact resulting from dredging (1997) was implicated in the initial decline in canopy-forming macroalgae (Turner and Cheshire 2002). Other studies on Gulf St Vincent reef systems have also confirmed that a number of areas are showing signs of stress, and there is concern for the long-term survival of these environments (Steffensen *et al.* 1989, Cheshire *et al.* 1998a, Turner and Cheshire 2002, Nicolson *et al.* 2003, Gorgula and Connell 2004, Turner 2004). However, the causal mechanisms for reef degradation are difficult to pinpoint, as reef health is likely to be the product of a range of both direct and indirect influences similar to those posing threats to local seagrass systems (Westphalen *et al.* 2005).

5.1.2 Status of reefs on Yorke Peninsula and Fleurieu Peninsula

Reefs on the Fleurieu and Yorke Peninsulas appear to be fundamentally healthy, particularly when cover of brown macroalgae is considered. However, the reef index results highlighted sites worthy of further investigation.

Two sites, Granite Island and Port Elliot, separated from other Fleurieu sites in the ordination (Figure 14), and received Caution Recommended ratings. Reef aspect (exposure to wave energy) may be a contributing factor. Expanding coastal development and increased recreational activity on this stretch of the coast might also be a factor. Nutrient and sediment loads from the Inman and Hindmarsh Rivers in Victor Harbour might influence the reef ecosystem in the region of

Granite Island. There is also a lot of sediment in the general area laid down by historical flows of the Murray River, and these two sites are the closest to the Murray Mouth.

Several sites around Yorke Peninsula rated Poor Condition or Caution Recommended, in particular Point Souttar, Point Riley, Troubridge Point, and Cable Hut Bay. Yorke Peninsula is generally less urbanised than the Fleurieu, hence the sites' lower status may be a reflection of natural factors related to topography, wave energy and/or current flow. The two sites ranked as Poor Condition, Point Souttar and Point Riley, are shallow sheltered locations with low relief. These sites are likely to be more sensitive to sedimentation, either natural or anthropogenic. These reefs, observed during the 2005 survey, may not in fact be in poor condition at all, but may simply have naturally lower algal diversity. Point Souttar in particular had high invertebrate diversity, and is located in an area of low wave exposure and mild currents. More information is needed before interpreting these reefs as degraded.

As it is the first time that the Yorke Peninsula sites have been surveyed, the data provide a valuable snapshot of the system as it was in 2005. The real strength of the Reef Health data is in detecting change over time, and these data sets will serve as a baseline for comparisons with future surveys.

The concept of ecosystem health is itself subjective (Turner *et al.* 2006a). The types of judgements made depend on a mixture of scientific, social and political objectives (Fairweather 1993). Judgements are often made about the state of a reef against expectations of what is deemed to be a healthy ecosystem (Fairweather 1999). Point Souttar provides a good example of this. To an invertebrate specialist, it was a fascinating site. To a macroalgal specialist, it was in poor health and appeared degraded.

Expansion of the reef health program to non-metropolitan South Australia in 2005 was an important next step in the development of reef health assessment and management. Consideration of reefs on the Fleurieu and Yorke Peninsulas has provided a greater understanding of the dynamics of reef systems over a broader swathe of South Australia's coasts. The advantages include:

- the collection of baseline data for southern Fleurieu and Yorke Peninsula reefs;
- greater clarity on the status of metropolitan reefs, by allowing a comparison of these reefs to others throughout the state;
- the development and interpretation of indices which give insight into the status of reefs;
- insights into how different types of environmental impacts (eg nutrients vs. sedimentation vs. coastal development) might influence reef health;
- a better appreciation of the suite of biological states that comprises 'healthy' reefs
- the establishment of a comprehensive database to inform coastal managers on the status of reefs throughout the state.

Future surveys should include marine protected areas, as well as reefs in high-risk areas adjacent to ports and harbours or patches of coast subject to substantial terrigenous inputs. At the larger scale, the potential influences of droughts (a factor of particular relevance at this point in time) and future global warming as agents of change to reef status (see below) should be considered.

5.2 Factors affecting the health of temperate reef systems

A range of anthropogenic impacts are known to occur on temperate reef environments (Turner et al. 2006a), particularly those near to urban areas. These can include: changes to water quality (water clarity, sedimentation levels, nutrient enrichment, changes in salinity, and the addition of toxicants); as well as other factors such as climate change, the establishment of opportunistic and exotic taxa, increases in extractive resource use (e.g. fishing, mining), and physical disturbance (e.g anchor and fixed mooring scars). It would be difficult to find a reef on the metropolitan coast that is not subject to some or all of these factors to some degree. Further, the affects of these impacts may be synergistic.

The difficulty in determining cause-and-effect is that damage to reefs is based largely on correlation, a problem also observed in studies of seagrass loss (Short and Wyllie-Echeverria 1996, Seddon 2000). Those factors which are likely to cause most concern are those that threaten the major structural habitat-forming species such as Laminarian and Fucalean macroalgae (i.e. robust canopy-forming species) and sponges (Cheshire *et al.* 1998b). The use of a range of environmental indices can assist in determining impacts reef health from different impact perspectives, but the indices need to be refined, and the affect of different environmental impacts on reef health determined.

The following sections (5.2.1 -5.2.3) have been included to provide background about the factors that can affect temperate reefs. These are discussed in the context of factors that have been implicated in the large-scale loss of seagrass on Adelaide's metropolitan coastline (Fotheringham 2002).

5.2.1 Toxicants and reduced salinity

Westphalen *et al.* (2005) summarised the potential for water quality factors on the Adelaide metropolitan coast to act as causes for seagrass loss. Toxicants other than freshwater (including heavy metals, pesticides and petrochemicals) were largely discounted as factors that have caused metropolitan seagrass loss in Adelaide (Wilkinson *et al.* 2005, Bryars *et al.* 2006a).

Freshwater inputs to the Adelaide metropolitan coast are substantial (~169 GL mean annual discharge, Wilkinson *et al.* 2005). However, the impacts of reduced salinity on seagrasses were found to be minimal (O'Loughlin 2004), and reductions in salinity due to freshwater inputs were found to be localised and/or short in duration (Kaempf *et al.* 2004). In terms of reef health, reduced salinity is unlikely to be a factor. Metropolitan reefs, particularly northern ones, are too deep and too far from shore to be affected (Bryars *et al.* 2006b).

5.2.2 Nutrients, turbidity and sedimentation

Eutrophication of nearshore systems has long been considered to be the major cause for seagrass loss (Short and Wyllie-Echeverria 1996) although there is limited direct evidence relating these processes (Short and Wyllie-Echeverria 1996, Seddon 2000). Similarly with reef systems, it is possible that elevated nutrients affect reef health, particularly in temperate macroalgal-dominated systems, but there are few data available.

Gulf St Vincent waters are considered oligotrophic (Steffensen *et al.* 1989), and the flora is thus adapted to cope with this situation. Therefore, even very slight increases in nutrients may have a profound influence on a system (Collings *et al.* 2006, Bryars *et al.* 2006a). Nutrient inputs to the Adelaide metropolitan coast include waste water treatment outfalls, stormwater and riverine inputs (Wilkinson *et al.* 2005a, Wilkinson *et al.* 2005b). There have been substantial improvements to nutrient loading and suspended solid discharge from waste water treatment following the decommissioning of sludge outfalls in 1993, and an ongoing program of improved water treatment and/or reclamation (Wilkinson e*t al.* 2005a). However, for the past 30 years the shallow (<5 m) near shore of Holdfast Bay has been shown to be relatively eutrophic, which has been coincident with major seagrass loss from Glenelg to Grange (Bryars *et al.* 2006a). While offshore nutrient levels remain low, nitrogen isotope data from seagrasses on the Adelaide coast indicate that nutrients from waste water treatment and industrial inputs are pervasive along the metropolitan coast (Bryars *et al.* 2006a). Reef systems on the metropolitan coast are likely to be subject to similar nutrient exposure, but the effects are largely unknown (although see Gorgula and Connell 2004).

In a review of the effects of sedimentation on reef systems, Airoldi (2003) states that rocky coasts are highly sensitive, but that the processes have rarely been directly examined. In general, heavy sediment loads result in shifts in reef composition to opportunistic species. This leads to a loss of canopy species and decreased density of grazers matched by increases in turf forming algae (Airoldi *et al.* 1995, Airoldi and Virgilio 1998). On the Adelaide coast, sediments appear to affect recruitment of late successional species, in particular large canopy-forming brown algae (Turner 2004). However, sedimentation rates are difficult to measure and are correlated with other factors, in particular wave action and depth, that will influence community structure (Airoldi 2003). Further, high sediment loads often correlate with increased nutrient levels and high turbidity in the water column. At West Island (Fleurieu Peninsula), Gorgula and Connell (2004) found that turf-forming algal cover increased under high sediment load, but that addition of nutrients to the sediment had a stronger influence, with nutrients alone having the greatest effect. Higher nutrient levels may not necessarily be accompanied by high turbidity and sedimentation.

However, turf algae act to trap sediments (Neumann *et al.* 1970), and increased growth can only enhance this capability.

Irving and Connell (2002) found that the effect of sedimentation varied relative to the light regime, with only slight negative influences under high light, but with more profound changes observed under reduced levels of light. This would suggest sedimentation influences should not be considered in isolation from turbidity.

Irrespective of the causes of seagrass loss on the Adelaide coast, their removal can have a critical effect on sediment stability and result in the modification of wave energy gradients (Short and Wyllie-Echeverria 1996, Fotheringham 2002). On the Adelaide coast, seagrass losses have primarily occurred in a broad strip close to shore (Westphalen *et al.* 2005), resulting in remobilisation of sediments which may have increased turbidity and/or sediment loads on adjacent reef systems.

It has been speculated in previous Reef Health studies on the Adelaide metropolitan coast that the loss (or lack) of robust canopy-forming species from the northern reefs is most likely due to a combined result of eutrophication, higher turbidity and/or sedimentation levels (e.g. Cheshire and Miller 1998a, Cheshire and Westphalen 2000, Greig 2000, Smith 2000, Turner 2004) that may be directly linked to terrigenous inputs and/or as a secondary result of seagrass decline. However, a direct causal link between any of these factors and reef status has not been fully established.

5.2.3 Other factors affecting reef status

Other than the effect of climate change, other impacting factors are best considered on a site-by-site basis.

Marine pest invasion is likely to correlate with proximity to sources and/or the frequency of vectors as most pests are found near the port and harbour facilities in which they first settled. Throughout the 2005 survey, only two reefs were found to have a pest species (*Sabella spallanzaii* on the Glenelg blocks and *Caulerpa racemosa* var. *cylindracea* on Semaphore Reef). Given that none of the reefs considered in this survey are within a harbour, the number of pests might be expected to be low. However, *Caulerpa taxifolia*, one of the top 100 'worlds worst' pest species is rapidly expanding in areal cover within the Port River (Westphalen and Rowling 2005). Inclusion of marine pests as part of any monitoring program should be considered essential as a simple component of good risk management, regardless of its utility as an index of ecosystem health.

Like marine pests, fishing pressure and related physical damage may relate to distance from boat ramps, although Hallett Cove, Second Valley, Cape Jervis and the Bluff are fishable from shore, as well as the perceived value of the site for particular species (commercial and/or recreational). More generally, recreational fishing pressure has increased in the last decade (ABARE 2006) while commercial operations have remained static or declined. The consequences of removing considerable numbers of higher order carnivores from marine ecosystems (in particular isolated reef systems) still needs to be more fully investigated for temperate systems. Further, the effect of physical damage, such as anchor scarring from recreational boating should also be considered.

5.3 Indices of reef status

Robust and foliaceous macroalgal taxa provide significant three-dimensional structure and thereby increase the complexity of reef habitats, the result of which is a diverse range of organisms associated with these communities. The opposite is generally true for assemblages lacking these larger taxa (Gee and Warwick 1994), and this is also known to have flow-on effects to other inhabitants of the reef, including fish (Carr 1994). However, additional indicators of reef health were developed to allow more robust interpretations of reef status than those based solely on macroalgal cover. None of the indices employed in the current study (Table 1) can stand alone, although some measures are certainly stronger and more reliable than others.

5.3.1 Cover indices

Canopy macroalgal cover is one of the more robust measures of reef health, as indicated by the consistent differences in cover of LIT functional groups observed across Adelaide's metropolitan coast since 1996. However, there are factors that could potentially confound this index, which relate to variation in macroalgal assemblages at different spatial scales, and in areas of different exposure etc. At most sites, the 1996, 1999 and 2005 surveys were not conducted at the exact same location on each reef for each survey and thus there is the potential for interpreting small-scale patchiness within reefs as being either:

1. a difference in algal composition between reefs within the same sampling period, or
2. a change in composition within the reef when comparing across sampling periods.

These confounding factors are compounded by seasonal differences in macroalgal cover that might occur if sampling is not undertaken at the same time of year. Seasonal fluctuations have been observed in a number of groups of macroalgae, generally related to their reproductive state (Edgar 1983).

The use of functional groups rather than species avoids much of the problem with small-scale spatial variation, particularly if a large number of samples are collected from each site. Across different reef systems, there may be substantial variation in species composition, but these organisms will generally fulfil ecological roles that are fundamentally the same and will thus tend to be similar in terms of morphology and frequently come from the same division. The use of functional groups can thus subsume a large degree of variability in species composition, but still consider ecologically relevant questions as to the functioning of the system.

There are a number of generalisations that can be made in terms of assemblages that may be expected on temperate reefs in South Australia (see Turner *et al.* 2006a). Across most of southern Australia, shallow reefs tend to be dominated by rich macroalgal communities. These assemblages often comprise a number of separately identifiable strata (based on size), although not all strata may be represented in any one stand (Shepherd and Sprigg 1976, Turner and Cheshire 2003). The uppermost stratum is comprised of robust brown taxa (Orders Fucales and Laminariales, Shepherd and Womersley 1981). Foliaceous representatives of all three macroalgal divisions often form a stipitate (sub) canopy that may be observed below the main canopy, or alternatively as dominants in areas unsuitable for the larger phaeophycean taxa (Shepherd and Sprigg 1976, Turner and Cheshire 2003). Similarly, smaller specimens (a few cm tall) may exist below the larger canopies, but are also capable of forming dense stands of 'turf'. Once established, turf beds have the ability to exclude larger taxa and thereby dominate patches of reef (Shepherd and Sprigg 1976, Kennelly 1987). The smallest of the macroalgae may only be a few millimetres tall and are often observed as an encrusting layer on the substrate (Shepherd and Sprigg 1976). Encrusting species are also able to dominate the substrate in areas less suitable for larger taxa (Dethier 1994). Alternatively, many can adapt to lower light conditions and survive even when overgrown by larger taxa (Cheshire 1985).

Large expanses in the cover of turf-forming algae have also been inferred as a potential indicator of reef degradation (see above). Like bare substrate, turfs form an important component of reef systems, particularly as an early coloniser of bare substrate (Littler and Littler 1984, Sala and Boudouresque 1997, Airoldi 1998, Baynes 1999).

Previous Reef Health studies have discussed the implication of mussels colonizing areas formerly dominated by larger canopy-forming algae (Cheshire and Westphalen 2000). Mussels have been observed to colonise large areas of reef following disturbance (Turner *pers obs*.). The ability of mussels to inhibit the establishment of robust brown macroalgae has been demonstrated experimentally (Smith 2000), at least over short timeframes (six months), although studies from temperate parts of the United States indicate that kelps are competitively superior, but limited by other factors (Witman 1987).

The proportion of bare substrate present in an area may also provide an indication of 'health'. Space is important in marine systems and competition for space is well documented in ecological studies (see points made in Turner *et al.* 2006a). Disturbance events (such as storms) play an important role in helping to maintain reef diversity through creation of space for new organisms to colonise.

Given that space is generally a limiting factor in temperate reef environments (Connell and Keough 1985, Butler and Chesson 1990), the persistence or expansion of bare substrate for extended periods, such as that observed on Horseshoe Reef across more than one Reef Health survey, may be indicative of ongoing disturbance. This disturbance may take the form of high levels of mortality (possibly resulting from pollutants or smothering by sediments, or high grazing pressure), or alternatively as a result of disruption to natural recruitment processes (impacts to reproduction, decreased survivorship of propagules). High grazing pressure (e.g. the formation of urchin barrens, Jones and Andrew 1990) is considered a natural process and is commonly observed to have significant influence on the biotic structure of reefs in the eastern states of Australia and the Pacific Northwest. However, this process appears to be of less significance in South Australia where urchin barrens are rare (Jones and Andrew 1990).

Unfortunately, the layered structure of macroalgal communities makes the other cover indices (turf algae, mussels and bare substrate) less useful, as it is certain that measures are underestimated at sites with dense cover. While this has been countered by leaving all sites below threshold levels with 'null' responses, the actual information value in these parameters is reduced, as they are only employed in a few reefs.

5.3.2 Abundance indices

Both of the fish abundance indices, site-attached fish and blue-throated wrasse, are likely to be significantly influenced by conditions at the time of the survey. Fish behaviour is known to change depending on environmental factors including visibility, water movement and time of day. Results obtained during the 2005 surveys should be no more or less prone to these factors than any other survey. In particular, we postulate that fish abundance is influenced by a combination of the amount of locally available habitat, fishing pressure and overall habitat quality.

Highly modified reef areas generally received poor scores for the site-attached fish index, except where the habitat was isolated, in which case the reef appeared to act as a fish aggregating device (e.g. Glenelg Dredge and Barge). In contrast, apparently healthy reefs did not always have high scores for this index, which maybe linked to recreational fishing pressure. Evidence for this can be seen on the central metropolitan coastline where higher than average (for the area) fish numbers on Noarlunga Reef are possibly a consequence of its protected status.

For the abundance of invertebrate predators, quadrat data were the chief data source, meaning that a single average could be obtained for each site that is less related to the conditions at the time of measurement (relative to fish abundance scores). Conversely, like blue-throated wrasse, there may be specific invertebrate predators that provide a more robust indication of reef status (i.e. are there any keystone species?). Thus there is a need for a greater understanding of the composition and role of invertebrate predators in temperate reef systems.

5.3.3 Invasive species

The presence of marine pests was not a particularly informative index as this returned a value of 'null' in most sites. However, information on a select group of marine pests should form part of reef health surveys as good risk management. Invasive taxa were recorded on only two reefs: *Sabella spallanzanii* on the Glenelg Blocks; and *Caulerpa racemosa* var. *cylindracea* on Semaphore Reef (although *C. racemosa* has yet to be officially declared a pest). This index is not, by itself, an indicator of reef status – the lack of invasive species does not imply good condition.

Caulerpa racemosa var. *cylindracea* occurs naturally in Western Australia, but there are several strains of this species, with a Mediterranean form considered highly invasive (Verlaque *et al.* 2000). This alga was first discovered in the Port River in 2001 (Womersley 2003) where it is rapidly becoming a significant component of the system (Westphalen and Rowling 2005). From the Port River, *Caulerpa racemosa* var. *cylindracea* appears to be spreading to sheltered locations along the metropolitan coast. The status of this alga as a threat to local systems has yet to be confirmed.

S. spallanzanii is potentially subject to management as part of the National System for the Prevention and Management of Invasive Marine Species, an intergovernmental agreement across the commonwealth, states and territories. Under the National System a target list of problematic pests is to be developed, the members of which are subject to management. *S. spallanzanii* is currently on the list, although its status as a problematic pest is yet to be confirmed (Parry *et al.* 1996)

5.3.4 Sedimentation

The sedimentation index requires better sampling rather than the subjective assessment currently employed, although estimates of sedimentation are difficult to obtain (see Airoldi 2003). Quantitative sediment sampling was not a practical consideration within the current survey protocol, which required that each reef be sampled within a single working day. Robust measures of sedimentation rates require a substantial and prolonged (i.e. seasonal) commitment of resources at each location.

5.3.5 Richness indices

The merits of species richness as an indicator of system status have long been argued, as previously discussed. Macroalgal species richness is confounded by obscuring layers of canopy and the combining of taxa within functional groups. Estimates of richness of mobile invertebrates are more straightforward, but both measures are still plagued by issues related to seasonal changes in population size. While richness may be important ecologically, its utility as an index is lessened because it is necessary to produce a species list. Inevitably, this requires destructive sampling and substantial taxonomic skills that are not always available.

5.3.6 Summary index

The average across all indices is probably a better indicator than any of its component contributors, although the use of an average will tend to shift values toward the middle of the range, particularly if a large number of values are involved in its calculation. The relatively large number of reefs in the caution category may be a product of this to a certain extent.

None of the indices can stand alone as an indicator for reef heath, which is why several factors were considered collectively. The notion of reef health in temperate systems is relatively new and the development of metrics aimed to indicate reef status is problematic. Open to debate are such issues as what these indices should be; how the data for each index should be collected, manipulated and interpreted; and how the various reefs are scored. This document is intended as a catalyst for this process.

Some indices, in particular macroalgal cover and site-attached fish, would appear to be better than others although both are inter-related (i.e. low macroalgal cover is likely to result in lower site-attached fish). There was no discernible pattern to mobile predator abundance, species richness of mobile invertebrates, or species richness of macroalgae (outside those sites with 100 index value for cover).

The indices were chosen as they were considered to be important factors in determining the health of reefs. However, some did not, in fact, influence most of the overall scores (notably bare substrate, mussel and turf covers, and high levels of sedimentation). Turfing macroalgae at the Glenelg Dredge, Glenelg Blocks and the seaward site at Horseshoe Reef had scores of zero (i.e. greater than 40% turf cover), but only the last was scored as in Poor Condition (the other two were Caution Recommended). The two inner sites on Noarlunga Reef had high areal covers of mussels, but again only one of these (Noarlunga South Inside) was rated Poor Condition. Large areas of bare substrate occurred on the southern inside and deep part of Noarlunga Reef and also on the inside of Horseshoe Reef, Semaphore and to a lesser extent, the deep site at Aldinga Reef. Only three of these five sites rated Poor Condition. Finally, high sedimentation was probably the most useful of this group, occurring at seven sites, including four which scored as Poor Condition (however, as previously discussed, high levels of sedimentation can occur naturally near river mouths and estuaries). Indices for bare substrate, mussel and turf covers, and high levels of sedimentation, as well as invasive taxa, were null in the majority of instances. The informative value of these indices and their contribution to the overall index is thus low. It is worth noting that reconsideration of the overall average reef index without these values altered the status of only two borderline reefs, Broken Bottom and Noarlunga South Inside, both of which shifted from the Poor Condition to Caution Recommended category.

6 Conclusions and Recommendations

Generally, the north to south gradient in reef health observed across metropolitan reefs in 1996 and 1999 was continued in 2005. Based on macroalgal functional group composition and cover, northern metropolitan reefs (from Semaphore to Broken Bottom) have remained in poor condition, and were dominated by red foliaceous and turfing macroalgae. There were signs of further decline on central metropolitan reefs (Seacliff to Southport), in particular Horseshoe Reef and some sites on Noarlunga Reef, with a loss of robust brown macroalgae, the establishment of mussel mats, and the creation of large areas of bare substrate in some instances. Southern reefs (Moana and Aldinga) remained mainly healthy, retaining most but not all, of their robust macroalgal canopy.

Similar analyses of macroalgal cover and composition at sites surveyed during 2005 on Yorke Peninsula and Fleurieu Peninsula showed reefs that were generally healthy, particularly when compared to metropolitan reefs. However, there was a high level of variability within regions, including some sites (notably Point Souttar and Point Riley on Yorke Peninsula) that had relatively low cover of canopy macroalgal species.

Ten additional health indices were developed in order to obtain a more robust indication of reef status. These were averaged to give an overall score for each reef, and then grouped into one of three categories (Poor, Caution Recommended and Good). Reconsideration of all sites using all these indices revealed a more complex situation than macroalgal functional group cover. A large number of sites across the metropolitan region fell into the Caution Recommended category, even within what was considered the healthier southern zone when assessed on macroalgal cover alone. Similarly, a few sites on the Fleurieu Peninsula coast (Granite Island and Port Elliot) were rated as Caution Recommended, while four sites on Yorke Peninsula rated as either Caution Recommended (Troubridge Point and Cable Hut Bay) or Poor Condition (Point Souttar and Point Riley).

The indices employed are not perfect and were variously informative, with the summary average or all-over score probably being the most useful. Notwithstanding, the use of a range of indices targeting different ecological aspects of reef ecosystems has led to a better understanding of the nature and complexity of these communities. The results and interpretations presented in this report highlight the difficulty associated with producing a robust but practical approach to assessing reef health. Further development of the indices will benefit from more targeted data collection for each index. However, the potential for alternative indices should also be considered and there is a need to collect a wider variety of physical data on reefs, with wave energy and sedimentation levels probably the most important.

In examining the overall health of reef systems, it would be useful to develop a greater conceptual understanding of the linkages between the biological assemblages and their broader environment. In its simplest form, this would allow for a more confident appraisal of the sorts of communities that are likely to inhabit different environments. In turn, this would provide a paradigm to better elucidate differences between natural and anthropogenic gradients.

Additionally, different types of putative impact should be targeted, including: industrial areas like Whyalla and Port Pirie; reefs in proximity to various coastal developments such as marinas or aquaculture operations; and reefs subject to different levels of extractive activities, such as fishing and mining. Such data would allow the metropolitan systems on the Adelaide coast to be placed in an appropriate context. It would further strengthen our ability to differentiate between impacted reefs and those with naturally low algal cover, expand our knowledge of what constitutes a 'healthy' reef, and assist in the development of management and remediation strategies for reef systems.

The following recommendations are based on the above findings:

- Baseline data needs to be extended to other reefs across South Australia (Eyre Peninsula, West and Southeast coasts). A range of sites including near pristine and putatively impacted areas should be included;
- Monitoring attention should be given to areas of high conservation value (including marine protected areas) as well as those areas likely to be subject to human impact;
- Further (and more focused) monitoring should be conducted of sites with 'Caution Recommended' ratings;
- Future investigations should collect data that are more comprehensive with respect to physical parameters, which will allow for greater predictability of the types of biotic assemblages that may well be expected under natural conditions;
- The indices need to be further refined, and preferably augmented with data on keystone species. 'Indicator' invertebrate species are worthy of further investigation in this respect;
- The potential influence of climate change on reef ecosystems needs investigation;
- The role of seagrass loss off Adelaide as an agent in reef health should also be considered;
- The development of a model linking biotic and physical data from reefs needs to be developed. Such a model would increase our understanding of what constitutes a healthy reef, and allow predictions (which can be tested) about likely impacts from disturbance. For example, a granite substrate in an exposed position should support a predictable macroalgal community that will differ from a sheltered sandstone substrate. The response of these reefs are likely to differ greatly to disturbances such as local dredging creating a sediment plume.

- Finally, more resources need to allocated to increasing the capacity for community based reef-monitoring initiatives (e.g. Reef Watch) as a cost effective method for increasing the volume of information that can be collected. Testing the validity of data collected by community based programs forms an important component of the surveys being carried out in late summer/autumn of 2007, as part of the Reef Health program.

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Appendix A: *In situ* sampling methodology used in the program

These methods were designed to collect information about a variety of organisms associated with reef habitat. Each was conducted by divers on SCUBA, who followed a 50 m transect laid at the start of the survey. Where multiple methods were undertaken on a single dive, they were carried out in the order listed below to maximise diver efficiency and minimise confounding issues such as altering fish behaviour. A key element to the development of any method is that it must remain relatively simple to allow consistency of application between different divers/surveys.

Laying of the transect line

To lay the transect line, divers descended to the predefined depth and commenced reeling out the survey tape in a predetermined direction, following the depth contour. The location of the transect line determined what was included or excluded from the survey. It was therefore important that in placing the line, divers satisfied the following criteria:

- Depth of the transect line was kept relatively constant, with no more than two metres difference between the minimum and maximum.
- The transect remained on reef habitat for its entire length unless this was impossible (e.g. if the reef is smaller than the length of the transect line – 50 m).
- Within the above constraints, the line was laid relatively straight (although diversions were sometimes necessary to avoid large obstructions and/or to maintain the appropriate depth).
- Actual placement of the line was haphazard, and no attempt made to include or exclude any taxa or features (except as described above).
- Where two teams entered the water at the same locality, they headed off in roughly opposite directions depending on the size of the reef.

Basic habitat survey

The sampling method was designed to obtain a broad overview of the site environment by examining the physical structure of the reef.

A diver swam the length of the 50m tape a couple of metres above the substrate, in order to observe the macroscopic structure of the reef. Records were made for all parameters listed in Table 9, and annotated with additional information where appropriate.

Table 9. Parameters used to describe the reef environment for the basic habitat assessment

Parameter	Definition
Composition	The substrate comprising the reef. Examples include natural materials such as granite, limestone, or calcarenite, as well as artificial structures like concrete, tyres, and wrecks.
Form	Description of how the above is arranged on the reef, examples include consolidated masses, boulder fields, or in the case of artificial structures, a regular arrangement of structural units.
Relief	An indication of the relief of the reef was obtained using the height of the reef above the sea floor. Minimum, maximum and the average height along the transect line is recorded to provide an indication of the range.
Profile	The aspect of the reef at the location of the transect line. Examples include horizontal, vertical, or sloping (include angle).
Sedimentation	The presence of sediment on the reef was qualitatively defined using the following four categories. High – fine silts and sediments are obvious as a layer covering the reef biota. Moderate – absent from larger taxa but visually obvious on the substrate, sediments are resuspended when the diver waves their hand near the substrate. Some – Sediment is present but not in sufficient quantities to produce noticeable plumes when a hand is waved over the substrate. Minimal – Very little sediment is observed, and what is there is bound to the substrate and biotic complex.
Rugosity	Structural complexity of the reef was estimated using a 3 m long piece of metal chain, which was moulded to the profile of the reef. Ten replicate measurements were made at 5 m intervals starting at the 5 m mark on the transect line. For each measurement the chain is laid along side the transect line and pressed down to follow the substrate. The length of the transect line that the chain spans is then measured and recorded on the datasheet. Due to the time requirement of this component, it was sometimes undertaken in conjunction with the slower benthic methods.
Habitat	Brief description of the biotic composition of the reef (e.g. macroalgal canopy dominated, red algal community, urchin barrens).
Depth	Average depth of the transect line.
Visibility	Visibility in metres at the site on the day of the survey.
Turbidity	Qualitative assessment of suspended sediment in the water column.
Direction	Direction of the transect line from the starting point expressed as a compass bearing.

Pest species assessment

The pest species survey was designed to be a rapid assessment for identifying pest species on the reef. Information was collected on both known invasive species and naturally occurring taxa that may be an indicator of underlying problems (Table 10).

Using the same transect line as the other surveys, the diver swam slowly, sweeping from side to side along the line specifically searching for all taxa on the pest list. In the event that a target taxon was observed, the diver made notes on abundance and areal cover of the taxon. For certain species (as identified in Table 10), a sample was also collected for later confirmation.

Table 10. Taxa included in the pest species survey.

Species	Exotic	Collect	Notes and current South Australian distribution where known
Caulerpa taxifolia	Yes	Yes	Established in Port River, some effort at eradication
Caulerpa racemosa	Yes	Yes	Established on northern metropolitan coastline and several boat harbours
Undaria pinnatifida	Yes	Yes	Not recorded in SA, but established in Victoria and Tasmania
Asterias amurensis	Yes	Yes	Not recorded SA, but established in Victoria and Tasmania
Sabella spallanzanii	Yes	No	Established on northern metropolitan coastline and several boat harbours
Musculista senhousia	Yes	No	Intertidal and subtidal habitats to a depth of 20 m
Ciona intestinalis	Yes	No	Established in Port River and some boat harbours
Carcinus maenas	Yes	No	Widespread
Ulva sp.	No	No	Can become a nuisance in areas impacted by high nutrient input
Brachidontes rostratus	No	No	Observed to colonise large areas of reef following disturbance

Pelagic fish and other large mobile animals[7]

This sampling was undertaken immediately after laying the tapeline and before the slower benthic procedures in order to minimise changes in animal behaviour due to the presence of divers in the water.

Prior to starting the transect the diver wrote down the names of any taxa observed during decent and laying of the line so as to reduce the requirement for this during the actual survey when the diver needed to be scanning for fish.

On commencing the survey, the diver swam along the transect line at a slow regular rate, just above the vegetation. The rate was as slow as possible but without stopping so as to avoid previously counted fish behind the diver from overtaking. Divers observed the arc in front of them, out to a distance of 2.5 m either side of the line and recorded the number and size of each species present within the designated area.

Organism sizes were scored into a series of classes based on total length at intervals of 2.5 cm (from 2.5 cm to 15 cm) and 5 cm (from 15 cm and above, with one additional size class of 37.5 cm collected for historical reasons). Sightings were recorded using tally marks on a waterproof survey form pre-ruled with columns for all size classes. In the case of larger fish, the size as well as the tally was recorded in the final column (Table 11). A scale marked on the margin of the survey form was used to help calibrate size estimates.

[7] This methodological description is adapted from Edmunds and Hart (2003).

Table 11. Example data entry for pelagic taxa

Size class (inches) (cm)	1 2.5	2 5	3 7.5	4 10	5 12.5	6 15	8 20	10 25	12 30	14 35	15 37.5	16+ 40+
Silver drummer		I		III			II					III @ 50cm, II @ 40cm
Magpie perch			III	IV								
Old wife			XII		III							

Divers needed to remain aware of any easily recognisable, previously sighted individuals to ensure that each individual was only recoded once during the survey. If in doubt, individuals were recorded, meaning there was a tendency to over- rather than under-count. All staff employed in fish surveys undertook training to firstly identify fish species, but also assign them to appropriate size classes.

In the event that the diver observed a large aggregation of a single species, an estimate was made of total abundance and recorded against the size class(s) for the group.

Characteristics of unidentified taxa were noted to facilitate *post hoc* identification using available texts, and or in consultation with other divers.

Cryptic fish and larger non sessile invertebrates[8]

This method was used to identify fish and other large non-sessile taxa that tended to be at least partially concealed by reef vegetation, or which occurred in crevices and under overhangs. Surveys were conducted along the same 50 m transect as the other surveys. Before starting the survey the diver determined an easy method of accurately gauging a 1 m distance to the side of the transect line. In many cases, this was the distance from their outstretched fingertip to opposite shoulder buckle, or similar.

Divers searched the substratum for large mobile invertebrates and cryptic fishes within the 1 m wide section on the shoreward side of the transect line. Where necessary, canopy algae were swept aside using both hands, and attention paid to small caves and crevices.

Counts (but not sizes) of all larger non-sessile invertebrates (>5 cm), along with cryptic or sedentary fish (Table 122) were recorded on the data sheet. Smaller and more numerous taxa, along with sessile invertebrates such as ascidians, were recorded using the benthic quadrat method described later (page 81).

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[8] This methodological description is adapted from Edmunds and Hart (2003)

Table 12. Megafaunal invertebrate and cryptic fish groups to be recorded during the survey.

Megafaunal invertebrates (>5 cm in size)	Crabs, rock lobster, hermit crabs, gastropods, bivalves, octopus, crinoids, sea stars, urchins, sea cucumbers
Cryptic fish families	Parascyllidae, Urolophidae, Muraenidae, Sygnathidae, Scorpaenidae, Apogonidae, Pempherididae, Gnathanacanthidae, Pomacentridae (juv), Bovichtidae, Tripterygiidae, Clinidae, and Gobiidae

The most specific taxon possible was used to identify invertebrates. Unknown or unidentifiable invertebrates were collected and taken to the surface for further examination. Unknown cryptic fish were sketched or photographed. In cases where the diver was only able to catch a glimpse of the organism (as it fled), these were recorded as unidentified.

Line Intercept Transects (LIT)[9]

The LIT transect was 20 m in length and commenced at the start of the main transect line, using it as a guide. In contrast to the method used in tropical systems (English *et al.* 1994), a weighted one metre stainless steel ruler was placed consecutively along the transect line in order to pin vegetation beneath it, as described below (based on Turner 1995).

Starting at the beginning of the transect line, the weighted ruler was placed as near as practical to the guide tape. To do this, the ruler was held above the line and lowered quickly into position. This ensured that the macroalgae was pinned, and did not slip out from under the ruler. Lowering the ruler was done in a relatively haphazard manner with no effort made to include or exclude specific individuals. With the ruler placed, the diver immediately took a mental snapshot of the pinned assemblage in case of movement caused by surge.

Divers noted the transitional points between one taxa and the next along one edge of the ruler. To do this the diver identified the taxon present at the beginning of the ruler and the point at which there was a transition to another taxon (Figure 18). The code for this taxon and transition point was then recorded on the data sheet (Table 13) and the process repeated until the end of the ruler. Divers recorded the organism encountered to the most specific taxonomic level possible. For taxa that might (during different life stages) fall within different functional categories, the applicable life form code was placed in brackets (e.g. *Sargassum* sp. may have the life form code BTURF, BFOLI or BBRANCH depending on its size). Unidentifiable taxa were given a unique but descriptive code name and collected for subsequent formal identification. Subsequent sightings of the same taxon were given the same name.

[9] Method based on Turner (1995)

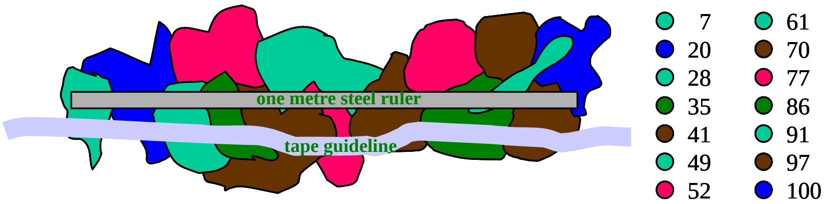

Figure 18. An example Line Intercept Transect and resulting data.

Table 13. Example of an LIT datasheet.

Metre	Transition	Taxon	Notes
1	23	*Ecklonia radiata*	
	37	TURF	Mixed species
	44	*Sargassum fallax*	
	59	BLOBE	*Padina* spp? (Bag A)
	72	*Cystophora subfarcinata*	
	100	*Zonaria spiralis*	
2	13	*Zonaria spiralis*	
	48	*BLOBE*	

Transitions were only recorded where there was a change from one taxon to another, and not for each individual plant / animal. Additionally, transitions were only recorded where the length of cover of a taxon was 3 cm or more. Smaller transitions were ignored for pragmatic reasons.

Where the line spanned a crevice in the substrate, data were only recorded where the distance between the ruler and biota was < 20 cm. Otherwise, the transition is recorded as missing data and given the code DDD.

On completion of the one metre segment the ruler was raised and relowered for the next segment along the transect line. This process continued until a continuous 20 m LIT had been completed.

Benthic quadrats[10]

Square quadrats measuring 50 × 50 cm were used to further sample macrophyte abundance and to record the abundance of small and sessile invertebrate taxa not covered during the cryptic fish and invertebrate survey (see above).

The quadrat was strung with 7 × 7 perpendicular wires equally spaced across the quadrat. This created 49 points located at the intersection of the crisscrossing wires, and the corner of the

[10] Point intercept method based on Edmunds and Hart (2003)

quadrat that is adjacent to and closest to the start of the transect line was used as the fiftieth point. Sampling of the quadrat was undertaken using three discrete counts as described below.

First, the canopy assemblage was sampled by recording the taxon present under each of the 50 points. Data were recorded as a cumulative total for each taxon (Figure 19).

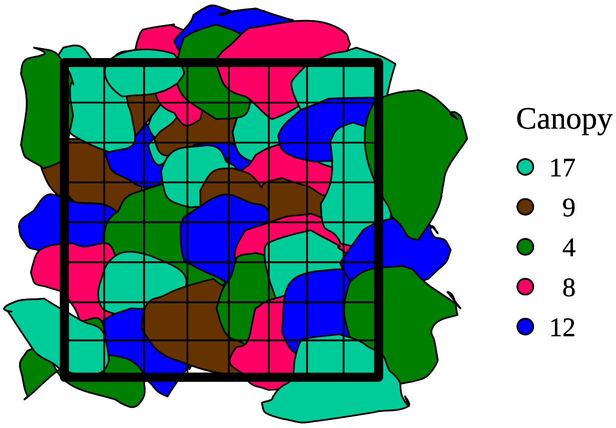

Figure 19 Using the point intercept method to quantify the canopy macroalgal assemblage.

Next, all sessile taxa (including understorey) and substrates were then scored using a similar method except that as each taxon was scored, it was (where possible, i.e. if not encrusting) pushed aside to reveal lower levels that are then scored.

Note: The process of 'peeling back' each taxon after it was scored meant that several taxa may be recorded under each point leading to a total score greater than fifty. The method is functionally equivalent to scoring a total percentage cover value for each taxon as if it occurred in isolation of any others (Figure 20).

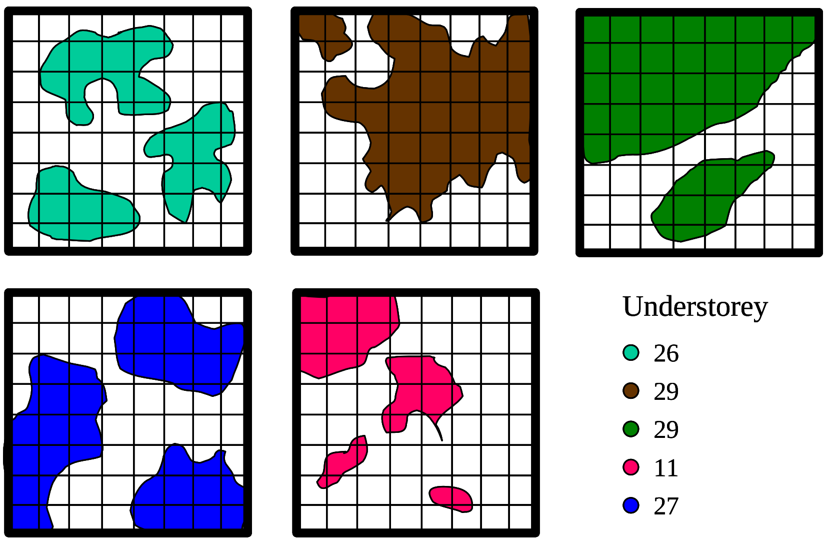

Figure 20. Layered sampling of the understorey using the point intercept for each taxon.

Finally, abundances for all invertebrates within the quadrat were also recorded. For this part, the perpendicular wires were ignored and estimates made for the entire area within the bounds of the quadrat. Counts were made for discrete organisms and percentage cover estimated for colonial and amorphous taxa. When estimating percentage cover, divers used the knowledge that each of the 64 squares in the quadrat covers approximately 1.5% to help accurately gauge area (either in terms of the accumulated number of squares or proportion of an individual square).

The same rules regarding the use of the most specific taxon and dealing with unidentifiable taxa that were used for the LIT method were also applied here.

Appendix B: Reef Health program standard survey protocol

The Reef Health standard survey protocol comprise the six methodologies described in Appendix A and is designed to be completed by a pair of divers in around one hour (Table 14). The order in which the methods were performed was designed to facilitate buddy contact between divers and efficiently carry out all required tasks. At each reef to be surveyed, four complete sets of surveys were undertaken to sufficiently characterise the site.

Table 14. Estimated times to complete survey tasks using proficient staff

Task	Direction	Diver A	Diver B	Time
1	Descend	Get to the bottom, clip on line and prepare first run		5 mins
2	Outward	Habitat survey	Reel out the transect tape, taking one quadrat and the chain to leave at the far end.	5 mins
3	Return	Pest survey	Pelagic fish survey	5-10 mins
4	Outward	Line intercept transect	Cryptic fish and invertebrates	35 mins
5	Return	Benthic quadrats	Benthic quadrats / Chain / Reel in line	20 mins
6	Ascend	Gather up all equipment and surface		5 mins

At the beginning of the survey, divers divided the tasks evenly between them and ran through a checklist to ensure that all equipment was ready. When diving from a boat, a buoy line was deployed to mark the starting point for the transect line and a GPS coordinate recorded.

Divers descended to the bottom and clipped any equipment not needed for the first two passes, to the buoy line and then attached the 50 m tape measure.

Pass 1 – outward swim

At the commencement of the survey one diver reeled out the transect line in the pre-determined direction, following the depth contour. While reeling out the line, the diver observed their surroundings and made a mental note of the fish taxa encountered. The second diver followed behind and gathered the information necessary to complete the habitat survey (page 76). The second diver also carried one quadrat and the chain to leave at the 50 m mark.

Once the transect line was positioned, both divers stopped for a few minutes to allow fish behaviour along the line to return to normal. During this time, the diver that was going to conduct the pelagic fish survey (page 78) prepared the data sheet and noted the names of all taxa spotted on the outward swim in order to save time during the actual survey. The second diver completed all information on the habitat survey form except for rugosity (as this was conducted later using the chain). Afterward, the second diver prepared for the pest species survey.

Pass 2 – return swim

Divers swam back along the transect line towards the start point, with the leading diver engaged in the pelagic fish survey while the trailing diver remained some distance behind to undertake the pest species survey. On returning to the start of the transect line, divers completed their respective survey forms and prepared for the benthic surveys.

Pass 3 – outward swim

Depending on the nature of the reef habitat, the order and timing of the benthic surveys often needed to be adjusted. Nominally, one diver commenced the cryptic fish and invertebrate survey (page 79) while the second diver undertook the line intercept transect survey (LIT, page 80). In order to do these methods simultaneously divers worked on opposite sides of the transect line.

At the twenty-metre mark, the LIT survey was completed and the relevant diver returned to the start of the transect line and commenced benthic quadrats (page 81) in sequence, starting at 2 m.

Pass 4 – return swim

On the return swim, the diver that previously did the cryptic survey began measuring rugosity every 5m starting at the 45 m mark using the chain. In addition, the diver also stopped to do benthic quadrats where necessary, starting at the 50 m mark. Eventually the two divers met at some point along the line, at which time one diver took responsibility for reeling in the line while the other complete the chain measurements.

Finally, divers gather up all equipment and ascended to the surface together.

Post dive debrief

On completion of the survey, divers undertook a post dive debrief. During debriefing, divers examined their datasheets and ensured that all relevant data had been recorded and that the information was legible. Annotations and comments were made as appropriate.

If unrecognisable fish were noted during the survey, an immediate attempt was made to identify the species using available references and the divers' notes. Similarly, organisms collected during the benthic surveys were examined and an attempt made to identify them. Where an organism could not be identified, it was given a permanent unique code from the Reef Health database and the specimen preserved for later expert identification. In all cases, data sheets were annotated with the outcome of these examinations.

Appendix C: Reporting codes used during data analysis

Table 15. Examples of the taxa represented by each of the life forms used during the Reef Health surveys. Reporting codes are those used in the current document, the remainder of the table is based on Cheshire and Westphalen (2000).

Reporting code	Life form code	Description	Representative genera
Robust brown algae	BRBRANCH	Brown robust algae with highly branched habit (blades not much broader than they are thick)	*Cystophora, Sargassum, Caulocystis, Acrocarpia, Scytothalia, Seirococcus, Xiphophora*
	BRFLAT	Brown robust algae, large flattened blades (much broader than thick), not membranous but leathery	*Ecklonia, Durvillaea, Macrocystis*
Foliaceous brown algae	BRFOLI	Brown foliaceous algae	*Halopteris, Cladostephus, Lobospira*
	BRLOBE	Brown lobed algae	*Zonaria, Padina, Lobophora*
	BRMEM	Brown membranous algae	*Scytosiphon*
Foliaceous red algae	RFOLI	Red foliaceous algae	*Plocamium, Phacelocarpus, Nizymenia, Gelidium, Pterocladia*
	RLOBE	Red lobed algae	*Peyssonnelia*
	RMEM	Red membranous algae	*Gloiosaccion*
	RROB	Red robust algae	*Osmundaria, Lenormandia*
Turfing & encrusting	BRENC	Brown encrusting algae	*Ralfsia*
	RCORAL	Red coralline algae	*Corallina, Metagoniolithon,*
	RENC	Red encrusting algae	*Sporolithon*
	TURF	Turfing algae (all colours)	*Sphacelaria, Ectocarpus, Ceramium, Cladophora*
Not reported	GFOLI	Green foliaceous algae	*Caulerpa, Cladophora, Bryopsis Chaetomorpha, Apjohnia, Codium,*
	GLOBE	Green lobed algae	*Dictyosphaeria, Avrainvillea*
	GLUMP	Green lumpy algae	*Codium*
	GMEM	Green membranous algae	*Ulva*
Animals	For purposes of comparison, all sessile animal taxa were aggregated		
Bare substrate	The presence of uninhabited reef substrate was also recorded		

Table 16. Fish species considered to be site-attached for purposes of index calculation

Acanthaluteres brownii	*Enoplosus armatus*	*Parapriacanthus elongatus*
Acanthaluteres vittiger	*Eocallionymus papilio*	*Parma victoriae*
Achoerodus gouldii	*Gobiidae* spp.	*Pempheris klunzingeri*
Aetapcus maculatus	*Helcogramma decurrens*	*Pempheris multiradiata*
Aploactisoma milesii	*Heteroclinus johnstoni*	*Phycodurus eques*
Aplodactylus arctidens	*Meuschenia flavolineata*	*Phyllopteryx taeniolatus*
Apogonidae spp.	*Meuschenia freycineti*	*Pictilabrus laticlavius*
Aracana aurita	*Meuschenia galii*	Pipefish undifferentiated
Aracana ornata	*Meuschenia hippocrepis*	*Rhycherus filamentosus*
Austrolabrus maculatus	*Neoodax balteatus*	*Stigmatopora nigra*
Blennidae spp.	*Nesogobius* spp.	Stinkfish undifferentiated
Bovichtus angustifrons	*Norfolkia clarkei*	Syngnathidae undifferentiated
Bullseye undifferentiated	*Notolabrus parilus*	*Tetractenos glaber*
Cheilodactylus nigripes	*Notolabrus tetricus*	*Tilodon sexfasciatus*
Chelmonops curiosus	*Odax acroptilus*	*Trachichthys australis*
Clinidae spp.	*Odax cyanomelas*	*Trachinops noarlungae*
Cnidoglanis macrocephalus	*Omegophora armilla*	*Vincentia conspersa*
Cochleoceps spp.	*Parablennius tasmanianus*	Wrasse undifferentiated
Cristiceps australis	*Parapercis haakei*	
Diodon nicthemerus	*Parapercis ramsayi*	
Dotalabrus aurantiacus	*Paraplesiops meleagris*	

Table 17. Mobile invertebrates used in index calculation

Agnewia tritoniformis	*Coscinasterias muricata*	*Pleuroploca australasia*
Allostichaster polyplax	*Cymatium parthenopeum*	*Prototyphis angasi*
Argobuccinium vexillum	*Dicathais orbita*	*Pterynotus triformis*
Buccinidae undifferentiated	*Fusinus australis*	*Ranella australasia*
Cabestana spengleri	*Jasus edwardsii*	*Semicassis semigranosum*
Cabestana tabulata	*Lepsiella flindersi*	*Sepia apama*
Cassis fimbriata	*Mitra glabra*	*Sepioteuthis australis*
Charonia lampas	*Murex* spp.	*Uniophora granifera*
Charonia powelli	*Muricopsis umbilicatus*	
Chicoreus denudatus	*Octopus tetricus*	
Conus anemone	*Penion mandarinus*	
Conus rutilus	*Penion maxima*	

Appendix D: Site descriptions for reefs included in the 2005 surveys

Site descriptions given here are generally based on information collected during the 2005 field survey program, unless otherwise indicated in the text. In total, 39 sites were surveyed using the methodology described in Appendix A. Sites were divided into three groups according to management areas (Adelaide Metropolitan coast, Fleurieu and Yorke Peninsulas). Within these groupings, site order corresponds to increasing distance along the coastline away from Adelaide.

A.i Adelaide metropolitan reefs

Table 18: Site description for reefs surveyed in the Adelaide Metropolitan area during 2005. Empty cells indicate a lack of data

Reef	Description	Composition	Relief	Exposure	Dominant biota
Semaphore	Broken bottom horizontal reef	Limestone	0.5 m	Low	Foliaceous red algae, sponges and ascidians. *Sargassum*, *Caulerpa* and *Caulocystis* are also common
Broken Bottom	Low profile horizontal broken bottom with a few boulders	Limestone	1 m	Low	Foliaceous red algae, sponges and the coral *Plesieastera*
Glenelg Dredge and Barge	Artificial reefs, established in 1985 by sinking two vessels off the coast of Glenelg	Metal	0 m	Low	Foliaceous and turfing red algae
Glenelg Blocks	Artificial reef comprised of four large concrete blocks interspersed with rubble and sand	Concrete	0 m	Low	Foliaceous and turfing red algae except on the corners and northern face where *Ecklonia radiata* grows abundantly
Seacliff Reef	Flat consolidated rock platform with small patches of sand	Limestone	1 m	Low	Sponges and *Sargassum* (mainly subgenus Arthrophycus). Also *Cystophora monilifera* and *Ecklonia radiata*

Reef	Description	Composition	Relief	Exposure	Dominant biota
Hallett Cove Reef	Approximately 50m offshore. One of the closest sites to the coast for this survey. It is a narrow undulating spur of rock rising 1 – 2 m above the adjacent sand.	Limestone	1 - 2 m	Low	*Ecklonia* and *Sargassum* with *Cystophora* being less abundant
Horseshoe Reef (inside & outside)	Formed from an arc of rock (like a horseshoe) with the open end towards the shore. On the seaward side, the reef drops from a steep platform to a series of broken but generally very flat expanses of stone that persist for some distance off shore. Toward shore, the reef becomes narrower and steeper comprising more of a boulder field than a solid rock structure. The reef has moderate to high sediment loads	Limestone		Low	Red coralline algae and the mussel *Brachidontes rostrata* dominate the reef; there is only a sparse cover of *Ecklonia* and fucoids taxa
Noarlunga (all sites)	The entire reef is an Aquatic Reserve, however, the northern part of the reef (and the inside in particular) is a popular recreational SCUBA diving and snorkel site, and the intertidal areas are subject to heavy trampling when exposed at low tide. The majority of the reef is comprised of boulders and is subject to moderate levels of sedimentation. Both the inside and outside of the northern section as well as the inside southern section were recorded as sloping reefs at angles between 22.5°- 45° were as the outside southern section and the deep sites were recorded as horizontal reefs	Limestone	1 - 3 m	Moderate to high (depending on tide)	The northern outer part of the reef was dominated by *Ecklonia radiata*, whereas other sites had assemblages that were more open. The reef variously consisted of *E. radiata*, and several species of fucoid. Species of *Caulerpa* and turfing communities were also common as were large areas of the mussel *Brachidontes rostrata.*
Southport	This reef is comprised of a series of flat platforms with small patches of sand and occasional rocky outcrops	Limestone	1 - 2 m		*Ecklonia*, *Sargassum* and *Cystophora* dominate the canopy. A large bare area dominated by the sea urchin *Heliocidaris* was observed (described as an urchin barren).
Moana (inside & outside)	Moana consists of a band of gently sloping rock platform that abruptly falls away on the shoreward side to form a steep slope above the seafloor	Limestone	2 - 3 m		*Ecklonia* dominates the canopy with the occasional *Sargassum*, *Cystophora* and *Scaberia*. The understorey is composed primarily of red encrusting algae.

Reef	**Description**	**Composition**	**Relief**	**Exposure**	**Dominant biota**
Aldinga	Aldinga reef is comprised of a series of gently sloping rock platforms with occasional prominent outcrops. The deep site is primarily broken bottom with the occasional boulder whereas the shallow site is primarily a consolidated flat platform with the occasional boulder	Limestone			*Sargassum* along with sparse *Cystophora* and *Ecklonia* dominate the canopy. There is also a rich understorey comprised of red foliaceous algae and *Lobophora*

A.ii Fleurieu Peninsula reefs

Table 19: Site description for reefs surveyed around Fleurieu Peninsula during 2005. Empty cells indicate a lack of data

Reef	Description	Composition	Relief	Exposure	Dominant biota
Carrickalinga	This reef is located approximately 100 m off the coast at a depth of 4 - 5 m. The reef consists of a band of gently sloping rock platform, rising 1 - 2 m above the seafloor; however, there is the occasional boulder, which may rise to 3 m above the sand level.	Limestone	1 - 3 m	Low	*Sargassum* and *Cystophora* with only sparse *Ecklonia*.
Second Valley	This reef is similar to Carrickalinga, consisting of a band of gently sloping rock platform that gradually meets the sand at 8 - 10 m. The platform slopes at an approximate angle of 30-40°. It is generally a consolidated reef, with the occasional boulder.	Limestone		Low	*Sargassum*, *Cystophora* and sparse *Ecklonia*. *Scaberia* was present in sand patches.
Cape Jervis (North and South)	The north reef consists of a rock platform comprised of boulders. The south reef is similar but has a gentle slope with the occasional boulder. Both reefs have minimal sedimentation. Surveys were undertaken at a depth of 5 - 6 m.	Limestone	1 - 3 m	Low	A mixture of *Sargassum* and *Cystophora* dominate the canopy at the north reef while the understorey was comprised of red encrusting algae. Large brown algae including *Sargassum*, *Cystophora*, *Acrocarpia*, *Seirococcus* and *Ecklonia* dominate the canopy at the south reef. The understorey consists of red coralline and red encrusting algae.
West Island	This reef is comprised of boulders, which slope down at an approximate angle of 45° and abruptly meet the sand at 13 – 15 m. Sedimentation is minimal.	Granite		Low	*Ecklonia*, *Acrocarpia* and *Cystophora* while red coralline and red encrusting appeared in the understorey.
The Bluff	A prominent headland, which falls directly into the sea forming a reef. The reef is a narrow band of sloping rock (22.5 - 45°) with many large boulders. Due to high relief and exposed position, there is minimal sedimentation.	Granite	3 m		*Ecklonia*, *Scytothalia*, *Seirococcus*, *Acrocarpia*, ands some *Cystophora*. The understorey consists of red foliaceous and red encrusting algae.

Reef	**Description**	**Composition**	**Relief**	**Exposure**	**Dominant biota**
Granite Island	The reef is a continuation of the coastal formation. The reef is a narrow sharply sloping (45 - 67.5°) band of rock consisting of large boulders with minimal sedimentation. The rocky outcrop abruptly meets the sand at 10 m.	Granite	3 m		*Ecklonia, Scytothalia* and *Acrocarpia* dominated the canopy, while red foliaceous and red encrusting form the majority of the understorey.
Port Elliot	The reef off Pullen Island consists of several sloping (45°) rocky outcrops comprised of boulders. The reef falls away and meets the sand at 6 – 8 m. Sedimentation is minimal.	Granite	2 m		*Ecklonia* is the dominating alga in the canopy while the red algae, encrusting, coralline and foliaceous dominate the understorey.

A.iii Yorke Peninsula reefs

Table 20: Site description for reefs surveyed around York Peninsula during 2005. Empty cells indicate a lack of data

Reef	Description	Composition	Relief	Exposure	Dominant biota
Edithburgh Pool	Narrow band of flat rock platform 50 m offshore and found at a depth of 4 m. Sedimentation is high with a layer of silt covering much of the biota.	Limestone	0.5 - 1 m		Mixed fucoids and very short *Ecklonia* with patches of seagrass most likely *Posidonia sinuosa* occurring sporadically throughout the reef. Red foliaceous algae are also common.
Troubridge Point	Flat rock platform with small patches of sand at a depth of 5 m.	Limestone	0.5 - 1 m		Mixed fucoids and sparse *Ecklonia* in the canopy dominate the algal community and red foliaceous algae are prominent in the understorey
Point Yorke	The exposed side of the reef has a horizontal aspect and comprises very large boulders. The sheltered side of the reef is also horizontal but is comprised of smaller boulders. Sedimentation varies depending on exposure, minimal on exposed side and moderate on sheltered side.	Limestone	1 - 3 m	Low to High	Mixed fucoids and in certain areas also *Ecklonia*. The understorey comprises a mixture of red coralline, small browns and green algae.
Marion Bay	Horizontal rock platform comprised of boulders, crevices, ledges and cutouts. Minimal sedimentation.	Limestone	3 m	High	Canopy species such as *Cystophora*, *Sargassum*, and red coralline, *Lobophora* and *Plocamium* in the understorey. This reef also has a large invertebrate community.
Cable Hut Bay	Flat rock platform with patches of sand and occurs at a depth of 5 m.	Limestone			Large brown algae including *Cystophora*, *Acrocarpia* and *Ecklonia*, dominate the canopy while the understorey is composed of red encrusting, red coralline and red foliaceous algae.
Corny Point	Low lying broken bottom reef with a moderate level of sedimentation.	Limestone	0 m		A mixed fucoid community dominates the canopy while the understorey is dominated by red foliaceous, red turfing algae and *Botryocladia*.

Reef	Description	Composition	Relief	Exposure	Dominant biota
Point Souttar	Low lying broken bottom reef occurring at a depth of 4 m. High sedimentation level with much of the biota covered in a layer of silt.	Limestone	0 m		*Sargassum* is the dominant brown algae and red foliaceous is the dominant red algae. Patches of the seagrass *Posidonia sinuosa* occur intermittently throughout the reef. Burrowing holothurians occur in large numbers across this reef.
Wardang Island	This reef varies from a flat rock platform to a sloping (45°) rock platform comprised of boulders. Sedimentation is at a minimal level.	Limestone	0.5 - 2 m		*Cystophora* (primarily *C. expansa*) is the dominant canopy taxon along with *Sargassum*, *Osmundaria* and *Scaberia*. Red algae dominate the understorey.
Goose Island	The reef around this island has two sections; exposed and sheltered. The exposed side consists of a gently sloping rock platform with boulders. The sheltered side comprises of a broken reef with boulders. The exposed side has minimal sedimentation while the sheltered side has moderate sedimentation; a layer of silt covers the biota on the sheltered side.	Limestone	1 - 3 m		*Sargassum* and *Cystophora* dominate the canopy; also present are *Scaberia*, *Ecklonia* and *Caulerpa* species. *Lobophora* and red turfing algae dominate the understorey.
Cape Elizabeth	Flat rock platform comprising boulders found at a depth of 3 - 5 m.	Limestone	0.2 - 0.5 m		*Sargassum* and *Cystophora* with isolated patches of *Ecklonia* and *Scaberia*. Red turfing algae, *Caulerpa flexilis* and *Lobophora* constitute the understorey. Sponges and ascidians are abundant. The northern section of the reef appears well grazed.
Point Riley	Low -lying broken bottom reef found at a depth of 3 - 5 m with high sedimentation.	Limestone			*Scaberia*, *Sargassum*, *Lobophora* and red foliaceous taxa. Patches of sparse seagrass occur along the reef. *Dictyopteris*, *Botryocladia* and *Cystophora* are also present. Urchins and sponges are the common invertebrates.